ECONOMIC VALUE OF EAST AFRICA'S TRANSBOUNDARY WILDLIFE LANDSCAPES

A Natural Capital Assessment of Four Selected Landscapes and Assessment of the Current Trajectory

February 2022

CONTRACT INFORMATION

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DISCLAIMER

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Front Cover Photo

Elephants as seen from the Amboseli National Park with Tanzania's Mt. Kilimanjaro in the background. The Kitenden Wildlife Corridor is a critically important transboundary passageway between the two ecosystems in the Eastern African Plains. Credit: Adam Henson/USAID.

Back Cover Photo

The chimpanzee (*Pan troglodytes*) in Kibale National Park, one of the largest intact swaths of Afromontane forest in East Africa. Credit: Nathan Chesterman/EI.

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ACRONYMS

AWF	African Wildlife Foundation
BAU	Business-as-Usual
CFR	Central Forest Reserve
CICES	Common International Classification of Ecosystem Services
CN	Curve Number
COVID-19	Coronavirus Disease of 2019
CGIAR-CSI	Consultative Group on International Agricultural Research Consortium for Spatial Information
CMIP5	Coupled Model Intercomparison Project (Fifth Phase)
CWMA	Community Wildlife Management Area
DEM	Digital Elevation Model
DRC	Democratic Republic of the Congo
EAC	East African Community
ECOTRUST	Environmental Conservation Trust of Uganda
EI	Environmental Incentives
ENSO	El Niño-Southern Oscillation
FAO	Food and Agriculture Organization of United Nations
FEGS-CS	The Final Ecosystem Goods and Services Classification System
FEWS NET	Famine Early Warning Systems Network
FONERWA	National Fund for the Environment
GCA	Game Controlled Area
GDP	Gross Domestic Product
GIS	Geographic Information System
GPS	Global Positioning System
GVA	Gross Value Added
HDI	Human Development Index
HF	Horizontal Flow
HH	Household
HWC	Human Wildlife Conflict
HydroSHEDS	Hydrological data and maps based on Shuttle Elevation Derivatives at multiple Scales
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs
IUCN	International Union for the Conservation of Nature
LSU	Large Stock Unit
KEA	USAID/Kenya and East Africa
KFS	Kenya Forest Service
KNBS	Kenya National Bureau of Statistics
KWS	Kenya Wildlife Services
MoE	Ministry of Environment
MoU	Memorandum of Understanding
MtC	Megatons of carbon
NASA	National Aeronautics and Space Administration

NBS	National Bureau of Statistics Tanzania
NCAA	Ngorongoro Conservation Area Authority
NDR	Nutrient Delivery Ratio
NDVI	Normalized Difference Vegetation Index
NESCS	National Ecosystem Services Classification System
NFA	National Forestry Authority
NISR	National Institute of Statistics Rwanda
NP	National Park
NPP	Net Primary Productivity
NPV	Net Present Value
OBPE	Office for the Protection of the Environment (Burundi)
PA	Protected Area
PES	Payment for Ecosystem Services
PUD	Photo User Day
RCP	Representative Concentration Pathway
RDB	Rwanda Development Board
REDD	Reduced Emissions from Deforestation and Degradation
REMA	Rwanda Environmental Management Agency
RFA	Rwanda Forestry Authority
RUSLE	Revised Universal Soil Loss Equation
SDM	Species Distribution Model
SDR	Sediment Delivery Ratio
SEEA	System of Environmental Economics Accounting
TANAPA	Tanzania National Parks Authority
TAWA	Tanzania Wildlife Management Authority
TDGDP	Tourism Direct Contribution to GDP
TEEB	The Economics of Ecosystems and Biodiversity
TFCA	Transboundary Conservation Area
TRAFFIC	The Wildlife Trade Monitoring Network
UBOS	Uganda Bureau of Statistics
UN	United Nations
UNDP	United Nations Development Program
UNECA	United Nations Economic Commission for Africa
UNEP	United Nations Environment Program
UNFCCC	United Nations Framework Convention on Climate Change
UNWTO	United Nations World Tourism Organization
USAID	United States Agency for International Development
U.S. EPA	United States Environmental Protection Agency
UWA	Uganda Wildlife Authority
VFR	Visiting Friends and Relatives
WMA	Wildlife Management Area
WTP	Willingness to Pay
WTTC	World Travel and Tourism Council

EXECUTIVE SUMMARY

INTRODUCTION

Wildlife and wildlife habitats are critically important assets in East Africa and underpin the very essence of the region. In addition to their significant heritage contribution, its vast wildlife landscapes have provided an important comparative economic advantage for East Africa's development and will be vital in ensuring its resilience in the face of mounting pressures.

Wildlife landscapes comprise contiguous and interconnected ecosystems that support self-sustaining and genetically viable populations of larger animal species as well as relatively intact communities of plants and animals. These ecological systems are part of the region's natural capital, which is defined as the ecosystems and natural resource stocks that supply ecosystem services contributing to human wellbeing. In addition to supporting wildlife and tourism, these landscapes provide ecosystem services, such as carbon sequestration, flow regulation, water quality amelioration, sediment retention, and pollination, which provide benefits to people living within and beyond the landscapes.

Because landscapes often transcend national boundaries, conserving and managing natural capital requires transnational coordination. USAID's Economics of Natural Capital in East Africa Project seeks to improve the conservation and management of iconic and important transboundary East African wildlife landscapes by providing policymakers and advocates with information on their economic value.

The aim of the Economic Value of East Africa's Transboundary Wildlife Landscapes report (this report), and the study it describes (this study), is to produce a description and valuation of the wildlife and wildlife habitats of four selected transboundary landscapes in East Africa: the **Great East African Plains** of southern Kenya and northern Tanzania; the **Northern Savannas** of South Sudan, Uganda, and Kenya, and adjacent Mount Elgon; the **Albertine Rift Forests** along the Albertine Rift Valley of Burundi, Rwanda, and southwest Uganda; and the **Rweru-Mugesera-Akagera Wetlands** of northern Burundi, eastern Rwanda, and northwest Tanzania (Figure I).

CONCEPTUAL FRAMEWORK

This study assesses the value of four important transboundary landscapes that support significant wildlife populations in East Africa. These landscapes can be considered natural capital assets in that they also provide significant economic benefits and contribute to human welfare. The various structural and organizational characteristics of the wildlife habitats or ecosystems in these landscapes determine their capacity to supply a range of ecosystem services that generate these benefits. This natural capital assessment, while not an accounting exercise, aligns with the building blocks of natural capital accounting in that it involves delineating ecosystems in a defined spatial area, assessing their condition, estimating their capacity for delivering ecosystem services, estimating the actual use and value of those services, and finally estimating the value of the ecosystem assets.

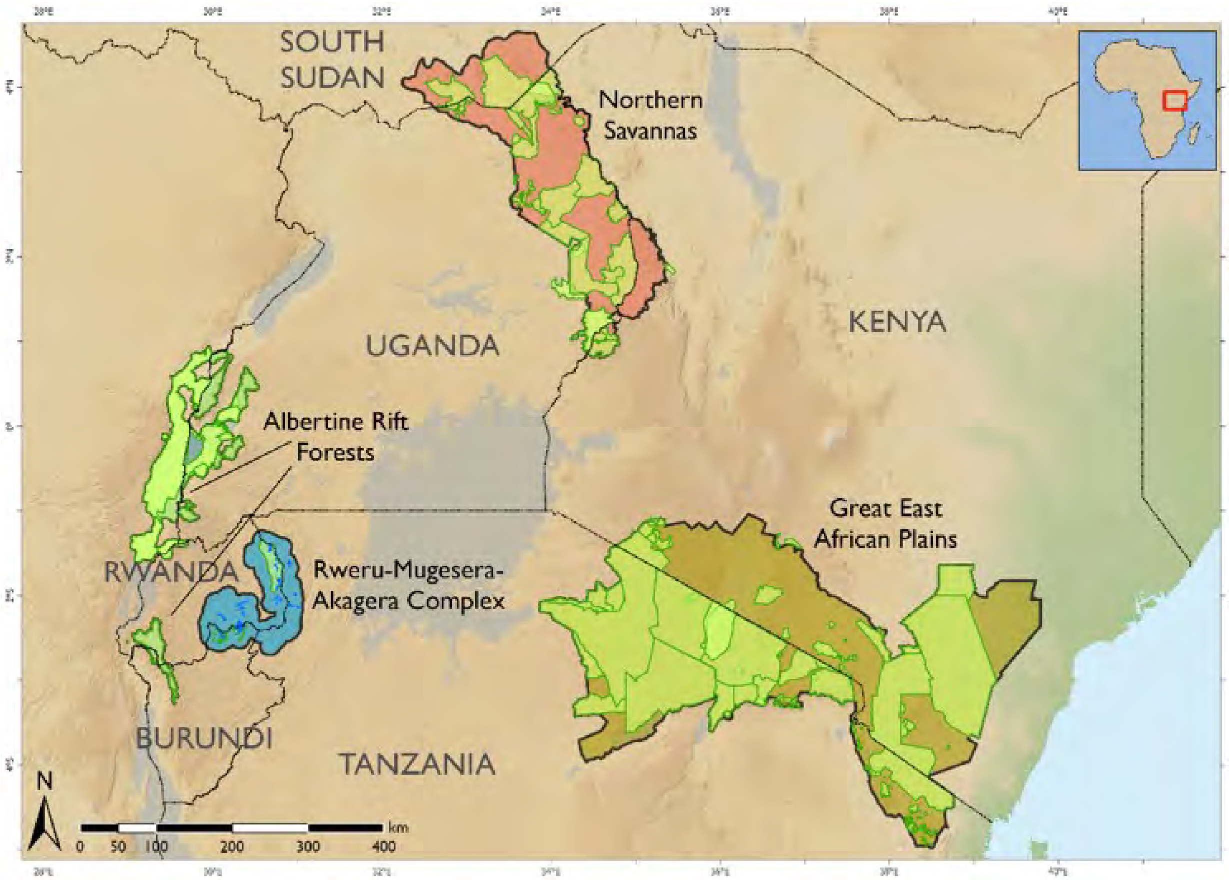


Figure 1. Location of the four study areas, shown in different colors, with protected areas in light green

Ecosystem services are typically classified into provisioning, cultural, and regulating services:

- **Provisioning services** are the harvestable resources supplied by ecosystems, such as wild foods, raw materials, and forage for livestock production. People throughout East Africa harvest a wide variety of wild plant and animal resources for nutrition, health, energy, and raw materials, particularly where there are limited economic opportunities. Resource availability is linked to both ecosystem characteristics and property rights, while demand is influenced by the socio-economic circumstances of households and the prices of alternatives. Ecosystem services provided by wildlife habitats are briefly described below.

- **Cultural services** are the ecosystem attributes (*e.g.*, beauty, rare species) that give rise to the "use values" gained through any type of activity ranging from adventure sports to birdwatching, religious or cultural ceremonies, or just passive observation, or the "non-use values" gained from knowing that they exist and can be enjoyed by future generations. These values can be observed through local use, domestic and international tourism, and the premiums paid for properties that are close to natural amenities, or they can be investigated through stated-preference surveys.

- **Regulating services** are the functions that ecosystems and their biota perform that benefit people in surrounding or downstream areas, or even distant areas. These services include carbon sequestration—the active removal of carbon from the atmosphere by vegetation growth—reducing the potential impacts of climate change, or the passive benefit of retaining the carbon stored in the landscape by avoiding deforestation and hence avoiding causing further climate change damages both locally and in the rest of the world. Other greenhouse gases are also regulated *in situ* if the natural habitats are healthy *e.g.*, leaching of ammonia and other substances is controlled. This category also includes the pollination of crops in nearby fields by insect pollinators that are supported by natural habitats. This is important in low-input, small-scale production systems, which are common in the study areas.

Three types of regulating services are strongly linked to catchment geography, hydrology, and land use. These are the regulation of water flows, the control of sediments, and the removal of excess nutrients that affect water quality. Natural vegetation regulates the flow of water by facilitating the infiltration and temporary storage of rainwater before it enters streamflow. This reduces the seasonal variation in flows and helps to maintain base flows during the dry season, which, in turn, reduces the need to store water during the wet season for use in the dry season. Natural vegetation also helps to slow down floodwaters during storm events, reducing potential damage. In addition, natural vegetation prevents erosion by stabilizing soil and intercepting rainfall, thereby reducing its erosivity, and can also trap eroded sediments that are transported from upstream. Stabilizing soil protects downstream areas from sedimentation, which can mitigate impacts on water storage capacity, hydropower generation, navigability of rivers, and the integrity of downstream lakes or coastal ecosystems. In addition, some of the nutrients in nutrient-enriched runoff can be removed when it passes through natural vegetation and wetlands in the landscape, mitigating downstream eutrophication, toxic algal blooms, deoxygenation, and fish kills that affect human health, water treatment costs, and fisheries. In addition to these active services, retaining ecosystems in their natural state, as opposed to replacing them with alternative uses such as agriculture or human settlements, avoids further losses of sediments and nutrient enrichment. This passive benefit is also considered in this study.

In general, cultural and provisioning services are used purposely, through joint contribution of natural and human-made capital and labor, and the resulting benefit can be valued in terms of gross value added (output value minus input costs, which is contribution to gross domestic product or GDP) or welfare gains (producer and consumer surplus). Regulating services are used inadvertently, and since their loss could lead to damages or require the prevention of such damages through engineering solutions, they are typically valued in terms of avoided costs.

STUDY APPROACH

The assessment team carried out this study in four distinct phases: 1) landscape selection; 2) data collection; 3) ecosystem delineation and classification; and 4) ecosystem services quantification and valuation.

Landscape selection. Four broad study areas were selected on the basis of inputs from stakeholders at an inception workshop, including technical experts from the East African Community (EAC) and wildlife-related non-governmental organizations.

Data collection. Once the landscapes were identified, information on the wildlife and ecosystem characteristics of the study areas, as well as for the region more generally, was collated and reviewed to understand their context and to identify the nature and potential spatial geography of ecosystem services supply and demand pertaining to the four regions. Where multiple datasets were available, these were carefully evaluated to select the most appropriate for the study. Based on data availability, the assessment was done for the situation as of 2018, as this was the latest available year for certain key datasets.

Ecosystem delineation and classification. Ecosystems were then delineated and classified at the regional scale, based on a combination of land cover, vegetation maps, and indicators of vegetation condition. The International Union for the Conservation of Nature's (IUCN's) Global Ecosystem typology was used as far as possible in grouping habitat types. The final classification comprised 72 habitat types, which includes a degraded and undegraded form of each natural habitat type where relevant. These were combined into 23 functional groups. The number of habitat types within each study area ranged from 19 in the Wetlands to 51 in the Great East African Plains.

The next step was to delineate the boundaries of the wildlife landscape study areas using spatial data. This was based on largely contiguous areas of natural habitats within a biome or broadly similar ecosystem types in and around the key protected areas that had been identified. Boundary delineation was also guided by topography to some extent. Although the areas were largely defined by contiguous natural habitat, the inclusion of some areas of human habitation and agriculture was unavoidable. In the case of the Albertine Rift Forests, the remaining wildlife habitat is largely confined to islands of protected areas, with the areas in between having been converted to agriculture. This meant that a contiguous wildlife landscape could not be defined within the East African countries alone. Here, it is critical to note that the remaining Albertine Rift Forest fragments in northern Rwanda and Uganda are connected (and kept viable) only via protected areas in the Democratic Republic of the Congo (DRC). The southern-most protected forest area in this landscape in Rwanda-Burundi is now completely isolated from the rest.

Ecosystem services quantification. Ecosystem services were then quantified in physical terms where appropriate and valued in terms of U.S. dollars per hectare per year. As far as possible, the approach involved estimating the actual use and value of each service based on the estimated capacity of the different ecosystem types to deliver services, and the estimated demand for the services. Our approach is spatial because values depend on context and vary in space as well as time. The landscape capacity to supply services varies with topography, climate, ecosystem type, and condition, and the human demand for services also varies spatially, with population density, infrastructure, and location. The combined flow of values was then used to estimate the asset value of these landscapes in terms of their net present value (NPV) over 30 years. The different services were quantified and valued as follows:

- ***Nature-based tourism***: Benefits to local countries were estimated in terms of the direct GDP contribution of attraction-based tourism, derived from national-level statistics and mapped to the landscape areas within each country using densities of geotagged photographs uploaded to the Flickr platform. Benefits to foreign visitors were estimated in terms of their consumer surplus, which is their willingness to pay over and above what they were required to pay, based on existing studies.

- ***Biodiversity existence***: Estimates were based on a meta-analysis of stated preference studies and relatively simple assumptions about the spatial allocation of this value in relation to global patterns of species richness to arrive at a ballpark estimate of regional and international willingness to pay for the conservation of biodiversity.

- ***Flow regulation***: The annual contribution to base flow was mapped to the landscape using InVEST modeling software. The service was calculated as the difference between this and the baseflow contribution of a hypothetical bare landscape and valued using unit costs of storage infrastructure.

- ***Soil erosion control***: Sediment yields were mapped using InVEST. The service was calculated as the difference between this and the sediment yield that would occur if the landscape were denuded of vegetation and valued using the costs of constructing sediment check-dams.

- ***Water quality amelioration***: Phosphorous outputs were estimated using InVEST and compared to both a bare ground scenario and a landscape transformed to agricultural use. The difference was valued using the modelled construction and maintenance costs of treatment wetlands to handle the equivalent load of phosphorous.

- ***Carbon storage***: The team used global datasets on above- and below-ground biomass and soil carbon. The carbon retention value of these stocks was calculated in terms of the avoided losses of economic output by the countries in the landscape as well as the rest of the world using recent published estimates of the global and disaggregated country-specific damage effects of climate change (social cost of carbon).

- ***Crop pollination***: The value of the service was estimated on the basis of recently published empirical studies carried out in Tanzania and Kenya. This required estimation of the proportion of the area within 1,000 meters (m) of all croplands in the landscape that was natural vegetation.

- ***Livestock forage***: This service was quantified in physical terms as the amount of production (in large stock units) supported, based on spatial data on livestock stocking rates. The annual production was valued using the average gross value added per large stock unit (LSU) derived from national accounts.

- ***Provision of harvested wild resources***: The study focused on small-scale use of wild biomass resources and did not include estimates of legal commercial harvesting of wild resources or illegal commercial-scale poaching of high value, endangered species. The stocks of resources (grouped into broad types) were estimated and mapped based on habitat characteristics, and adjusted for their availability based on land tenure, and then the demand for the resources was mapped based on population density, household characteristics, and livelihoods from census data and the literature, taking livelihood zones into consideration. The use of resources was estimated and mapped using a spatial model in which aggregate use was limited by the availability of resources within a typical range of collection. Use was valued through the most appropriate market prices from the literature.

Validation and stakeholder inputs. The study findings were presented to a diverse group of stakeholders as follows: first the EAC Secretariat and technical experts from the six partner states interrogated the report and provided feedback, followed by stakeholders representing the government, private sector, development partners, research and academia, NGOs, and the local communities based in the specific landscapes. Based on their feedback, and data gaps identified during the assessment, interviews were held with key informants to obtain further information, which was then integrated into the final report.

EXPECTATIONS UNDER A BUSINESS-AS-USUAL SCENARIO

A qualitative assessment was made of the direction and potential magnitude of the effects of a variety of pressures on wildlife habitats and wildlife, and the implications for the supply of ecosystem services under a business-as-usual (BAU) scenario, in other words without any management interventions or changes in policy. To do this, information on past changes in land cover and on projected changes in population, climate, species distributions, and crop suitability were collated and analyzed. The problem was analyzed in terms of the conceptual framework of drivers of biodiversity loss provided in Figure II.

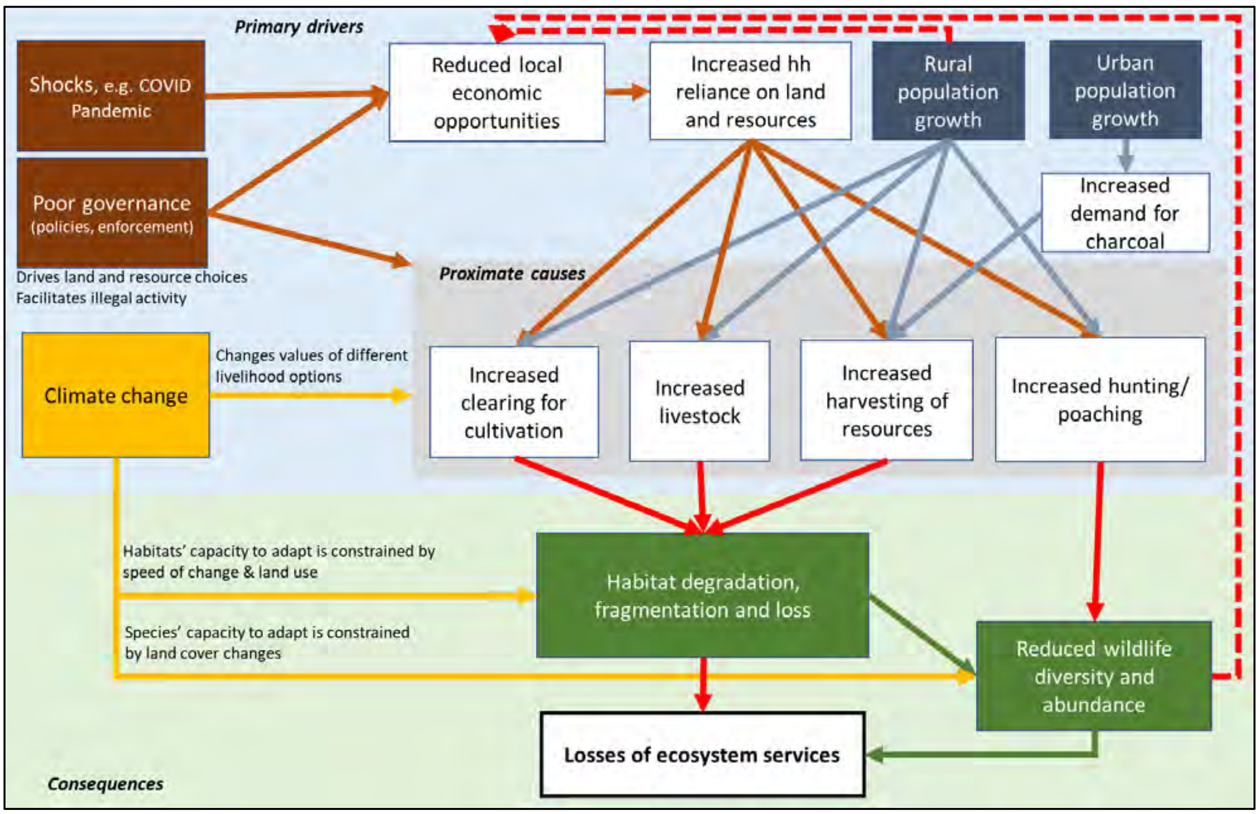


Figure II. Conceptual model of the potential impact pathways leading to loss of ecosystem services over time under a business-as-usual scenarios

The team used existing population projections for rural and urban growth. For land cover changes, findings from the Copernicus 100m land cover, which only go back to 2015, were supplemented with an analysis of the Copernicus 300m dataset, which goes back to 1992. Due to shortcomings in these land cover datasets, we also used Global Forest Change for 2000-2019 as a proxy for land cover change. This dataset is based on the methods originally described by Hansen *et al.* (2013). Historical temperature and rainfall data were obtained from WorldClim, while future projections (2040-2060) were obtained from the Coupled Model Inter-Comparison Project, under the representative concentration pathway 8.5 scenario. The potential impact of a change in climate on the suitability of crop production for key crops across the different landscapes was evaluated using the FAO's EcoCrop analytical tool and the accompanying database. The impacts of climate change on species diversity in the study area was estimated by compiling the outputs of species distribution models for vertebrates in the study areas. These are based on the same climate data as listed above. In addition, the outputs for some key charismatic species were mapped. These impacts were then considered jointly, also taking into account the likely impacts of the COVID-19 pandemic. For predicting changes in ecosystem services under a BAU scenario, projections of future expansion of cultivation and urban areas by 2050 were derived from trends in land cover, deforestation data, and the literature. The impacts these changes would have on various ecosystem services were then estimated.

REGIONAL CONTEXT OF THE STUDY AREAS

East Africa—Burundi, Kenya, Rwanda, South Sudan, Tanzania, and Uganda—is a region of exceptional climatic, topographic, and ecological diversity. Vegetation ranges from forest in the mountainous areas of the Albertine Rift and the humid Indian Ocean coast, to arid shrubland across large areas of northern Kenya, with a range of woodland and savanna types in between. The region is renowned for its wildlife such as the exceptional large herbivore populations in the grassland plains of Tanzania and Kenya, as well as the less well-known wildlife populations found in the grasslands and wetlands of South Sudan. The forest habitats of Uganda, Rwanda, and Burundi are internationally renowned for their population of mountain gorillas and other large primates.

The main topographical features are the dual north-south troughs and mountain escarpments of the African Rift Valley. The western arm, or Albertine Rift, extends from north to south through Uganda, Rwanda, Burundi, and Tanzania as well the DRC. It includes Lakes Albert, Edward, and George in the northern half, which are ultimately connected to the White Nile and Lakes Kivu and Tanganyika in the southern half. The eastern arm, or Gregory Rift, extends through western Kenya and central Tanzania, and includes Lakes Turkana, Baringo, Naivasha, Natron, and Manyara in a landlocked drainage basin. Between these is the large catchment area, encompassing parts of five countries, that drains into Lake Victoria, the largest lake in Africa and source of the White Nile River. East of the Gregory Rift, a number of river basins drain eastwards into the Indian Ocean. The region also features several large isolated volcanic mountains, including Mount Kilimanjaro, Mount Kenya, and Mount Elgon, and the Eastern Arc Mountains in Kenya and Tanzania.

The climate of the region is very diverse. Seasonality of rainfall changes from a southern hemisphere summer rainfall season (around December-April) in the south of Tanzania, through a tropical zone of two rainfall periods (March-May and September-November) or more contiguous rainfall around Lake Victoria Basin (March-November), to a northern hemisphere summer rainfall season (around May-October) in northern Uganda and South Sudan. Rainfall is high in the western half of the region and along the coastal belt in the east, and also in the isolated mountain areas in between. The remaining areas are comparatively dry, particularly in Kenya and northeast and central Tanzania, which experience longer dry seasons and more variable rainfall. Temperature is closely linked to altitude, with cooler temperatures at greater altitude. Future climate predictions are that the region will become hotter and slightly wetter on average. Extreme events such as droughts are likely to become more common, but there are no good models that predict their frequency with any accuracy.

The climatic and topographical diversity of the region is reflected in the extraordinary range of natural habitats across East Africa. A belt of coastal rainforest occurs in the warm, wet Indian coastal regions. Moving inland, *Acacia-Commiphora* bushland and wooded grassland is dominant over much of northern, eastern, and southern Kenya and northern Tanzania. Patches of Afromontane forest also occur in mountainous parts of this generally dry region. Further south, the bushland gives way to miombo woodland, which dominates central and southern Tanzania, while semi-desert shrubland and desert occur in the driest parts of northern Kenya. Over the central highlands and Lake Victoria Basin regions of Kenya, *Acacia-Commiphora* bushland gives way to evergreen bushland, forest, and moist *Combretum* wooded grassland, but much of this has been cultivated. Further west, rainforest was the dominant natural vegetation type across much of the wetter regions to the north and west of Lake Victoria, but little remains today.

East Africa is one of the poorest regions in the world (UNECA, 2020). Despite reasonable economics growth in recent years, its countries rank poorly on the Human Development Index and face steep developmental challenges. Population growth is amongst the highest in the world, having increased from 35 million in 1960 to 185 million in 2018: a five-fold increase in less than 60 years. Rwanda and Burundi are the two most densely populated countries in Africa. Much of this population is rural, although often in fairly densely populated rural areas. Population projections predict that by 2050, the urban populations will increase by about four-fold relative to 2018, and rural populations will increase by about 40 percent.

THE GREAT EAST AFRICAN PLAINS

The Great East African Plains wildlife landscape supports the largest wildlife populations on earth (Figure III). The Serengeti-Mara wildlife landscape, with its rolling hills and open grasslands, supports an incredible profusion and variety of wildlife, including big cats and herds of elephants, which can be seen

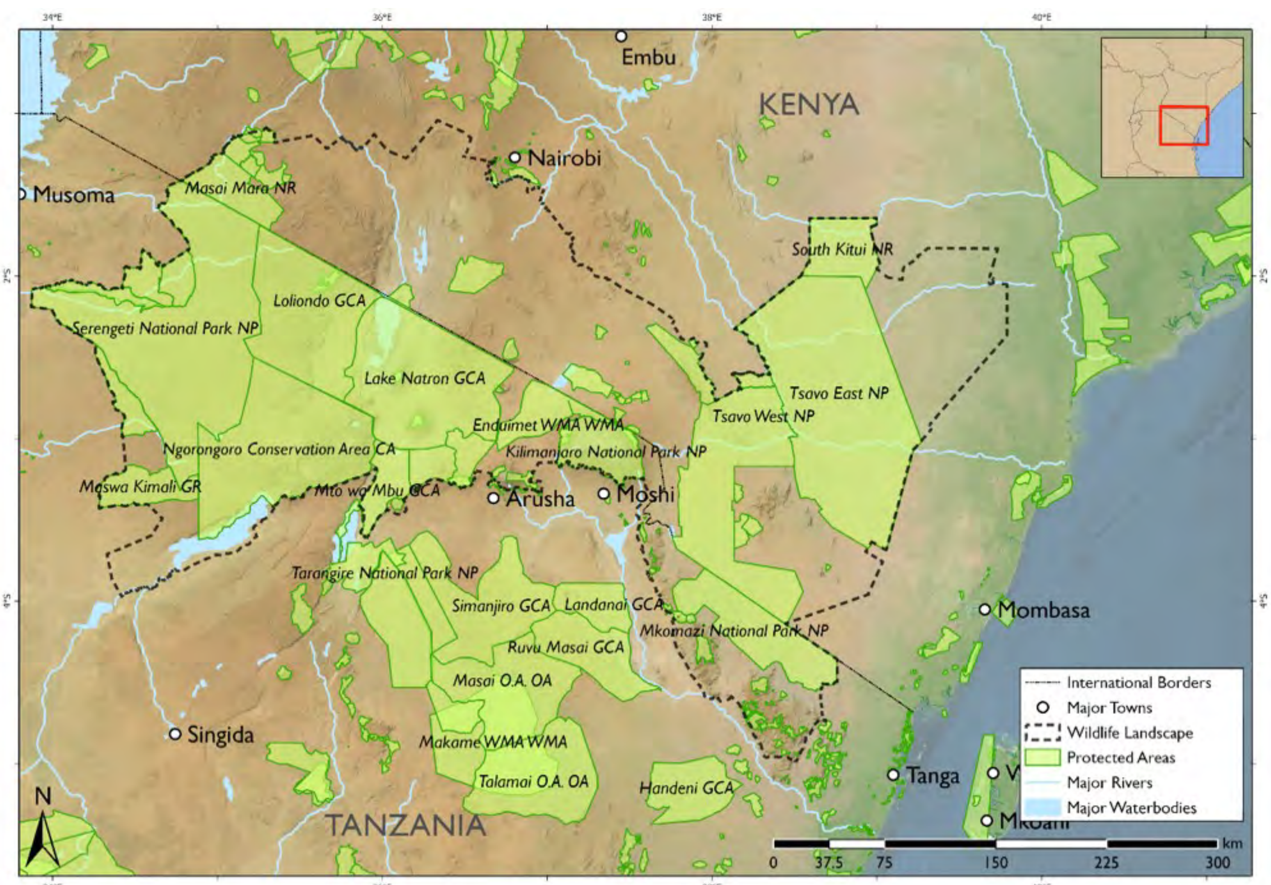


Figure III. Great East African Plains wildlife landscape showing the location of its protected areas up close throughout the year. The Great Wildebeest Migration takes place between August and October each year and is the world's largest migration of wildlife, where more than 1 million wildebeest travel from the Serengeti National Park to the Masai Mara National Reserve. This landscape is also home to the world's largest populations of zebra, eland, lion, cheetah, hyena, and gazelles.

This area encompasses some of the most famous protected areas in Africa, drawing over a million visitors each year prior to the COVID-19 pandemic and bringing significant tourism benefits to Kenya and Tanzania. The total direct contribution to GDP of nature-based tourism to the two countries was estimated to be more than US$1.2 billion in 2018, the highest of the four study areas. Most of this value is associated with the Serengeti National Park and Ngorongoro Conservation Area in Tanzania and the Masai Mara National Reserve in Kenya. **These protected areas account for 21 percent of Tanzania's and 11 percent of Kenya's total tourism value in 2018, respectively**. Nature-based tourism in this landscape also generates an estimated US$1.5 billion in net benefits (consumer surplus) to international visitors.

Keeping the wildlife habitats of this landscape in their current natural condition generates costs savings for the region that could be worth about US$3.2 billion per year, through regulation of hydrological processes and atmospheric carbon. Based on a high-level modeling exercise, these systems contribute an estimated 9 million m^3 in terms of rainwater infiltration and temporary storage, worth US$1 billion per year. They are estimated to retain about 1.8 billion tons (= metric tons) of sediment per year, which would otherwise end up in lakes, reservoirs, estuaries, and coastal environments; this service has a replacement cost value of US$2.2 billion per year. In addition, the wildlife habitats of the landscapes within the catchment areas of Lake Victoria are estimated to reduce phosphorous loadings by some 853-4,855 tons per year (depending on what alternative land use it is compared to), which has a replacement cost of up to US$871,000 per year. These estimates should be refined in the future with more detailed modeling at finer scales, and with the provision of reliable monitoring data on environmental processes in Kenya and Tanzania. Based on satellite data, the vegetation and soils of this wildlife landscape also store an estimated 4.6 billion tons of carbon, the retention of which, according to the most recent estimates, would avoid local climate change damages of some US$788 million per year. In addition, retention of these carbon stocks avoids damages of almost US$400 billion per year at a global scale, which eclipses the GDP output of the East African region of just under US$200 billion.

The wildlife habitats also contribute to agricultural production within the landscape and around their margins. Wild pollinators in the wildlife landscape were estimated to increase crop production by some US$592 million per year. In addition, the Great East African Plains are home to pastoral communities who are dependent on extensive livestock production. The natural rangelands were estimated to support livestock production worth some US$557.5 million per year in terms of contribution to GDP. This is more than 7 percent of total livestock production in Kenya and Tanzania.

Wildlife habitats outside of strictly protected areas (and to some extent within these areas, although not always legally) provide a wide array of wild resources that play an important role in supporting the livelihoods of people. People living in or close to the Great East African Plains wildlife landscape were estimated to harvest some 5.6 million m^3 of firewood, 55,000 tons of wild fruits, vegetables, and medicinal plants, and 406,000 liters of honey, with an estimated total value of US$195.7 million per year. These consumptive use values were twice as high in Tanzania as in Kenya because of the larger population within that country's landscape.

Including a very conservative estimate of the existence value of biodiversity, the wildlife landscape is estimated to be worth at least $508/ha/year on average to East Africa, and more than $31,600/ha globally. In Kenya, the total estimated value (US$2.511 billion per year) of the wildlife landscape represents 3 percent of the country's GDP. In Tanzania, the US$4.071 billion per year is 7 percent of GDP.

Table I. Summary of the benefits derived from ecosystem services of the Great East African Plains wildlife landscape. All values in US$ millions per year

	KENYA	TANZANIA	SUB-TOTAL REGION	REST OF WORLD	TOTAL
Nature-based tourism	507.8	707.2	1,215.0	1,544	2,759
Biodiversity existence	0.8	0.6	1.5	5,372	5,373
Flow regulation	548.8	454.1	1,002.9	-	1,003
Erosion control	594.1	1,632.6	2,226.7	-	2,227
Water quality amelioration	0.3	0.3	0.7	-	1
Carbon storage	290.1	497.8	787.9	397,136	397,924
Crop pollination	253.9	338.1	592.0	-	592
Livestock production	247.8	309.6	557.4	-	557
Harvested resources	66.9	128.7	195.6	-	196
Total value $ millions per year	**2,510.6**	**4,069.1**	**6,579.7**	**404,052**	**410,631**
Total value $ per ha per year	**365.3**	**668.5**	**507.6**	**31,169**	**31,676**

The above values could change dramatically over the next decades under a BAU scenario. Population in and around the Great East African Plains landscape is projected to increase by 1.5 percent per year. Already 1 percent of the wildlife landscape is being lost annually to the expansion of cultivated area. This has important implications for the next 30 years. Furthermore, the projected annual increase in urban populations of 5.7 percent per year coupled with the increasing GDP per capita in Tanzania and Kenya will lead to rapidly rising demand for resources. In addition, the ranges of many species will shift as a result of climate change, with expected resulting contraction of wildlife species ranges and populations. Notwithstanding the great deal of uncertainty involved in future projections, the pressures on wildlife and wildlife habitats are expected to change as follows:

- Habitat transformation could continue to fragment wildlife landscapes, particularly as a result of policies encouraging land subdivision and fencing, which will isolate and threaten the long-term viability of wildlife populations across the region. Based on current trends, it was predicted that cultivation could triple in area, rising from 9.4 percent of the landscape in 2018 to 28.5 percent in 2050 under a BAU scenario.

- The rate of habitat degradation could increase exponentially mainly as a result of population growth, increased livestock numbers, and exponentially increasing urban charcoal demand. Livestock numbers could increase by up to 65 percent in the Kenyan portion of the landscape and 93 percent in the Tanzanian portion by 2050, significantly increasing pressure on rangelands. With many rangelands in the landscape already degraded, these increases would likely lead to further loss of rangeland productivity, compromising the livelihoods of pastoral households and the ability of the landscape to respond to pressures like drought and climate change. Demand for woody resources could increase by 33 percent in the Kenyan portion and 75 percent in the Tanzanian portion over the same period. This could result in significant loss of woody cover and increased fuelwood shortages, compromising important wildlife habitat while reducing the availability of the main energy source for households in the region.

- Tolerance for wildlife and conservation could decrease as human populations, livestock, and cultivation increase.

- Poaching could have a significant impact on wildlife populations as a result of reduced opportunities for income from crops, livestock, and tourism, and particularly in the short- to medium-term by the impacts of the COVID-19 pandemic on tourism and the economy in general. Based on population growth alone, demand for bushmeat could increase by 34 percent in the Kenyan portion of the landscape and 75 percent in the Tanzanian portion by 2050. This demand will likely be increasingly unsustainable, driving further declines in wildlife populations in conjunction with habitat loss.

The potential overall effects of the above pressures on wildlife and wildlife habitats on ecosystem services under a BAU scenario can be summarized as follows (also see Table II):

- Nature-based tourism value has declined significantly due to COVID-19, and recovery will be threatened by wildlife losses, declining habitat quality and wilderness value, and climate change. It was predicted that annual nature-based tourism value would decline by US$76 million in the Kenyan portion of the landscape, and US$85 million in the Tanzanian portion by 2050 under a BAU scenario.

- Water availability in the dry season is expected to decrease, primarily due to the expansion of cultivation and resulting increases in water use by crops. Baseflow could decline by 21.2 percent by 2050, with an annual replacement cost of US$352 million. Baseflow could decline even further if irrigation expands, as the model primarily estimated the increase in water use from land use change alone.

- Freshwater systems are expected to become more polluted. In the portion of the landscape draining into Lake Victoria, phosphorus export could increase by a factor of 2.68, with an annual treatment cost of US$558,000.

- Erosion and sedimentation are expected to increase, with a 9.2 percent reduction in the capacity of the landscape to retain sediment by 2050 under a BAU scenario. This represents an annual cost of US$204 million in maintenance and lost reservoir storage capacity.

- Cultivation and urban expansion will mean the landscape will contribute to further local and global climate change through net carbon emissions, with the release of 5.1 percent (235.4 metric tons of carbon or MtC) of carbon stored in the landscape predicted by 2050. This represents an annual cost of US$40 million to the region in climate-change-related damages.

- The landscape capacity to support agricultural livelihoods will be compromised, affecting the ecological integrity of protected areas.

Table II: Estimated changes in the value of ecosystem services and water treatment costs by 2050 caused by land use changes under a BAU scenario for the Great East African Plains. For services with a global value, both total value to the world and value to the East African region only are shown (latter value in parentheses).

ECOSYSTEM SERVICE	CURRENT VALUE (US$)	2050 VALUE (BAU) (US$)	% CHANGE
Nature-based tourism	2,758.8m (1,215.0m)	2,391.7m (1,053.3m)	-13.3
Biodiversity existence	5,373.5m (1.5m)	4,222.2m (1.2m)	-21.4
Flow regulation	1,002.8m	650.6m	-35.1
Erosion control	2,226.7m	2,023.0m	-9.2
Carbon storage	787.9m (397.9b)	747.6m (376.8b)	-5.1
Water treatment costs	481.5k	640.8k	+33.1

THE NORTHERN SAVANNAS

The Northern Savannas wildlife landscape is a remote region that has a diverse assemblage of mammal and bird species (Figure IV). It is a rugged area where grasses are dotted with iconic tree species such as red thorn acacias and desert dates. Sausage trees and fan palms live along important perennial waterways. More than 500 bird species and more than 86 mammal species, including leopard, cheetah, wild dog, and elephant live in the northern part of the study region.

The contribution of the wildlife landscape to tourism value in 2018 was estimated to be US$8.9 million per year: US$6.6 million in Uganda and US$2.3 million in Kenya. This area of Uganda, in particular Kidepo Valley National Park, has experienced a significant rise in the number of tourists over the last five years. While the tourism industry in South Sudan is currently non-existent, the wildlife areas in this country have the potential to generate significant nature-based tourism value in the future. Nature-based tourism also generates an estimated $11 million in net benefits (consumer surplus) to international visitors.

The hydrologically linked ecosystem services have significant value in this wildlife landscape. The natural vegetation helps rainwater to infiltrate into the ground and contribute to river flows during the dry season (termed “base flows”). The replacement cost of this service could be as much as US$515.4 million per year. An estimated 1.3 billion tons of sediment are retained by the natural ecosystems per year, with a replacement cost value of some US$1.6 billion. Natural vegetation in the landscape reduces phosphorus loadings of some 795-1,258 tons (depending on the alternative land use) from reaching Lake Kyoga, which has a replacement value of about US$503,000 to US$573,600 per year. These estimates should be refined in the future with more detailed modeling at finer scales, and with the provision of reliable monitoring data on environmental processes in Uganda, Kenya, and South Sudan. Based on satellite data, the vegetation and soils of this wildlife landscape also store an estimated 2.2 billion tons of carbon, which is estimated to avoid local climate change damages of some US$260 million per year. In addition, these carbon stocks avoid damages of some US$150 billion per year at a global scale.

Figure IV. The Northern Savannas wildlife landscape and protected areas

The wildlife habitats of the Northern Savannas landscape also contribute to agricultural production within the landscape and around its margins. Wild pollinators there were estimated to increase crop production by some US$144.3 million per year. In addition, the natural rangelands were estimated to support livestock production worth some US$372.3 million per year.

The wildlife landscape of the Northern Savannas provides a wide array of wild resources that play an important role in supporting the livelihoods of people, especially in the northern-most parts of the study region. People living in or close to the Northern Savannas wildlife landscape were estimated to harvest some 4 million m^3 of firewood, 137,000 tons of wild fruits, vegetables, and medicinal plants, and 1.7 million liters of honey, with an estimated total value of US$313.5 million per year.

Including a conservative estimate of the existence value of biodiversity, the wildlife landscape is estimated to be worth at least $650/ha/year on average to East Africa, and more than $31,000 per ha globally. In Uganda, the total value of the wildlife landscape (US$1,790 million per year) equates to 7 percent of the country's GDP; in South Sudan the contribution of the wildlife landscape (US$1,118 million per year) to GDP is closer to 9 percent, and in Kenya is lower, contributing (US$ 558.7) around 1 percent to GDP.

Table III. Order-of-magnitude estimates of the benefits derived from ecosystem services of the Northern Savannas wildlife landscape. All values in US$ millions per year

	KENYA	SOUTH SUDAN	UGANDA	REGION	REST OF WORLD	TOTAL
Nature-based tourism	2.3	?	6.6	8.9	11	20
Biodiversity existence	0.3	0.1	0.1	0.6	2,024	2,024
Flow regulation	46.1	112.6	356.7	515.4	-	515
Erosion control	246.8	631.0	684.1	1 561.9	-	1 562
Water quality amelioration	0.2	0.2	0.2	0.5	-	0.6
Carbon storage	57.0	55.6	147.5	260.1	149,720	149,980
Crop pollination	32.3	15.3	96.7	144.3	-	144
Livestock production	70.9	64.2	237.1	372.2	-	372
Harvested resources	60.3	117.9	135.3	313.5	-	314
Total value $ millions per year	**516.2**	**996.9**	**1,664.3**	**3,177.4**	**151,755**	**154,933**
Total value $ per ha per year	**995.6**	**890.1**	**512.7**	**650.5**	**31,067**	**31,717**

These values could change dramatically over the next decades under a BAU scenario. While the expansion of cultivated areas is relatively slow at 0.1 percent of the wildlife landscape per year, rural populations in and around the Northern Savannas landscape could increase by 1.5 percent per year if they follow national trends. This has important implications for the next 30 years. Furthermore, the projected annual increase in urban populations of 5.9 percent per year coupled with the increasing GDP per capita in the three countries will lead to rapidly rising demand for resources. Additionally, the ranges of many species will shift as a result of climate change, with expected resulting contraction of wildlife species ranges and populations. Notwithstanding the great deal of uncertainty involved in future projections, the pressures on wildlife and wildlife habitats are expected to change as follows (also see Table IV):

- Agricultural expansion could continue to reduce and fragment wildlife habitats. It was predicted that the area of cultivation would increase from 5.1 percent of the landscape in 2018 to 7.4 percent by 2050 under a BAU scenario. This would result in the conversion of an additional 109,000 ha of habitat.

- Habitat degradation is likely to worsen due to increased overharvesting of resources and overgrazing. Demand for woody resources was predicted to increase by 35 percent by 2050 under a BAU scenario. Livestock numbers could double over this period in the Ugandan portion of the landscape and increase by 65 percent in the Kenyan portion. Trends in livestock numbers are less clear for South Sudan.

- Tolerance for wildlife and conservation could decrease as human populations, livestock, and cultivation increase.

- Poaching could increase, significantly affecting wildlife populations. It was predicted that population growth would lead to an increase in bushmeat demand of 30 percent by 2050 under a BAU scenario.

The potential overall effects of the above pressures on wildlife and wildlife habitats on ecosystem services under a BAU scenario can be summarized as follows:

- Nature-based tourism has declined due to COVID-19, and future recovery will be uncertain due to the loss of wildlife, declining habitat quality, insecurity, and climate change. Under a BAU scenario, annual tourism value could decline by US$1.45 million in the Ugandan portion of the landscape and US$280,000 in the Kenyan portion by 2050. The South Sudan portion currently has no tourism, and it is unclear how likely this is to change in the future.

- Erosion and sedimentation are expected to increase, with a 0.4 percent reduction in the capacity of the landscape to retain sediment by 2050 under a BAU scenario. This represents an annual cost of US$6 million in maintenance and lost reservoir storage.

- Water availability in the dry season is expected to decrease as a result of reduced vegetation cover and the expansion of cultivation. Baseflow could decline by 2.5 percent by 2050, with an annual replacement cost of US$23 million. Baseflow could decline even further if irrigation expands, as the model primarily estimated the increase in water use from land use change alone.

- Freshwater ecosystems are expected to become more polluted as a result of increasing agriculture. Phosphorus export was predicted to increase by 4.7 percent by 2050 in the portion of the landscape that drains into Lake Kyoga, with an annual treatment cost of US$223,000.

- Landscape degradation is expected to contribute to further local and global climate change, with a predicted release of 0.5 percent (10.7 MtC) of the carbon stored by the landscape by 2050. This represents a loss in value of US$560,000.
- The landscape's capacity to support agricultural livelihoods will be compromised, affecting the ecological integrity of protected areas.

Table IV: Estimated changes in the value of ecosystem services and water treatment costs by 2050 caused by land use changes under a BAU scenario for the Northern Savannas landscape. For services with a global value, both total value to the world and value to the East African region only are shown (latter value in parentheses).

ECOSYSTEM SERVICE	CURRENT VALUE (US$)	2050 VALUE (BAU) (US$)	% CHANGE
Nature-based tourism	20.2m (8.9m)	16.2m (7.2m)	-19.5
Biodiversity existence	2,024.8m (0.6m)	1,973.6m (0.5m)	-2.5
Flow regulation	515.4m	492.5m	-4.4
Erosion control	1,561.9/m	1,556.0m	-0.4
Carbon storage	150.0b (260.1m)	149.5b (258.8m)	-0.5
Water treatment costs	481.5k	557.8k	+1.3

THE ALBERTINE RIFT FORESTS

The Albertine Rift Forests are among the most biodiverse forests in the world and one of the most important regions for conservation in Africa (Figure V). What remains of the forest is largely confined to protected areas. Species richness is exceptionally high, with over 50 percent of Africa's bird species and 40 percent of the continent's mammals found in the Albertine Rift region. This diversity is all the more remarkable considering the region accounts for just 1 percent of Africa's surface area. Importantly for conservation purposes, the Albertine Rift also holds more endemic and globally threatened vertebrates than any other region in mainland Africa.

Primarily because of their primate populations and their management as high value, low-impact tourism resources, the protected areas bring significant tourism benefits to Rwanda and Uganda. Due to lower visitor numbers, the contribution of protected areas to tourism revenue is much more modest for Burundi. The total direct contribution to GDP of nature-based tourism in the Albertine Rift Forests landscape was estimated to be US$50.3 million in 2018, the bulk of which is associated with Queen Elizabeth and Mgahinga Gorilla National Parks in Uganda and Volcanoes National Park in Rwanda. The national parks in this region generate US$27.6 million or 4 percent of tourism value in Uganda, US$11.8 million or 3 percent in Rwanda, and US$0.4 million or 1 percent in Burundi. These values only represent tourism's direct contribution to GDP and do not include the knock-on effects for other sectors. The total economic impact of tourism expenditure is therefore much higher than reported here. Nature-based tourism also generates an estimated $83 million in net benefits to overseas visitors.

Keeping the forest habitats of this landscape in their current natural condition generates costs savings for the region that could be worth about US$750 million per year, through regulation of hydrological

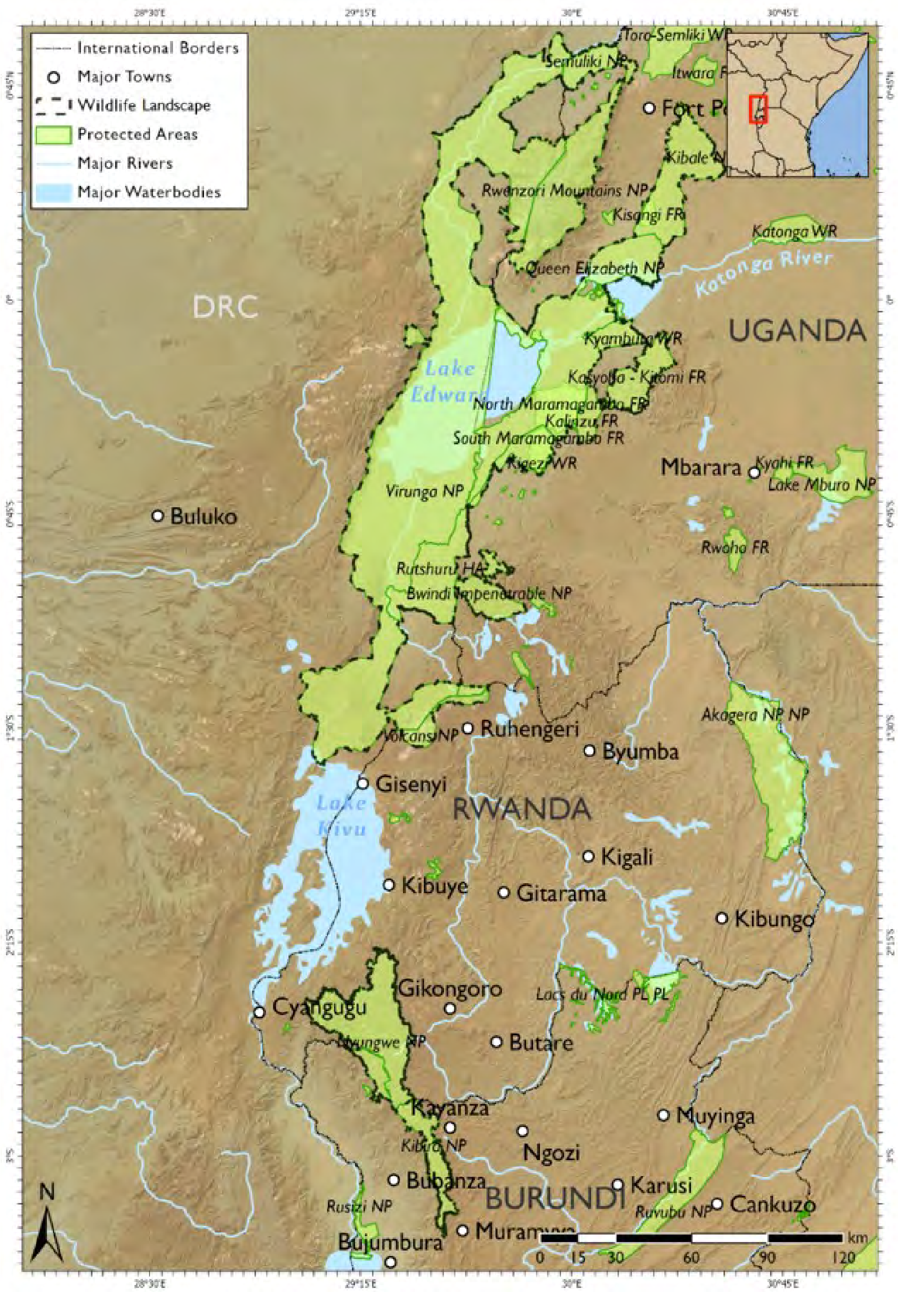


Figure V. The Albertine Rift Forests wildlife landscape and protected areas

processes and atmospheric carbon. Based on the high-level modeling exercise carried out in this study, these systems are estimated to retain about 498 million tons of sediment per year that would otherwise end up in rivers, wetlands, and lakes. This service has a replacement cost value of US$612 million per year. In addition, the forest habitats of this landscape are estimated to reduce phosphorous loadings by some 165-2,244 tons per year (depending on the alternative land use), which has a replacement cost of between US$331,360 and US$682,469. It appears that these forests do not have a smoothing effect on baseflows, however. This is not an uncommon finding for forested areas. These estimates should be refined in future with more detailed modeling at finer scales, with the provision of reliable monitoring data on environmental processes in Uganda, Rwanda, and Burundi, and should ideally be extended to incorporate the DRC. Based on satellite data, the vegetation and soils of this wildlife landscape also store an estimated 643 million tons of carbon, the retention of which, according to the most recent estimates, would avoid local climate change damages of some US$63 million per year. In addition, retention of these carbon stocks avoids damages of some US$42 billion per year at a global scale.

The forest habitats also contribute to agricultural production around the margins of this landscape. Wild pollinators were estimated to increase crop production by some US$36.2 million per year. The forests of the Albertine Rift provide a wide array of wild resources that play an important role in supporting the livelihoods of people. Wild plant and animal resources are harvested for food, medicine, energy, and raw materials, particularly where there are limited economic opportunities, and have an estimated total value of US$352.4 million per year.

Including conservative estimates of the existence value of biodiversity (including a study of international willingness to pay – Hatfield & Malleret-King 2005), the wildlife landscape is estimated to be worth at least $1,430/ha/year on average to East Africa and more than $56,000/ha/year globally (Table V). The forest wildlife landscapes in Rwanda and Uganda, with total values of US$305 and US$754 million per year, respectively, contribute 3 percent of GDP in these countries. In Burundi, this value (US$128 million per year) represents 4 percent of GDP.

Table V. Summary of the benefits derived from ecosystem services of the Albertine Rift Forest wildlife habitats. All values in US$ millions per year.

	BURUNDI	RWANDA	UGANDA	REGION	REST OF WORLD	TOTAL
Nature-based tourism	0.5	13.3	36.5	50.3	83	134
Biodiversity existence	-	0.0	0.1	0.1	322	322
Erosion control	65.9	128.8	417.1	611.8	-	612
Water quality amelioration	0.2	0.2	0.2	0.5	-	0.5
Carbon storage	0.1	4.6	57.8	62.6	42,216	42,279
Crop pollination	4.7	3.4	28.2	36.3	-	36
Harvested resources	49.3	139.9	162.9	352.1	-	352
Total value $ millions per year	**120.7**	**290.2**	**702.8**	**1,113.7**	**42,622**	**43,736**
Total value $ per ha per year	**2,554.5**	**2,462.6**	**1,148.1**	**1,432.9**	**54,838**	**56,271**

The above values could change dramatically over the next decades under a BAU scenario. While the expansion of cultivated areas is relatively slow at 0.1 percent of the wildlife landscape per year, rural populations in and around the Albertine Rift Forests landscape could increase by 2.0 percent per year if they follow national trends. This has important implications for the next 30 years. Furthermore, the projected annual increase in urban populations of 6.6 percent per year coupled with the increasing GDP per capita in two of the three countries will lead to rapidly rising demand for resources. In addition, the ranges of many species will shift as a result of climate change, with expected resulting contraction of wildlife species ranges and populations. Notwithstanding the great deal of uncertainty involved in future projections, the pressures on wildlife and wildlife habitats are expected to change as follows:

- Deforestation could accelerate with increasing demand for timber, charcoal production, and land for cultivation, shrinking available intact wildlife habitat. Extrapolating from past trends, it was predicted that 15.5 percent of existing forest cover could be lost by 2050 under a BAU scenario. This could further threaten unique wildlife species in the area, including endangered, charismatic species like gorillas, chimpanzees, and forest elephants, whose ranges have already been substantially contracted by historical habitat loss. Future deforestation could also have a negative impact on GDP through compromising ecosystem services and tourism, as expanded on below.

- Future encroachment and clearing of forested areas could increase the risk of novel zoonotic disease emergence due to the increased contact between wildlife, people, and livestock.

- Landscape connectivity could be further compromised, threatening the viability of wildlife populations.

- Key wildlife species could disappear due to shrinking suitable climatic ranges.

- Tolerance for wildlife and conservation could decrease as human populations, livestock, and cultivation increase.

- Poaching is likely to increase, having a significant impact on wildlife populations. Due to dense, growing populations, demand for bushmeat is predicted to increase by 74 percent by 2050.

The potential overall effects of the above pressures on wildlife and wildlife habitats on ecosystem services under a BAU scenario can be summarized as follows (also see Table VI):

- Wildlife tourism revenue has declined significantly due to COVID-19 but may recover well if key attractions and habitats can be maintained. Under a BAU scenario, annual tourism value could increase by US$5.3 million and US4.2 million by 2050 in the Rwandan and Ugandan portions of the landscape respectively, if gorilla conservation efforts remain effective. In contrast, annual tourism value was predicted to decline by US$400,000 in Burundi due to poorly developed tourism products, insecurity, and forest encroachment.

- Total annual runoff might increase but so will flood risk, while dry season flows might decline. A 3.1 percent decline in baseflow was predicted by 2050 due to the expansion of cultivation. This would have an annual replacement cost of US$13 million. Baseflow could decline further if irrigation expands, as the model primarily estimated the increase in water use from land use change alone.

- Soil erosion and sedimentation are expected to increase. Deforestation and expansion of cultivation could reduce the capacity of the landscape to retain sediment by 1.3 percent by 2050. This represents an annual cost of US$8 million in maintenance and lost reservoir storage.

- Nutrient pollution of lakes and watercourses is expected to worsen. The amount of phosphorus exported to waterbodies is predicted to increase by a factor of 3.9 by 2050, with an annual treatment cost of US$338,000.
- Deforestation and encroaching urbanization is likely to make a significant contribution to local and global climate change. It is predicted that 7.6 percent (48.7 MtC) of carbon stored by the landscape will be released by 2050, representing a loss in value of US$4.7 million.

Table VI. Estimated changes in the value of ecosystem services and water treatment costs by 2050 caused by land use changes under a BAU scenario for the Albertine Rift Forest landscape. For services with a global value, both total value to the world and value to the East African region only are shown (latter value in parentheses).

ECOSYSTEM SERVICE	CURRENT VALUE (US$)	2050 VALUE (BAU) (US$)	% CHANGE
Nature-based tourism	83.4m (50.3m)	99.1m (59.7m)	+18.7
Biodiversity existence	322.2m (87.6k)	296.7m (80.7k)	-7.9
Flow regulation		-12.7m	-3.1
Erosion control	611.8m	603.7m	-1.3
Carbon storage	42.2b (62.6m)	39.0b (57.9m)	-7.6
Water treatment costs	261.3k	364.3k	+39.4

THE RWERU-MUGESERA-AKAGERA WETLANDS

The Rweru-Mugesera-Akagera wetland complex in Burundi, Rwanda, and Tanzania is one of the largest wetland areas in the basins surrounding Lake Victoria (Figure VI). Large areas of papyrus swamps and several open water lakes cover this area, providing home to a wide array of birds and wildlife. Parts of the wetland system are protected in Burundi and Rwanda, with Akagera National Park being one of the largest protected wetlands in East Africa.

The contribution of the Rweru-Mugesera-Akagera Wetlands to tourism value was estimated to be US$5.3 million in 2018: US$4.5 million in Rwanda, US$0.7 million in Tanzania, and US$0.08 million in Burundi. In Rwanda, this represents just over 1 percent of the total tourism value in the country. The tourism value of Akagera National Park was estimated to be US$2.6 million per year (US$26/ha/y), accounting for 50 percent of the total tourism value across the Rweru-Mugesera-Akagera Wetlands. Nature-based tourism also generates an estimated $7 million in net benefits to international visitors.

The large wetland system, with its extensive areas of papyrus-dominated swamps, can remove large quantities of the nutrients that enter it as a result of human activities in its catchment areas. These nutrients would otherwise reach Lake Victoria, adding to the problems of eutrophication there. This service was estimated to be worth on the order of US$0.7 million per year. In addition, the high biomass of the wetland system stores an estimated 92 million tons of carbon, which is estimated to avoid local climate change damage costs on the order of US$8 million per year and global damages of US$7 billion per year.

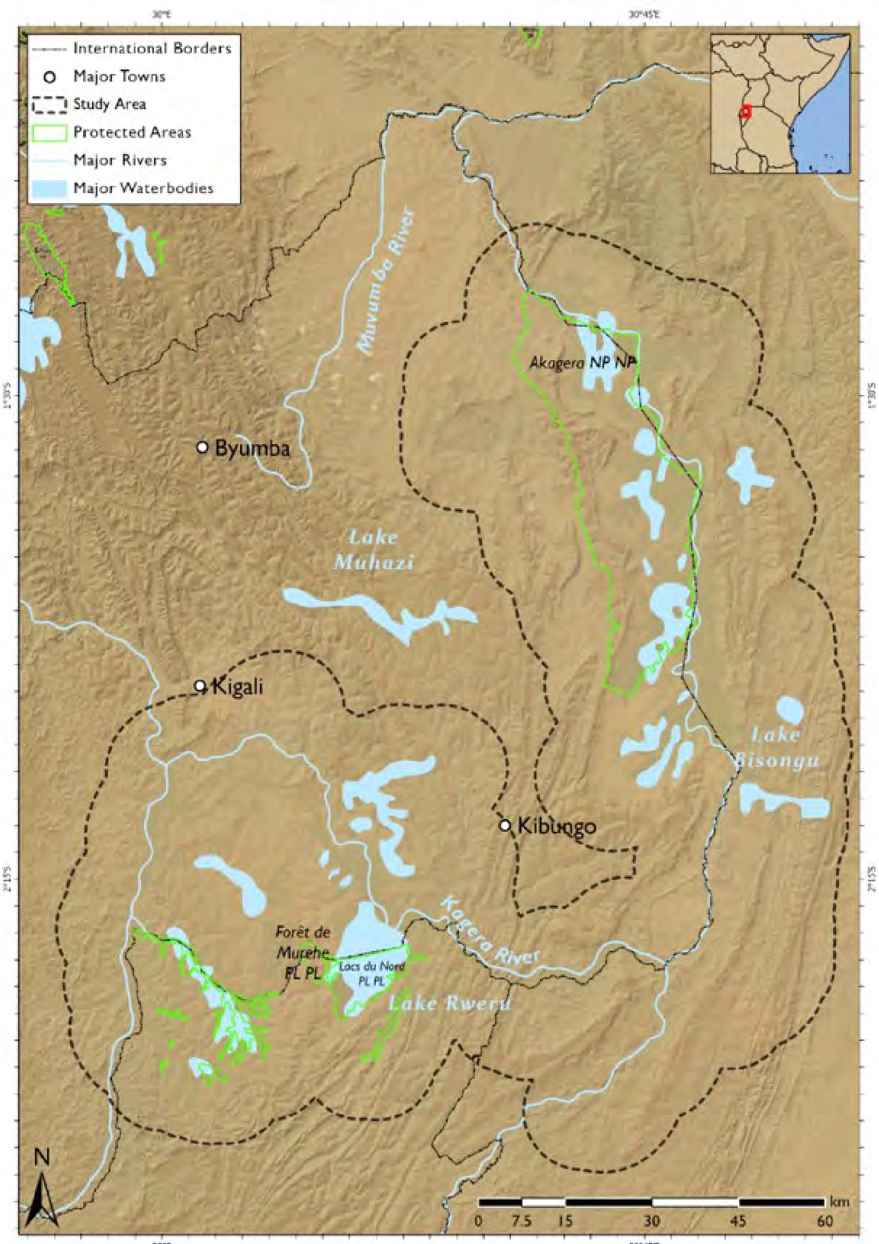


Figure VI. The Rweru-Mugesera-Akagera Wetland complex and associated protected areas

Wetland resources, harvested for materials and food, play an important role in supporting the livelihoods of people across this region. The value of all wild resources harvested totaled US$50.2 million for the wetland complex study region. Including a conservative estimate of the existence value of biodiversity, the wildlife landscape is estimated to be worth at least $300/ha/year on average to East

Africa, and almost $35,000/ha/year globally. The total values (millions per year) for the wetland complex in each country represent less than 1 percent contribution to GDP.

Table VII. Summary of the benefits derived from ecosystem services of the Rweru-Mugesera-Akagera wetland complex. All values in US$ millions per year.

	BURUNDI	RWANDA	TANZANIA	REGION	REST OF WORLD	TOTAL
Nature-based tourism	0.08	4.50	0.70	5.28	7	12
Biodiversity existence	-	0.003	0.02	0.02	89	89
Water quality amelioration	0.23	0.23	0.23	0.70	-	1
Carbon storage	0.01	1.81	6.34	8.15	7,328	7,337
Harvested resources	12.40	26.10	11.70	50.20	-	50
Total value $ millions per year	**12.7**	**32.6**	**19.0**	**64.4**	**7,424**	**7,488**
Total value $ per ha per year	**524.7**	**257.7**	**298.2**	**299.9**	**34,598**	**34,898**

The wetland's potential importance has been eroded by encroachment around it. It could sustain a much more significant wildlife landscape if some of the surrounding areas were restored. Right now, its most important local benefit seems to be the provision of natural resources that are typically harvested by poor households, *i.e.,* it acts as a safety net.

The above values could change dramatically over the next decades under a BAU scenario. Rural populations in and around the Rweru-Mugesera-Akagera Wetlands could increase by 1.9 percent per year if they follow national trends. Indeed, the area under cultivation within 10km of the wetlands has been increasing at a rate of 15,580 hectares per year. This has important implications for the next 30 years. Furthermore, the projected annual increase in urban populations of 6.0 percent per annum coupled with the increasing GDP per capita in two of the three countries will lead to rapidly rising demand for resources. Added to this, the ranges of many species will shift as a result of climate change, with expected resulting contraction of wildlife species ranges and populations. Notwithstanding the great deal of uncertainty involved in future projections, the pressures on wildlife and wildlife habitats are expected to change as follows:

- Further conversion of wetland habitat could occur as a result of population growth and increasing land scarcity.
- Nutrient and industrial pollution could worsen as more land is converted to agriculture and as poorly serviced urban areas in the catchment expand.
- Water hyacinth could continue to spread, negatively affecting ecosystems and aquatic life.
- The wetlands could experience increased siltation.

- Wetland integrity could be compromised by water abstraction and other hydrological modifications.
- Large wildlife could disappear from all but Akagera National Park.

The potential overall effects of the above pressures on wildlife and wildlife habitats on ecosystem services under a BAU scenario can be summarized as follows:

- Wildlife tourism revenue has been negatively affected by COVID-19. However, over time, a US$1.8 million increase in tourism value is predicted by 2050 for the Rwandan portion of the wetlands, primarily due to effective management of Akagera National Park. Conversely, tourism value is predicted to decline by US$57,000 and US$6,000 in the Tanzania and Burundi portions of the wetlands, respectively, due to poorly developed tourism products, rapid population growth, and habitat loss.
- The ability of the wetlands to remove pollutants is expected to decrease.
- Availability of water for agriculture and domestic use is expected to decrease.
- Water quality in the system is also expected to decrease.
- Wetland degradation is expected to increase the severity of local and global climate change.
- The stocks of fish, papyrus, and other resources are expected to decline. Due to rapid population growth, demand for fish is predicted to increase by 113 percent, while demand for papyrus will increase by 75 percent by 2050.

Table VIII. Estimated changes in the value of ecosystem services and water treatment costs by 2050 caused by land use changes under a BAU scenario for the Rweru-Mugesera-Akagera Wetlands. For services with a global value, both total value to the world and value to the East African region only are shown (latter value in parentheses).

ECOSYSTEM SERVICE	CURRENT VALUE (US$)	2050 VALUE (BAU) (US$)	% CHANGE
Nature-based tourism	12.0m (5.3m)	15.9m (7.0m)	+32.5
Carbon storage	7.3b (8.2m)	7.2b (8.1m)	-1.5

CONCLUSION, POLICY IMPLICATIONS, AND NEXT STEPS

This preliminary assessment provides a first regional-scale assessment of a relatively comprehensive suite of ecosystem services in four priority transboundary wildlife landscapes of the EAC states. This is therefore an important initial contribution to the understanding of the economic benefits provided by the region's natural capital. Using conservative assumptions, the study estimates that within these relatively undeveloped landscapes that still offer significant and viable habitat for wildlife populations, ecosystems generate services of about $300/ha/year for the wetland, $500/ha/year for the savanna, $700/ha/year for the plains, and $1,500/ha/year for the forest landscapes on average. Benefits to the different countries also vary, with the national portions of the different landscapes bringing benefits ranging from $260/ha/year for wetlands in Rwanda to $2,700/ha/year for forests in Burundi. The benefits at global scale are orders of magnitude greater than this, with the values ranging from $32,000 to $56,000/ha/year on average for the four landscapes. This difference is largely because of the significant benefit of carbon retention in avoiding increases in future climate change damages around the world. These are "ballpark" estimates, based on best available information and large-scale, thus relatively coarse, modelling and assumptions. Nevertheless, they provide a first indication of the potentially very high value of these areas that are already well noted for their conservation importance. Indeed, the total combined value of the wildlife landscapes within each country represents a significant contribution to GDP. In Burundi, the total combined estimated value of wildlife landscapes equates to 5 percent of GDP, in Kenya 3 percent, Rwanda 4 percent, Uganda 9 percent, Tanzania 7 percent, and South Sudan 9 percent.

The wildlife landscapes selected for this study are of international renown as tourism destinations, and it is largely assumed that their primary value is tourism. However, this assumption puts the landscapes in jeopardy from a policy perspective, since the tourism economy is vulnerable to shocks such the COVID-19 pandemic, regional political instability, or global economic recession. Indeed, tourism values are high, particularly in the Great East African Plains landscape, where tourism is estimated to generate direct benefits on the order of $2.7 billion per year. However, even from a regional perspective, this value is well exceeded by the value of other, less obvious, regulating ecosystem services, particularly erosion control, flow regulation, and carbon sequestration. This is an important finding of the study. Harvested resources also form a large proportion of the local value, particularly for the forest and wetland landscape, but these values are also a potential threat to the landscape if use is unsustainable.

The global benefits of the study areas are significant and have important policy implications. They not only include climate benefits, but also very large values held by global society for the conservation of wild habitats and species, and significant benefits derived by tourists visiting the areas. These positive externalities of the region could be internalized. In fact, this is already occurring to some extent. Global society's willingness to pay for conservation measures to avoid carbon emissions as well as for the conservation of wildlife is partly reflected in donor payments. Tourists' consumer surplus could be further captured through optimal, differentiated pricing systems.

The local benefits are also significant. While the values vary geographically, the average values noted above suggest that the national benefits of conservation action will likely exceed local scale opportunity costs in terms of forfeited small-scale activities. This makes it feasible to introduce stronger policies and actions to ensure the continued protection, integrity, and connectivity of the habitats that remain.

The threats, however, are significant too. Across the region, population is still growing rapidly, and this poses one of the greatest threats to the future value of these landscapes, along with climate change. Population and climate as the primary drivers of the threats to the wildlife landscapes are extremely difficult to change, but countries will potentially need to put policies in place to do so, or at least to draw the pressures to less vulnerable areas. Sometimes threats to the unique and precious features of landscapes are caused by improper management decisions, not because of irresponsibility, but rather due to lack of crucial information such as economic value of habitats. This study identified important areas in the landscapes for the supply and delivery of ecosystem services. It would therefore be possible to provide governance-based incentives, *e.g.*, designing a land-use development plan that balances multiple private and public values in the landscapes.

There is a range of policy options that could help to produce a win-win situation. Given the high stakes, very careful consideration of these options is required. The next steps of the study will therefore involve the investigation of feasible policy interventions that will be effective in retaining the biodiversity and economic value of these wildlife landscapes.

INTRODUCTION

PROJECT BACKGROUND AND RATIONALE

Wildlife and wildlife habitats are critically important natural capital assets in East Africa, and in many respects underpin the very essence of the region. In addition to their significant heritage contribution at local to global scales, the vast wildlife landscapes of the region have provided an important comparative economic advantage for East Africa's development and will be vital in ensuring its resilience in the face of mounting pressures.

Wildlife landscapes, as defined in this study, comprise contiguous and interconnected ecosystems that support self-sustaining and genetically viable populations of larger animal species as well as relatively intact communities of plants and animals. These ecological systems are part of the region's natural capital, defined as the ecosystems and natural resource stocks that supply ecosystem services that in turn contribute to human wellbeing.

In fact, natural systems provide a wide range of ecosystem services. Not only do they support wildlife, which have a range of cultural values including nature-based tourism, but they also provide carbon sequestration, flow regulation, water quality amelioration, sediment retention, and pollination, collectively known as regulating services, which provide benefits to people living beyond the landscapes. Natural systems also supply goods such as harvested raw materials, bushmeat, and grazing (provisioning services); and provide the opportunity for recreation, tourism, and spiritual fulfillment (cultural services).

There tends to be a tradeoff between deriving benefits from the consumptive use of an ecosystem's natural resources and deriving benefits from cultural activities and regulating services. For this reason, areas are set aside for formal protection from consumptive uses to preferentially deliver specific services such as the protection of biodiversity for nature-based tourism, for the enjoyment of future generations, or to secure water supply. In turn, an ecosystem's cultural and regulating services may be reduced by consumptive use of resources, particularly where such use results in measurable ecological modification such as changes in wild animal densities, tree densities, or land cover. Economic analysis considers these sorts of trade-offs facing society in the light of their overall impacts on human wellbeing. It helps communities and their decision makers find the optimal balance between conservation and development, in their varying forms.

Over time, the demand for land and resources is likely to increase, and decisions regarding the extent to which land is set aside for wildlife and the provision of cultural and regulating services will become increasingly pertinent. In East Africa, the extent and connectivity of wildlife landscapes has already been markedly reduced by the expansion of human settlements, agriculture, and mining, largely driven by the more than seven-fold growth in population from about 26 million people in 1950 to 196 million in 2020. This exponential growth, coupled with accelerating climate change and global issues such as the recent COVID-19 pandemic, have put increasing pressure on the natural environment. These pressures threaten to further diminish ecological systems' integrity and capacity to deliver ecosystem services, including those that cushion households from economic shocks. Local communities depend on wildlife and wild spaces for revenue and livelihoods, but rapid urban, agricultural, and industrial development has fragmented and degraded wildlife habitats, and created new avenues for illegal wildlife trafficking. Habitat destruction and illegal wildlife trafficking are as much an economic problem as a conservation one. Yet, these same diminishing assets are now increasingly vital to sustain human wellbeing as it becomes apparent that, beyond some point, their degradation comes at a high price to society.

With the world in the so-called Anthropocene Epoch, marked by unprecedented human impacts on the earth's climate and natural systems, it has become imperative to take stock of natural capital and monitoring its extent and contribution to human wellbeing in order to inform the sustainability of future development policies. This is borne out in the Convention on Biodiversity's Aichi Targets,[1] which require countries to account for their natural capital, as well as the UN's recent efforts to formalize methods for accounting for natural capital as part of a System of Environmental Economic Accounting (UN 2014). Natural capital accounting will go a long way to understanding the value of ecosystems and implications of changing land use to better inform policies and decision-making for conservation and development. The member countries of the East African Community (EAC)—Burundi, Kenya, Rwanda, South Sudan, Tanzania, and Uganda—have recognized the need to better understand the extent and value of their natural capital.

In East Africa, as in most places, wildlife landscapes do not stop at national boundaries; such efforts therefore need to be regional. Indeed, most of the iconic wildlife landscapes of the region share boundaries among its six countries, as well as with their neighboring countries such as Ethiopia and the Democratic Republic of the Congo (DRC). It will be crucial for East African nations to work with their neighbors to deal with conservation challenges.

To this end, USAID/Kenya and East Africa (KEA) has funded the Economic Valuation of Natural Capital in East Africa Project to explore the relationship between biodiversity, ecosystem productivity and ecosystem services, and economic prosperity and economic growth within transboundary landscapes in East Africa. The project is undertaking an inventory and valuation of wildlife resources in order to generate accessible and actionable evidence to better demonstrate the economic value of wildlife and wildlife habitats in the EAC, and to support stakeholders in using findings to strengthen policies for the conservation and management of transboundary natural capital. This report forms part of the broader study.

The project supports USAID/KEA and the EAC strategic priorities for harmonization of policy and legal frameworks, sustainable management of key transboundary ecosystems, and anti-poaching and combating wildlife trafficking, as well as learning and leadership for biodiversity conservation. This study involves working in close collaboration with the EAC to prioritize resources for conservation and in supporting transboundary natural resource management. There is a need for both an increased real and perceived value of natural capital for the region to achieve sustainable growth. Results from this study will enable institutions and organizations (*e.g.*, regional institutions like the EAC, relevant East African research institutes, development partners, and non-government and civil society organizations) to engage in evidence-based policy analysis and advocacy functions on conservation and the management of natural capital. This study also supports a key deliverable under the EAC Strategy to Combat Poaching, Illegal Trade, and Trafficking of Wildlife and Wildlife Products.

[1] Aichi Target 2 states that by 2020, at the latest, biodiversity values should have been integrated into national and local development and poverty reduction strategies and planning processes.

SCOPE OF THIS STUDY

The aim of this technical report is to provide a baseline description and valuation of the wildlife and wildlife habitats of four selected transboundary landscapes in East Africa, and to discuss the potential consequences of a business-as-usual scenario in which nothing is done to change observed trends, and in the light of projected climate change and the post-COVID-19 economic outlook. The next phase of the study will consider possible policy directions and courses of action to obtain the most advantage from these unique areas through a more detailed scenario analysis.

INTRODUCING THE LANDSCAPES

In consultation with the EAC Partner States and stakeholders, four large transboundary landscapes were broadly identified for inclusion in the study (Figure 1; see Appendix 1 for details). These are the **Great East African Plains** of southern Kenya and northern Tanzania; the **Northern Savannas** of South Sudan, Uganda, and Kenya; the **Albertine Rift Forests** along the Albertine Rift Valley of Burundi, Rwanda, and southwest Uganda; and the **Rweru-Mugesera-Akagera Wetlands** of northern Burundi, eastern Rwanda, and northwest Tanzania. Within each of these, wildlife landscapes were delineated based on protected area boundaries, possible migration corridors, and surrounding contiguous areas of primarily natural land cover where wildlife are likely to persist.

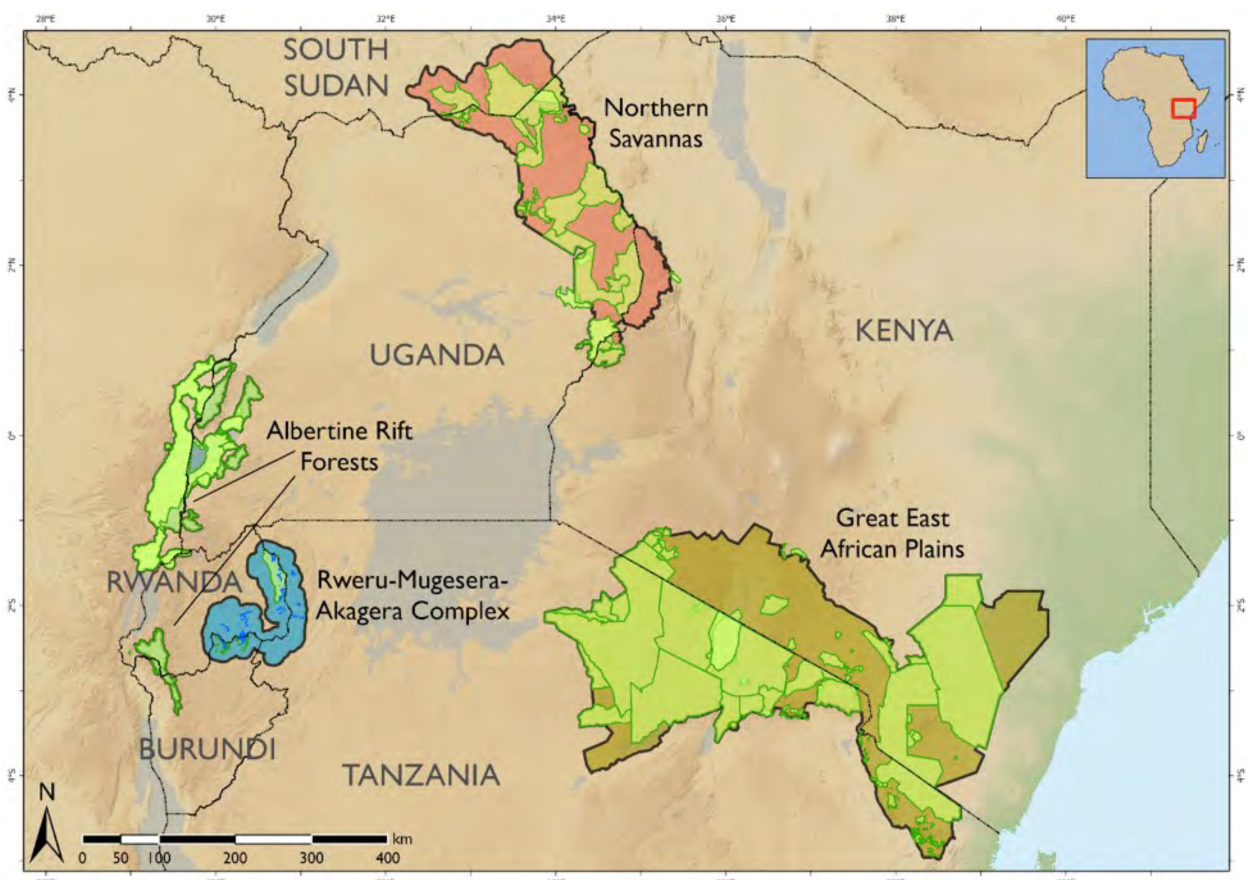


Figure 1. Location of the four selected transboundary wildlife landscapes, shown in different colors, with protected areas (per UNEP-WCMC & IUCN 2020) in light green

The four landscapes are briefly described below:

The Great East African Plains (129,634 km^2): This is a savanna-grassland-dominated landscape straddling Tanzania and Kenya. It includes several iconic protected areas along the Kenya-Tanzania border, such as the Serengeti-Mara, Ngorongoro Conservation Area, Arusha National Park, Amboseli-Kilimanjaro conservation area, and Tsavo-Chyulu Hills region. The landscape is home to 80 percent of the large mammals found in Kenya and Tanzania, and generates significant tourist revenue.

The Northern Savannas (48,848 km^2): This is a savanna-woodland landscape in northeast Uganda and extending into South Sudan to the north and Kenya to the east. It includes the adjacent Kidepo Game Reserve/Kidepo Valley National Park complex and the Nimule National Park/Otze Forest Reserve in Uganda and South Sudan, the Karamoja cluster conservation areas in Uganda and their neighboring community conservancies in Kenya, and Mount Elgon National Park on the border between Kenya and Uganda. This landscape includes sites critical for the protection of globally or nationally threatened habitats, and therefore is a priority for investment by the Uganda Biodiversity Fund.

The Albertine Rift Forests (7,772 km^2): This landscape comprises the remaining forest areas of the Albertine Rift Valley in Uganda, Rwanda, and Burundi that are largely connected to one another via protected areas in bordering DRC. The landscape includes Queen Elizabeth National Park in Uganda, the Greater Virunga Landscape between Uganda and Rwanda, and the Nyungwe-Kibira transboundary area between Rwanda and Burundi. This landscape contains 52 percent of all bird species and 39 percent of all mammal species found in Africa, including many endemics and most of the world's remaining population of mountain gorillas.

The Rweru-Mugesera-Akagera Wetlands (2,146 km^2): This landscape comprises the interconnected transboundary wetland complexes encompassing the Lacs du Nord protected landscape in Burundi, the Akagera National Park in Rwanda, and adjacent areas of Tanzania. The swamp-fringed lakes contain incredible biodiversity and include rare species like the shoebill stork. More than 400 bird species have been recorded here.

STRUCTURE OF THE REPORT

The document is structured as follows:

- Chapter 2 outlines the concepts of natural capital, ecosystem services, and value;
- Chapter 3 describes the overall approach and assessment methods;
- Chapter 4 provides the regional context by describing East Africa in terms of topography, geography, climate, environment, and socioeconomics, as well as a brief outline of future projections for population and climate;
- Chapters 5 to 8 provide the baseline descriptions of each of the four wildlife landscapes. For each wildlife landscape area, we describe:
  - The key wildlife habitats and wildlife, including core areas, critical habitat, and connectivity;
  - The people living in and around these habitats and their main livelihood activities;
  - The ecosystem services generated by the wildlife landscapes and their contribution to livelihoods and economic production; and
  - The main threats and their likely trajectories and impacts under a BAU scenario.
- Chapter 9 is a short concluding chapter proving a comparative overview and suggesting some of the potential policy implications and next steps for the study.

The appendices include further information on:

- How the study regions were selected;
- The delineation and grouping of habitat types;
- The data and methods used for the valuation of harvested resources and hydrologically linked services; and
- Projections of climate change impacts on vertebrate taxa, including selected species.

CONCEPTUAL FRAMEWORK

NATURAL CAPITAL ASSESSMENT

This study assesses the value of four key areas that support significant wildlife populations in East Africa. These areas can be considered natural capital assets in that they also provide significant economic benefits and contribute to human welfare. Natural capital is broadly defined as the stock of all-natural resources, including minerals, soils, air, water, and all living organisms. Ecosystems, and hence wildlife habitats, are an element of natural capital. The various structural and organizational characteristics of the ecosystems in these landscapes determines their capacity to supply a range of ecosystem services that generate these benefits.

It should be noted that over time the concepts of ecosystems and natural capital have expanded to incorporate human-modified ecosystems such as cultivated fields and urban parks as well as natural ecosystems. In reality, there is a continuum of human interference, and it is not always easy to draw clear distinctions between natural and anthropogenic ecosystems. In this study, we focus on natural or largely natural systems as wildlife habitats, including areas that are used for livestock or resource extraction.

This natural capital assessment, while not an accounting exercise (the output of which is a set of accounting spreadsheets and cross-tabulations), aligns with the building blocks of natural capital accounting and the System of Environmental Economics Accounting - Experimental Ecosystem Accounts (SEEA EEA; UN 2014) in that it involves delineating ecosystems in a defined spatial area, assessing their condition, estimating their capacity for delivering ecosystem services, estimating the actual use and value of those services, and finally estimating the value of the ecosystem assets. As such, the valuation of wildlife and wildlife habitats has been undertaken in a way that will be compatible with and could contribute to natural capital accounting in East Africa.

Understanding how human activities affect natural capital, and how changes in ecosystem extent and condition affect economic outputs and human welfare enables policy and decision-makers to properly evaluate trade-offs and secure a sustainable future. This requires understanding the links between ecosystem structure and function and the supply of ecosystem services, and how these services contribute to human wellbeing.

ECOSYSTEM SERVICES

Ecosystems can be described in terms of their structure and organization, such as their species composition, woody biomass, etc., and it is this structure and organization that determines their functioning, resilience, and productivity. The structure and organization of ecosystems can also be thought of as their *attributes*, which determine how they are used or appreciated for purposes such as recreation, religious ceremonies, or sense of place. Their productivity is what sustains the harvesting of living resources for a range of purposes and can be thought of as the supply of *goods* by ecosystems. Their functioning supports human activities beyond the ecosystem, from crop production to water supply, which can be thought of as the supply of *services* by ecosystems. The attributes of ecosystems and their capacity to supply goods and services are strongly linked to ecosystem condition. The concepts of ecosystem attributes, goods, and services are now more commonly referred to as cultural, provisioning, and regulating services, respectively (Figure 2).

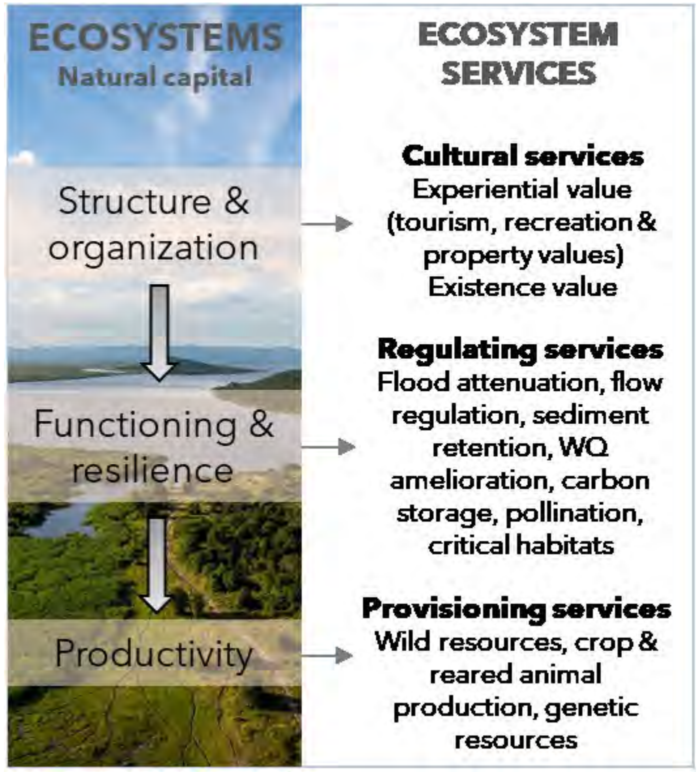


Figure 2. The link between ecosystems and the services they provide

Source: Based on Turpie et al., 2001

The understanding of ecosystem services and their valuation has advanced considerably since the concept first emerged in the 1980s and 1990s (Ehrlich & Mooney, 1983; Costanza *et al*., 1997; Daily, 1997). Over the years, several conceptual frameworks and classification systems for ecosystem services have been proposed. The most commonly used classifications are those of the Millennium Ecosystem Assessment (2003, 2005), The Economics of Ecosystems and Biodiversity (TEEB) classification (2010), the Final Ecosystem Goods and Services Classification System (FEGS-CS), the National Ecosystem Services Classification System (NESCS) proposed by the U.S. Environmental Protection Agency (Landers & Nahlik, 2013; U.S. EPA, 2015), and the Common International Classification of Ecosystem Services (CICES; Haines-Young & Potschin 2013, 2017). In this study, we largely follow the approach and terminology currently adopted by the System of Environmental Economics Accounting (SEEA) for the definition and classification of ecosystem services, which was initially informed by CICES (see UN 2017). These pertain to anthropogenic as well as natural systems.

Ecosystem services provided by natural wildlife habitats and the benefits derived therefrom are briefly discussed in the following sections and summarized in Table 1. The discussion also touches briefly on some aspects of their assessment in practice.

Table 1. A summary of the main types of ecosystem services from natural and modified ecosystems

BROAD CATEGORY	ECOSYSTEM SERVICE	DESCRIPTION AND PHYSICAL MEASURE
Provisioning services	Production of wild biomass*	Wild natural resources harvested from ecosystems for subsistence or small-scale production, in terms of kg or m^3 per hectares per year.
	In situ ecosystem inputs to reared animal production*	Numbers of livestock supported per ha, standardized in terms of large stock units per ha.
	Genetic resources	Genes and varieties obtained and their influence on pharmaceutical sales and crop and livestock production.

BROAD CATEGORY	ECOSYSTEM SERVICE	DESCRIPTION AND PHYSICAL MEASURE
Cultural services	Experiential value associated with active or passive use*	Experiential fulfillment associated with active or passive use, through any type of activity ranging from adventure sport to birdwatching to religious activities or cultural ceremonies.
	Existence value	Fulfillment associated with knowledge of existence for intrinsic value or for present or future generations.
Regulating services	Flood attenuation	Smoothing of fluvial flows during storm events through interception, infiltration, storage, and landscape roughness; reducing the flood peak volume, velocity, and flood height in the receiving area; and reduction of coastal flooding by the sea through dampening storm surges and limiting run-up distance by coastal ecosystems such as coral reefs, mangroves, and dunes. Estimated in terms of flooding characteristics under different storm return periods or categories.
	Seasonal flow regulation*	Smoothing of flow over the longer duration through infiltration and storage, reducing need for storage to achieve a given yield. Measured in terms of higher dry season flows relative to without-service situation.
	Sediment retention*	Reducing soil loss and sediment transportation to downstream environments through holding soils *in situ* (by vegetative cover) or through trapping eroded sediments (by slowing down movement of water through the landscape, *e.g.*, in a wetland). Measured in terms of the difference in amount of sediment retained (m^3 per year) at key points between the observed land cover and a situation of bare and degraded landscape (for wetlands, this means loss of holding capacity).
	Water quality amelioration*	Reducing nutrients transported to downstream environments as a result of uptake in the environment. Measured in terms of the difference in the nutrient loads (kg per year) delivered at key points between the observed land cover situation and a situation of fully transformed and degraded landscape (for wetlands, this means loss of holding capacity).
	Carbon storage and sequestration*	Stocks of carbon in each time period, expressed as tons of carbon per ha; annual additions and subtractions are not estimated but net changes are tabulated between two time periods.
	Pollination* and pest control	Pollination of crops and control of crop pests by animals living in surrounding environments. Measured as difference in output of the serviced areas. Note that this requires attributing some of the ecosystem inputs to crop production to surrounding habitat rather than the land under crops.
	Critical habitat for fisheries and wildlife	Provision of critical habitat for populations that are utilized in other locations, such as fish nursery areas, wildlife breeding areas, or migratory staging areas. As for the above service, this requires attributing some of the ecosystem inputs to these activities to the critical habitat areas rather than the areas in which the activities take place.

* Included in this study.

Source: Turpie *et al.*, 2020

PROVISIONING SERVICES

Provisioning services are the harvestable resources supplied by ecosystems, such as wild foods, raw materials, and forage for livestock production. Harvest can include commercial extraction of natural resources such as timber. It can also include subsistence use or small-scale production of, for example, wild plants and animals for nutrition, health, energy, and raw materials. The latter uses occur particularly where there are limited economic opportunities and generally go unrecorded.

The valuation of provisioning services is usually based on estimates of actual use. The value is therefore determined based on factors such as the available stocks of resources, the institutions that govern access to them (*i.e.*, property rights) and the demand for the resources. For small-scale or subsistence users, demand is influenced by socio-economic circumstances of households and prices of alternatives.

In assessing provisioning services, ideally one should consider stocks and sustainable yields as well as use. If only use is measured, and that use exceeds the sustainable yield, then one might overestimate the value of the system if stock changes are not taken into account and it is assumed that such use can be sustained. Similarly, the status of stocks should be assessed, so that low levels of use can be correctly ascribed to supply or demand factors, and appropriate management interventions can be applied in the former case.

It should be noted that this study does not list water as a provisioning service because it is not produced by ecosystems. Rather, it regards ecosystem services pertaining to water supply as those that regulate the timing and location of water flows, which affect the costs of collecting water for use (as described below under "regulating services"). To also regard water as a provisioning service would lead to double counting. The study also does not include mineral resources as ecosystem services.

CULTURAL SERVICES

Cultural services obtained from ecosystems include the contributions that ecosystems make to aesthetic, recreational, spiritual, scientific, and educational fulfillment derived from a range of activities or from the passive appreciation of nature. These "experiential" benefits or "use values" are gained through any type of activity ranging from adventure sports to birdwatching, religious or cultural ceremonies, or just passive observation. In addition, cultural services include the satisfaction obtained from the knowledge of the existence of biodiversity and its possible enjoyment by future generations. These less tangible "existence" and "bequest" values, or "non-use" values, are not associated with direct visitation or observation and can be global in extent. In both cases, the capacity for ecosystems to supply these services is determined by characteristics such as beauty, rarity, species present, and condition.

The cultural or amenity value of landscapes and ecosystems is derived from their combinations of natural attributes, such as size, beauty, and rarity, and human-made enhancements such as roads, waterholes, viewpoints, and other tourism infrastructure. These attributes determine the extent to which each area is suitable or attractive for recreational use, religious use, or spiritual fulfillment. Their value or actual contribution to human welfare also ultimately depend on factors that influence demand for these services, such as the number and income levels of people living in the vicinity, as well as by people living elsewhere. The amenity derived from these ecosystems and landscapes can come in the form of tourists enjoying visiting an area, people using and gaining wellbeing from the ecosystems in which they live, or people gaining enjoyment from knowing that such ecosystems and landscapes exist.

Because of the intangible nature of many of these values, the welfare gains generated by the supply of cultural services are difficult to estimate and map (Milcu *et al.*, 2013). In theory, the value of the cultural services provided by existing natural areas is what people would demand in compensation for giving up the benefits they receive from those areas. This can be estimated through the use of stated preference methods, involving surveys of users that elicit their willingness to pay to retain or willingness to accept compensation to forgo a certain state of the world. However, the inherent methodological biases of these methods can be extremely challenging and require very comprehensive studies.

Nevertheless, the welfare gains associated with the amenity of ecosystems and landscapes are reflected to a large extent through the tourism market, which is tangible and observable. Visitors pay to travel to and stay in an area where they will have access to or views of different types of amenities. Another potential manifestation of amenity value is in the premiums that people pay for properties that are close to the areas they wish to visit or that have an enjoyable view. This type of value is usually realized within urban areas, where access to natural areas is more limited. In relatively remote wildlife landscapes, tourism value is typically the dominant economic value, apart from non-use value, which is potentially global and much larger.

REGULATING SERVICES

Regulating services are the functions that ecosystems and their biota perform that benefit people in surrounding, downstream, or even distant areas. These services include carbon sequestration and storage, which reduces the potential impacts of climate change through the active removal of carbon from the atmosphere by vegetation growth or through the passive retention of carbon stored in the landscape by avoiding deforestation. This category also includes the pollination of crops in nearby fields by insect pollinators that are supported by natural habitats. This is particularly important for the low-input, small-scale production systems common in the study areas. Three types of regulating services are strongly linked to catchment geography, hydrology, and land use: the regulation of water flows, control of sediments, and uptake of nutrients that affect water quality. These more complex services are described in more detail below.

CROP POLLINATION

Pollination services are widely recognized as critical for human wellbeing and survival given their vital role in ensuring food security. However, the value of wild pollinators remains unclear. This is of concern for sub-Saharan Africa, a region highly dependent on subsistence agriculture as a main source of livelihood (Tibesigwa *et al.,* 2019). The presence of wild pollinators is directly linked to natural vegetation (Kremen *et al.*, 2004), which plays a critical role in certain life cycle stages of pollinator species, such as through the provision of nesting sites or forage at certain times of year. Insects are responsible for 80-85 percent of all pollinated commercial crops, which represents about one-third of global food production (Allen-Wardell *et al.*, 1998; Klein *et al.*, 2007).

Much of the agricultural production in and around the study areas is small-scale production involving a relatively low level of inputs. While these farmers focus predominantly on wind-pollinated crops such as maize, many farmers also grow a range of fruit and vegetables that are almost exclusively pollinated by wild pollinators, mostly bees. Although often not the dominant crops by areal extent, these insect-pollinated fruit and vegetable species may make a valuable contribution to the dietary diversity and micronutrient intake of smallholder farmers (Jones, 2017). The loss of this pollination service could thus have a significant impact on the health of rural families. The crop pollination service is defined as the increase in crop production in pollinator-dependent crops that are supplied by the natural ecosystem assets surrounding cropland to the economic user of the land (*i.e.*, the farmer, Horlings *et al.,* 2020). The economic benefit is therefore increased crop production. The wild pollination service is primarily provided by surrounding natural habitat rather than the land under crops. Therefore, pollination is counted as an input from surrounding ecosystems.

CARBON SEQUESTRATION AND STORAGE

It has been estimated that climate change, caused by increases in the emissions of greenhouse gases such as carbon dioxide, will carry a cost of about 2-7 percent of GDP in different parts of the world by 2050 (Fankhauser & Tol, 1997). Natural systems are understood to make a significant contribution to global climate regulation through the sequestration and storage of carbon. About half of the biomass of vegetation, both above and below ground, comprises carbon. Furthermore, carbon accumulates in the soils as a result of leaf litter. The capacity for carbon sequestration and storage therefore varies between different types of ecosystems and in different locations. When natural systems are degraded or cleared, much of this carbon is released into the atmosphere, especially if the degradation is for fuel wood production or due to burning for grazing (Hoffa *et al.,* 1999). These emissions contribute to global climate change, which is expected to lead to changes in biodiversity and ecosystem functioning, changes in water availability, more frequent and severe droughts and floods, increases in heat-related illness, and impacts on agriculture and energy production (IPCC, 2007). These impacts will affect economies and human wellbeing on a global scale, but more so in developing countries that are more reliant on land and natural resources (Tol, 2012). Adaptation to these changes could come at a high cost. The conservation and restoration of natural systems thus helps to reduce the rate at which greenhouse gases accumulate in the atmosphere and the consequent impacts of climate change. This benefits the countries involved as well as the rest of the world. To some extent, the benefits to the rest of the world can also elicit local revenues through mechanisms such as the UN's Reduced Emissions from Deforestation and Degradation (REDD) program.

Concerns about the loss of natural systems and the impacts of climate change have motivated efforts to quantify the role and value of these ecosystems in the global carbon cycle and have also encouraged international efforts to retain and restore woody biomass in natural systems. As signatories to the United Nations Framework Convention on Climate Change (UNFCCC), the countries of East Africa will need to consider this in meeting their commitments.

The benefit of both sequestration of carbon from the atmosphere and limiting the release of stored carbon through ecosystem degradation is a reduced impact of climate change because of decreased concentrations of carbon dioxide in the atmosphere. Termed the "social cost of carbon," the damages that would be incurred under climate change are typically estimated in terms of changes in GDP, which is therefore a directly compatible measure for ecosystem accounting. An alternative way to value the service is using its value in markets that have developed as a result of government and private efforts to "neutralize" carbon emissions. Some studies do both (Horlings *et al.,* 2019). In this study, the social cost of carbon was preferred because the marginal price of carbon in markets is not realistic at scale.

FLOW REGULATION

Ecosystems can reduce variation in downstream river flows over the longer duration through infiltration and temporary storage in the catchment areas, reducing the need for built storage to achieve a given yield through the year. Seasonal variation in river flows is primarily determined by seasonal patterns in rainfall, with higher flows being experienced in months of higher rainfall. However, the seasonal variation in surface runoff from a river basin may be lower than the rainfall variation, since some of the rainfall that falls during the rainy season percolates into the ground, flows underground at a slower rate than surface flows, then enters rivers further downstream via springs or seepage areas. These groundwater-derived flows, or base flows, help to maintain river flows during periods of lower rainfall.

Flow regulation through infiltration and temporary storage of rainfall is likely to be more important where there is high seasonality in rainfall patterns, and especially where demand is strongly seasonal, such as for irrigation during the dry season, or where people depend directly on rivers for their water supply. Where seasonal variability is high, the amount of water available for use in the low flow period can be increased by building reservoirs that capture water during high flow periods.

The more built storage capacity there is in a basin, the more water there is available for dry season use, and the greater the yield as a proportion of total runoff. For a given streamflow, there is a relationship between reservoir capacity, the yield obtained from the reservoir, and the reliability of this yield (assurance of supply, usually expressed as return period or percentage of years in which the yield is not obtained; Vogel *et al.,* 1999, 2007; McMahon *et al.,* 2007). In general, a greater variation in runoff would mean that more storage capacity is needed to obtain a given yield, with all else equal (Turpie *et al.*, 2020). Furthermore, smaller reservoirs are relatively more sensitive to intra-annual variation in flows, which is the component of variation that is more likely to be influenced by land use and ecosystem characteristics in the catchment areas. Particularly vulnerable are the run-off river users, who have very small storage (*e.g.*, a weir) or no storage capacity.

Ecosystems can reduce temporal variation in water flows, particularly on an intra-annual basis, relative to the variation in rainfall. Without this service, dry season flows would be expected to be lower, increasing the need for storage. Therefore, water supply infrastructure, and reservoir capacity in particular, can be treated as a substitute for the service provided by ecosystems.

EROSION CONTROL

Vegetative cover prevents erosion by stabilizing soil and intercepting rainfall, thereby reducing its erosivity (De Groot *et al.,* 2002). This is particularly valuable where soils are highly erodible. Vegetated areas, especially wetlands, may also capture the sediments that are eroded from agricultural and degraded lands and transported in surface flows, preventing them from entering streams and rivers (Blumenfeld *et al.,* 2009, Conte *et al.,* 2011). This protects downstream areas from the impacts of sedimentation, which can include impacts on water storage capacity, hydropower generation, and navigability of rivers (Pimentel *et al.,* 1995), as well as the integrity of downstream lakes or coastal ecosystems. In urban contexts, elevated sediment loads also have to be removed from sewerage systems, storm water drainage systems, and harbors. The extent to which sediments end up in river systems is determined by several factors including soils, rainfall patterns (amount and intensity), slope, and the type and amount of vegetative cover.

WATER QUALITY AMELIORATION

Nutrient-enriched runoff from agricultural and urban land can have a negative impact on the water quality of downstream aquatic ecosystems. The excess nutrients introduced to these systems can change their trophic status in a process known as eutrophication. This is usually accompanied by increased abundance of algae and plant growth, which changes the nature and composition of these systems, affecting the benefits that can be derived from them. At extremes, it can lead to toxic algal blooms, loss of dissolved oxygen, and fish kills. Still water bodies, such as reservoirs and lakes, are particularly susceptible to this type of degradation. Where water is collected or extracted for drinking water supply, the elevated levels of algae, as well as nutrients and suspended sediments, increases the costs of water treatment. In the study area, natural freshwater lakes are particularly important both for water supply and for fisheries. The lakes in the region have been heavily affected by nutrient enrichment, including Lake Victoria (see Box 1).

Box 1. Eutrophication of Lake Victoria

Lake Victoria is some 68,800 km^2 and has a catchment area of about 194,000 km^2. The catchment area includes parts of the lakeshore countries of Tanzania, Uganda, and Kenya, as well as parts of Rwanda and Burundi. The lake, which is 40m deep on average, receives 80 percent of its water input directly from rainfall and the rest from catchment runoff (Mugidde, Hecky & Ndawula, 2005). This is balanced by evaporation and outflows into the White Nile River at Jinja, Uganda.

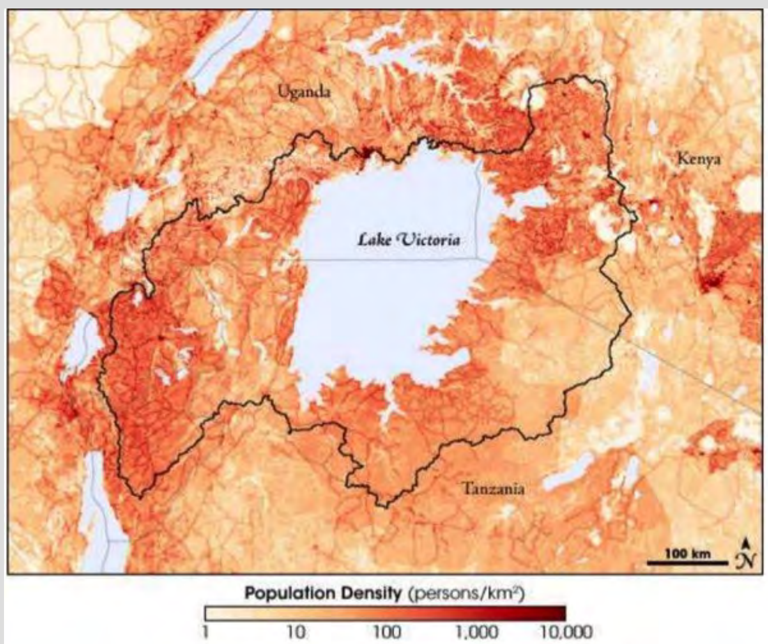


Figure 3. Lake Victoria and its catchment area, showing population density

Source: Kolding *et al.*, 2014

Lake Victoria is an important source of fresh water and fisheries to the surrounding countries. Major changes to the lake as a result of growing human interference started being observed in the 1950s. Land-based activities such as deforestation, increasingly intensive agriculture, and urban expansion led to increased nutrient loading and progressive enrichment of the lake (Scheren, Zanting & Lemmens, 2000). This led to changes in the phytoplankton composition (Gophen, Ochumba & Kaufman, 1995), frequent algal blooms (Ochumba & Kibaara, 1989), deoxygenation of the water, and periodic large-scale fish kills (Hecky *et al.*, 1994). Overfishing since the 1920s (Kolding, Mkumbo & Zwieten, 2013), the introduction of the exotic and highly predatory Nile perch *Lates niloticus* in the 1950s (Goudswaard, Witte & Katunzi, 2008), and deteriorating water quality led to the decimation of endemic cichlid species by the 1980s (Witte *et al.*, 1992). The eutrophication of the lake also fueled the invasion of water hyacinth *Eichhornia crassipes* after its first appearance in 1989. These changes further eroded the benefits formerly obtained from the lake.

Since the 1980s, the Nile perch, once a top predator, has lost its dominance. Also, there have been some improvements in water quality in the open waters of the lake, possibly due to the resurgence of smaller fish species, including phytoplankton feeders (Sitoki *et al.*, 2010). Nevertheless, water quality has remained a problem. For example, algal blooms have persisted as a result of loads of phosphorous and nitrogen entering the lake from human activities in the catchment area (mainly agriculture and sewage) that stimulate the growth of algae and other aquatic plants (Mugidde *et al.*, 2005).

Total phosphorous in the lake has continued to increase in line with population growth (Kolding *et al.*, 2013). At the same time, the total fish catch has increased dramatically, largely because of increasing effort. However, the experimental catch per unit effort of bottom feeding (demersal) species (measured in kg per hour) decreased from the mid-1970s and has remained low since the mid-1980s, and fish composition has changed from being dominated by native cichlids, which were mostly caught close to shore, to introduced species (mainly Nile perch), to the native small, silver cyprinid *Rastrineobola argentea* known as *dagaa*, *mukene*, or *omena*, which is mostly caught offshore. Thus, while total catches continued to increase thanks to adaptation of fishing behavior to the changing ecosystem, what is not clear from the statistics is the probable decrease in inshore fish biomass that occurred over the same period.

Sixty-eight percent of nitrogen and 44 percent of phosphorous is introduced from the atmosphere via rainfall, 1-2 percent is from wastewater treatment plants, and most of the remainder enters via rivers from the catchment area (Agwanda & Iqbal, 2019). Phosphorus is typically a limiting nutrient for algal growth; increasing its availability therefore supports increased algal production. Although the exact relationship between nutrient loads entering the lake and the productivity of fish is unknown, it ultimately leads to fish die-offs. The problem is recognized as being severe enough that engineering solutions have been considered to reduce these loads (Agwanda & Iqbal, 2019).

Indeed, recognizing the importance of Lake Victoria to the shoreline countries, several international initiatives and agreements have been introduced to secure the integrity of the lake ecosystem. For example, the Lake Victoria Environmental Management Program was initiated in 1997 as a regional effort to address its environmental challenges. The project is now in its third phase, and involves all the countries situated in the basin (Burundi, Kenya, Rwanda, Tanzania, and Uganda) (World Bank, 2018).

Natural vegetation can help mitigate the effects of anthropogenic nutrient enrichment of aquatic ecosystems. Some of the nutrients in this enriched runoff can be removed when it passes through natural vegetation and wetlands in the landscape, ameliorating the pollution problem before it reaches downstream ecosystems and locations where water is abstracted for use. Together, natural vegetation's active and passive services are valued as the costs avoided as a result of retaining the ecosystem in its natural condition. The ecosystems' removal of pollution through ecological process such as vegetative growth is the *active* aspect of the service. The capacity to perform this active service will be linked to the characteristics and condition of the ecosystem, and the use of the service will depend on the amount of anthropogenic activity upstream of the ecosystem. In addition, retaining ecosystems in their natural state, as opposed to replacing them with alternative systems that support, for example, agriculture or human settlements, usually maintains a higher quality of water leaving the area than if the transformation took place. This is the *passive* aspect of the service. Therefore, the demand for active and passive services comes from the users of water downstream of these ecosystems and both contribute to their value.

CONCEPTS AND MEASURES OF VALUE

This assessment uses Total Economic Value typology to estimate the value of ecosystem services. This framework identifies four primary types of value that nature can produce: direct use value, indirect use value, option value, and non-use value. Direct use value is linked to provisioning and some cultural values such as recreation and tourism. Indirect use value is linked to regulating services and is indirect because these services provide inputs into benefits derived beyond the ecosystem in question. In this context, non-use values would be linked to the feeling of satisfaction from knowledge of the continued existence of elements of biodiversity, without necessarily involving active use or observation. This value is closely linked to the concept of intrinsic value, which is the value of biodiversity in itself.

There are different ways to evaluate how changes in the environment affect people. Economic analysis such as cost-benefit analysis considers how choices affect people using welfare measures of value. This measurement is the sum of producer surplus (net earnings to producers) and consumer surplus (the benefits that people derive from consumption, over and above what they had to pay for it). Many policymakers are more familiar with measures of economic production from the national accounts, such as gross value added (GVA) and GDP. These effectively estimate the total value of goods and services produced, and by extension, the total income generated for the various actors in the economy. In natural capital accounting, values of ecosystems and their services are presented in terms of "exchange value," which is compatible with national accounts (UN, 2017). This does not include consumer surplus and is not a measure of welfare. In this study, we focus on GVA.

Importantly, ecosystem values are generated through the combined use of natural and human-made and other capital. For example, if one further develops the tourism infrastructure of a location or invests in more marketing, the tourism value may increase. It is useful to determine how changes in natural capital affect its services, holding other inputs constant. In a baseline analysis, however, it is most practical to focus on the value of the benefits derived with the combination of nature and other inputs rather than to try and attribute the value of a single component.

In general, cultural and provisioning services are used purposely, through joint contribution of natural and human-made capital and labor, and the resulting benefit was valued in terms of GVA (which is a contribution to GDP). GVA is calculated as the output value minus intermediate expenditure on inputs and includes all the income generated to the owners of factors of production. Regulating services are used inadvertently, and since their loss could lead to damages or require the prevention of such damages through engineering solutions, they were valued in terms of avoided costs.

APPROACH AND METHODS

DATA COLLATION AND REGIONAL OVERVIEW

Information on the wildlife and ecosystem characteristics of the four transboundary landscapes, and for the East African region as a whole, was collated and reviewed to understand the landscapes' and regional context and to identify the nature and potential spatial geography of ecosystem services' supply and demand. This involved the collation of global, regional, and national spatial datasets relating to the geography, biodiversity, and people of East Africa. These datasets were also interrogated for their potential usefulness for the more detailed aspects of the study, considering the need to conduct transboundary analyses. In general, datasets that spanned the region were chosen in preference to local datasets. Based on the initial assessment of data availability, the most recent year possible for the valuation baseline was deemed to be 2018, as this was the latest available date for certain key datasets at the time of project initiation. Thus, this baseline study was based on land cover data for that year.

DELINEATION OF ECOSYSTEM TYPES

Following the data collation and regional overview, a detailed analysis was undertaken at a regional (East Africa) scale of spatial data on vegetation types, ecoregions, land cover, and biomass in conjunction with relevant ecological literature. Estimating the supply of ecosystem services across the landscape requires a detailed representation of land cover and vegetation characteristics, as these have a strong influence over the spatial distribution and value of services provided. Our chosen ecosystem delineation method involved a combination of the land cover, vegetation type, and land condition datasets. We felt that combining these datasets to produce a single map of habitat types would produce the most accurate possible representation of habitats across East Africa for the purposes of ecosystem service valuation. We also used the IUCN's Global Ecosystem typology to aid with the grouping of the habitat types we identified.

The following datasets were used to produce our habitat map:

- Copernicus 100-meter land cover data for 2018 (Copernicus Global Land Operations);
- Potential natural vegetation map for East and Southern Africa (van Breugel *et al.,* 2015) (WWF Ecoregions map used instead for South Sudan); and
- Change in the normalized difference vegetation index (NDVI) between 2001 and 2018 (Trends.Earth).

The land cover data uses satellite imagery to provide a snapshot of current land use and vegetation structure, whilst the vegetation map provides a static depiction of the distribution of floral communities before the influence of humans. We selected the Copernicus 100-meter land cover data, as it had consistent coverage of the whole region, and distinguishes more land cover classes than many other global land cover datasets. For example, it distinguishes between multiple types of open and closed forest classes, while other global land cover datasets may only have a single land cover class for forest. The potential natural vegetation map for East and Southern Africa (see van Breugel *et al*., 2015) was found to be the most comprehensive and consistent vegetation dataset available for the region.

However, it did not include South Sudan, where we instead used the WWF Ecoregions map. Finally, we used NDVI data from *Trends.Earth* as the best available regionally consistent proxy measure of ecosystem condition. See Box 2 for further information on land cover data sets, the meaning and calculation of NDVI, and how the various data were combined in making the habitat map.

Combining the land cover and the vegetation map enabled us to enhance the detail of areas with natural land cover classes when making our habitat map. For example, use of the vegetation map allowed us to identify specific woodland and forest types in areas mapped as being currently forested by the land cover data. For areas with human-modified land cover (*e.g.*, cultivation), the land cover dataset was prioritized. For regions where the original vegetation was mapped as forest or denser woodland, the land cover data was used to infer ecosystem condition *e.g.*, forest areas now classed as shrubland were considered degraded. Lower biomass vegetation types were classed as healthy or degraded based on an NDVI trend layer, with a negative trend indicating degradation.

The final classification comprised 72 habitat types across all regions, which includes a degraded and undegraded form of each natural habitat type where relevant (see Appendix 2 for the full list). There are 16 forest habitat types, 14 woodland habitat types, 12 grassland/wooded grassland habitat types, 19 bushland/shrubland habitat types, 5 aquatic habitat types, 4 desert/bare habitat types, and 2 anthropogenic types (cultivation built-up). The number of habitat types within each study area ranged from 19 in the Wetlands to 51 habitat types in the Great East African Plains.

Box 2: The building blocks of the habitat maps

The habitat map used in this study involved building a combined map using several overlapping datasets by making careful and logical decisions related to the ecological properties of the data represented in the datasets. Each of these datasets comprises information derived from a variety of sources and through numerous methods. Some information about these datasets, and how they are derived and interpreted is provided below.

DERIVING LAND COVER MAPS FROM SATELLITE IMAGERY

Land cover maps depict the earth's surface in terms of the physical natural and modified land cover or land use types, such as "grassland," "woodland," "urban residential," "roads," or "irrigated crops." Modern land cover maps are generally derived from satellite imagery. Satellites orbiting the earth take photographs of the earth's surface that are repeated at regular intervals. Many images are taken at the same time, using different sensors that focus on particular wavelengths of light (spectral bands). Each of these images is made up of thousands of little squares, or "pixels," with higher resolution images being made up of smaller squares. Each pixel records the pattern of light being reflected from that part of the earth's surface. The patterns of light are strongly related to the actual land cover, so that each type of land cover can be recognized in terms of its "spectral signature" (referring to which parts of the light spectrum are reflected more strongly). In some instances, the different types of land cover are very easy to discern from the spectral signature. For example, waterbodies have a very clear spectral signature. But in other cases, the signal varies a lot and it can be more difficult to discern different land cover types. Certain combinations of bands are better than others at detecting particular land covers. One has to analyze the spectral data from different pixels and group them into land cover types. This classification can be done using a computer algorithm that determines spectrally similar pixels from different satellite bands. One can either rely entirely on the computer algorithms to do this ("unsupervised classification") or it can be done with expert input, using additional information collected at ground level ("supervised classification").

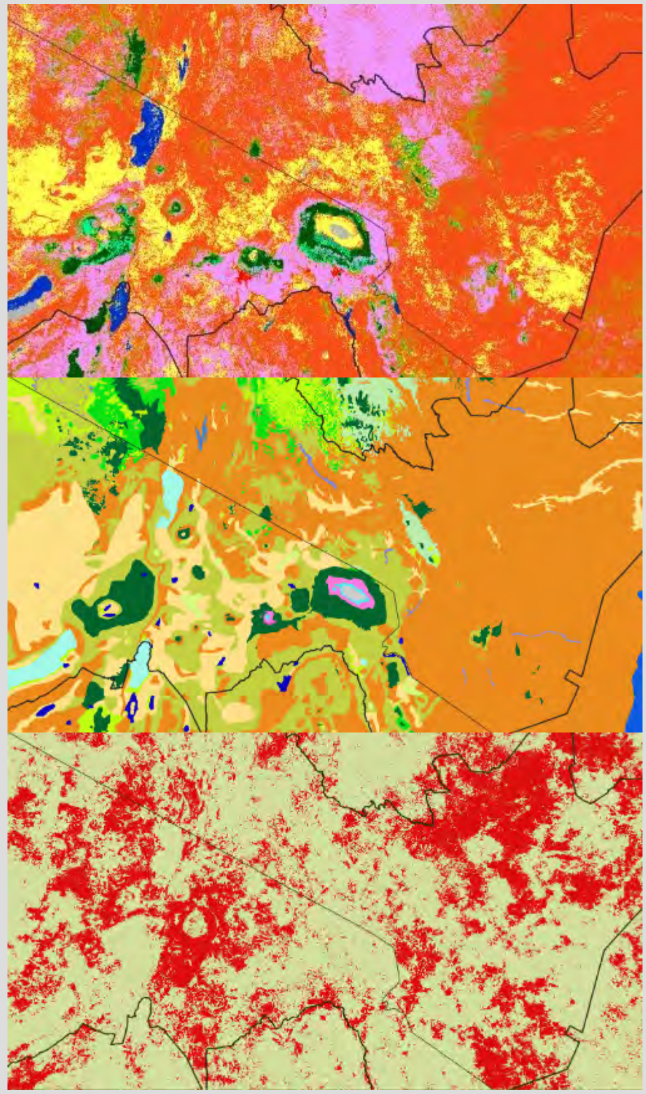


Figure 4. The three datasets that were combined to map habitat types across East Africa. Land cover (top, Copernicus), potential natural vegetation map for East and Southern Africa (middle, van Breugel et al.*, 2015), and change in NDVI (bottom, CI, 2018)*

Land cover/land use maps are useful for decision-making, and in an ecological sense, for determining changes over time using comparable datasets from different periods of time.

The imagery used in this study is of a relatively coarse resolution of 100 x 100 m. This is suitable for regional mapping and analysis due to its more rapid processing time, as opposed to high resolution data (< 30 m). Commonly used satellite data include Sentinel-1 and -2, Landsat, and PROBA-V (the names refer to the names of the satellites). In this study, the Copernicus land cover data at 100 m resolution was used.

ASSESSING VEGETATION CONDITION USING NORMALIZED DIFFERENCE VEGETATION INDEX

It is also useful to understand vegetation condition within a land cover class. One of the key indicators of degradation is land productivity, or the biological productive capacity of land, usually represented as net primary productivity (NPP, the net amount of carbon assimilated following photosynthesis in plants following metabolism, *i.e.,* the energy stored as biomass in plants). However, NPP is difficult and costly to estimate. Therefore, at large scales, the best way to do this is using the normalized difference vegetation index (NDVI). NDVI is determined from satellite imagery by calculating the ratio of visible and near-infrared light reflected by vegetation. Healthy vegetation generally reflects more near infrared-red light relative to visible light. A lower than average or negative trend in NDVI over time for a pixel can be used as an indication of degradation.

DELINEATION OF THE WILDLIFE LANDSCAPES

Following identification of the broad location of the target wildlife landscapes for the study, the boundaries of the wildlife landscapes were defined using spatial data. This was based on largely contiguous areas of natural habitats within a biome or broadly similar ecosystem types, in and around the key protected areas that had been identified. In reality, none of the wildlife landscapes fall purely within a single biome or major ecosystem type. For example, the Northern Savannas and Great East African Plains wildlife landscapes include patches of forest, while the Albertine Rift Forests landscape encompasses some areas of savanna woodlands. However, we avoided extending the wildlife landscapes into regions *dominated* by ecosystem types that were incongruous with the representative ecosystems associated with each region. For example, the Albertine Rift Forests landscape did not extend further north where savannas replace forest as the dominant biome. We used a variety of data sources when making such decisions, including the potential natural vegetation map, WWF Ecoregions, relief features, and climatic data. For example, the Northern Savannas landscape was delineated on its eastern boundary using altitude. East of this, the terrain descends into what might be delineated as a distinct rift valley landscape of its own.

Using land cover data, we also aimed to exclude extensive areas of contiguous cultivation (*i.e., in* excess of several km^2 with little to no natural habitat in between) and/or urban/built-up land from the wildlife landscapes, as these areas would not be suitable for habitation by wildlife, particularly for larger mammals. However, as we also aimed to delineate continuous wildlife landscapes as far as possible, inclusion of some areas of human habitation and agriculture was unavoidable. Furthermore, exclusion of any cultivation would not always be warranted, as some less intensively modified habitats might still serve as important wildlife habitat or corridors, particularly in regions where smallholder agriculture is interspersed with areas of more natural cover. Notably, a contiguous wildlife landscape could not reasonably be delineated for the Albertine Rift Forests, as extensive habitat conversion has resulted in a lack of connectivity between many of the remaining patches of wildlife habitat in this region, resulting in the landscape being composed of separated areas. While this study is confined to East Africa, it is critical to note that parts of this landscape are connected (and kept viable) via protected areas in the DRC. In other cases, decisions may have been more practical, such as using an administrative boundary for delineation of part of the landscape, although this has been kept to a minimum.

DESCRIPTION AND VALUATION OF ECOSYSTEM SERVICES

Ecosystem services were quantified in physical terms where appropriate and valued in terms of U.S. dollars per hectare per year. Our approach involved estimating the actual use and value of each service based on the estimated capacity of the different ecosystem types to deliver services, and the estimated demand for the services. The value of the benefits derived from use of the service were determined net of the inputs of human-made capital as far as possible.

While the focus of the study was to elucidate the value of the wildlife landscapes, the valuation study looked beyond the limits of these landscapes for certain services. In some cases, this served to provide some indication of how the ecosystem service values derived from the wildlife landscapes differ from those of the surrounding land areas, especially where there has been a conversion of land from similar wildlife habitats to alternative land uses. For example, cultural service values such as tourism would be expected to differ between the wildlife landscapes and surrounding transformed areas, and this could inform future scenarios under which land conversion continues to encroach on wildlife habitat. Another example is the use of provisioning services, which is expected to be higher in areas surrounding the landscapes as well as areas within the landscapes that are not formally protected, but that could also undermine the supply of other services when exploitation is above sustainable levels. In other cases, this was because the services were linked to hydrological processes, in which case it was necessary to consider the upstream and/or downstream drainage basins.

Our approach is spatial, because values depend on context and vary in space as well as time. The landscape capacity to supply services varies with topography, climate, ecosystem type, and condition, and the human demand for services also varies spatially, with population density, infrastructure, and location. Different spatial modeling and valuation approaches were used for the valuation of provisioning, cultural, and regulating services (Table 2). Our understanding and interpretation of the different ecosystem services in the region and the methods used for their quantification and valuation in this study are outlined in a separate section below, with methods described in more detail in Appendices 3 and 4.

Table 2. Summary of the valuation methods used for each ecosystem service

CATEGORY	ECOSYSTEM SERVICE TYPE	VALUATION METHOD USED
Cultural	Nature-based tourism	Tourism direct contribution to GDP (TDGDP) and geotagged photographs to assign value spatially
Regulating	Flow regulation	Avoided costs of water supply infrastructure for existing supply systems and avoided costs of obtaining water for those people that depend on instream flows for their domestic water supplies
	Erosion control/sediment retention	Replacement cost of the storage capacity that would be lost without the service
	Water quality amelioration	Extra costs of water treatment that would be incurred without the service
	Carbon storage and sequestration	Avoided damage costs using social cost of carbon
	Agricultural support services (pollination)	Benefit transfer, using a production function based on detailed plot-level panel data and surrounding land cover
Provisioning	Production of harvested wild biomass	Resource rent method (total revenue minus intermediate costs, labor costs, depreciation, and return on fixed capital)
	Livestock forage production	Livestock direct contribution to GDP and gridded livestock of the world database to assign value spatially

NATURE-BASED TOURISM

In the study areas, experiential values include everything from the local use of wildlife habitats for various recreational or cultural purposes to domestic and international tourism. While tourism statistics are often available, information on the local use of ecosystems generally goes unrecorded. Both the valuation of local cultural use and non-use values associated with nature is usually tackled using survey-based methods. While some international models have been developed to estimate the value of local recreational use of ecosystems, these are wholly unsuited to the study area.

In this study, we have focused on the tourism value of the wildlife landscapes, since this is the dominant value of the wildlife landscapes in monetary terms, and the only cultural value for which data could be collated within the scope of the study. This is not to say that the areas do not have local cultural significance. In fact, these wildlife landscapes have played a significant role in shaping the diverse cultures and livelihoods of the people that live in and around them and continue to be intricately linked to their lives. These connections are often deep and powerful, and best not reduced to monetary terms. They are also complicated, having changed with various influences over the years, shifting socio-economic circumstances, and increasing conflict with wildlife as populations grow and create greater competition. These are issues that need further study, and that will also be considered further in the next phases of this project. Tourism value was estimated both in terms of benefits to the region (with value added to GDP as a proxy for producers' surplus), and in terms of benefits to international tourists (in the form of consumer surplus).

ESTIMATION OF THE CONTRIBUTION OF TOURISM TO NATIONAL GDP

Tourism benefits to the region were estimated in terms of direct value added to GDP, as a proxy for producers' surplus. This was estimated by spatially disaggregating national-level tourism data to determine the contribution of the study areas using a combination of national and sub-national tourism data and the density of geotagged photographs uploaded to the internet to map tourism value to ecosystems and other attractions (see Turpie *et al.,* 2017). Tourism's direct contribution to GDP was extracted for all the countries of the region from a consistent set of data sources (the World Travel and Tourism Council – WTTC; WTTC, 2020c, 2020a, 2020d, 2020b, 2020e). The proportion of tourism expenditure attributed to visiting attractions, as opposed to activities such as visiting family and friends, attending conferences or religious events, or receiving medical treatment, was then estimated for each category of tourists (holiday, visiting friends and relatives, business, and other) based on information collated from individual country tourism statistics reports and information related to tourist spending patterns (Tanzania NBS, 2017; Uganda Ministry of Tourism Wildlife & Antiquities, 2018; NISR, 2019; KNBS, 2020). Tourists whose main purpose is either visiting friends or family or business tend to spend much less of their money on visiting attractions than holiday/leisure tourists. These types of tourists do, however, make up a large proportion of the total tourism spending and so these contributions are not insignificant. Information on the breakdown of visitor activity and expenditure was not available for the six countries in this study, thus the following assumptions were made, based on data from South Africa (Table 3).

Table 3. Main purpose of visit and the percent of spending on visiting tourism attractions

MAIN PURPOSE	% OF SPENDING ON ATTRACTIONS
Holiday	100
Visiting Friends and Relatives	2
Business	4
Other	15

The spatial distribution of tourism value was mapped based on the density of geotagged photographs uploaded on the website flickr.com. These densities were obtained using the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) Recreation Model 3.5.0, which uses an application programming interface to get data from the website into a grid specified by the user (in this case a hexagon grid). Densities of geotagged photographs uploaded to platforms such as Flickr provide a means of mapping value to tourism attractions, rather than to the places where tourists spend their money (*e.g.*, at their accommodations), so is more accurate in assigning the tourism value to the actual attractions that caused the expenditure. The model calculates the average annual photo-user-days (PUDs) for each grid cell (5 km x 5 km) across the period 2005-2017. The model used the latitude/longitude data from photographs as well as the photographer's username and photo date to calculate PUDs. One PUD is one unique photographer who took at least one photo in a specific location on a single day.

Empirical evidence supports the use of this method. Wood *et al.* (2013) found that using geotagged photographs from Flickr served as a reliable proxy for empirical visitation rates, and can provide opportunities for understanding which elements of nature attract people to locations and whether changes in ecosystems will alter visitation rates (Wood *et al.*, 2013). Also using Flickr, Lee and Tsou (2018) studied geotagged photos from the Grand Canyon area and found that the frequency of uploaded monthly photos was similar to total tourist numbers counted at the site. The study also used spatiotemporal movement patterns of tourists in conjunction with the uploaded photos to show how this approach can be used for the improvement of national park facility management and regional tourism planning (Lee & Tsou, 2018). Barros, Moya-Gómez & Gutiérrez (2019) explored the potential of geotagged data to analyze visitors' behavior in a national park in Spain. Using geotagged photo data from Flickr and GPS tracks from a web platform called Wikiloc, the study determined the spatial distribution of visitors, the points of interest with the most visits, itinerary network, temporal distribution, and visitors' country of origin, which was used to improve national park facilities and management.

ESTIMATION OF CONSUMER SURPLUS ACCRUING TO THE REST OF THE WORLD

International visitors to the landscapes also derive benefits from their experience. This can be expressed in monetary terms as their willingness to pay (WTP) over and above what they had to pay to visit the study areas. Consumer surplus associated with tourism is typically measured using the Travel Cost Method. Consumer surplus is estimated using a demand curve that is derived from a model of visitation rates in relation to costs, and through imputation of hypothetical cost increases. This study drew on the results of Travel Cost Method studies that have been carried out in the Great East African Plains (Ntuli, unpublished) and in the Albertine Rift Valley landscapes (Hatfield & Malleret-King, 2007). The ratio of consumer surplus to expenditure to in-country expenditure was used to derive the estimated consumer surplus, based on the estimates made above. Direct value added by tourism was taken to be 45 percent of tourism expenditure (Hatfield & Malleret-King, 2007).

BIODIVERSITY EXISTENCE

There are no published estimates of global WTP for wildlife conservation, as motivated by existence values. To derive such an estimate is fraught with uncertainty, yet to ignore this value may also lead to a bias in policy and decision making. Thus, for this study, responding to the demand for such an estimate, an order-of-magnitude estimate was provided for regional and global WTP for the conservation of biodiversity in the study areas. Based on the metadata analysis of Jacobsen & Hanley (2009), and using

World Bank statistics on GDP per capita (US$) and population for 2018, global and regional annual willingness to pay for terrestrial biodiversity conservation were estimated. From these, ballpark estimates for global and regional WTP for conservation in the study areas were derived using simple assumptions. In the case of global values, it was assumed that WTP would be focused on the world's remaining natural areas that support biodiversity. The latter was assumed to be 30 percent of the continental area, since at least half of the worlds' natural habitats have already been lost, and since at least 30 percent of the earth's surface is required to be under conservation management in order to prevent major species loss (Hannah *et al*., 2020). It was then assumed that within these areas, WTP would be distributed roughly in proportion to mammal species richness (Figure 5; as a proxy for biodiversity, but also often the main focus of peoples' attention). This was a conservative assumption in that, in reality, there would be spatial discrimination within remaining wildlife habitats based on their current levels of biodiversity. For example, the global average mammal diversity at the 10km x 10km scale is 58 species, whereas the average for East Africa is 117 and the average for the four landscapes is 156. The estimated WTP of East Africans for biodiversity protection was assumed to be focused entirely on local region and was estimated as an average WTP using 30 percent of the region's extent.

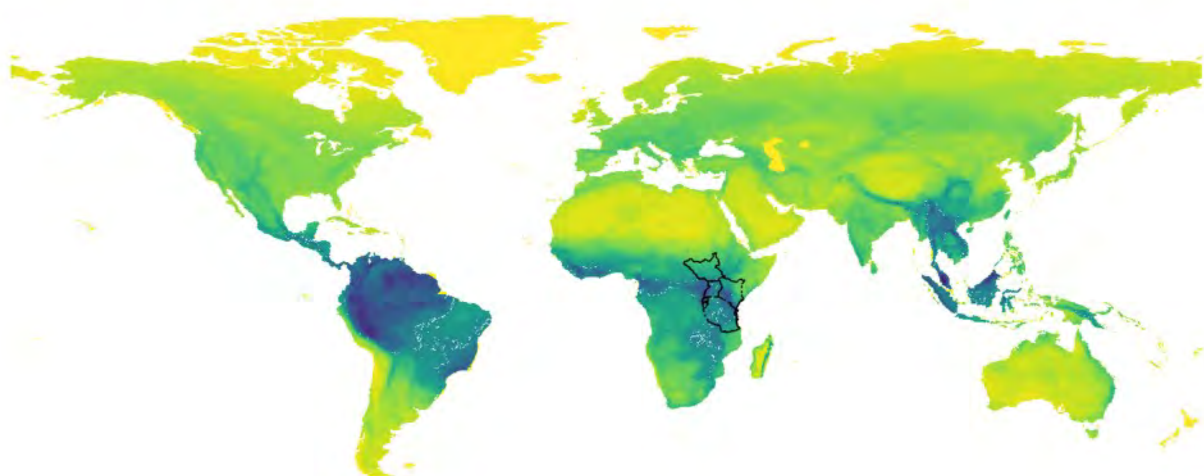


Figure 5. Map showing spatial variation in mammal species richness at global scale, and the location of the East African countries. Species richness ranges from lowest = yellow to highest = dark blue.

Source: Jenkins, Pimm & Joppa (2013)

FLOW REGULATION

Flow regulation was estimated using the InVEST 3.8.7 seasonal water yield model, which estimates quickflow (associated with rainfall events), local recharge, and baseflow, which helps to sustain river flows during the dry season. As inputs, we used the Copernicus Global Land Operations 2018 land cover map and the Hydrological data and maps based on Shuttle Elevation Derivatives at multiple Scales (HydroSHEDS) hydrologically conditioned global digital elevation model (Lehner, Verdin & Jarvis, 2008), which has a resolution of 90 m. The watershed and sub-watershed boundaries were derived from the global HydroBASINS dataset of watershed and sub-watershed boundaries (Lehner & Grill, 2013). In delineating regions for modeling, we ensured that all watersheds encompassed by the wildlife landscapes were included, as well as adjacent watersheds that might benefit from the hydrological services provided by the wildlife landscapes.

In addition to these base layers, the model required spatial inputs on monthly reference evapotranspiration (in mm), which was downloaded from the Consortium of International Agricultural Research Centers - Consortium for Spatial Information (CGIAR-CSI) database, and the water requirements of different vegetation/land cover types (measured by the plant evapotranspiration coefficient *Kc*, in agricultural contexts known as a crop factor). These were based on the few available studies from the region, along with comparable estimates from beyond East Africa (Descheemaeker *et al.*, 2011; Beatty *et al.*, 2018; Bagstad *et al.*, 2020). Using the model, we estimated the contribution of each pixel to baseflow, *i.e.*, water that reaches a stream relative to a denuded landscape.

The loss of this service would lead to a greater proportion of runoff occurring in the wet season, necessitating more storage to meet dry season water demands. Because a model of this nature cannot be used to estimate variation, the service was valued using the difference in annual contribution to baseflow as a proxy for the storage capacity saved. The replacement cost was estimated as the average reservoir construction cost per unit capacity. The physical modeling methods are described in more detail in Appendix 3.

EROSION CONTROL

To model the erosion control service, we used the InVEST sediment delivery ratio model, which estimated sediment retention and export through combining estimated soil loss with a connectivity index. The revised universal soil loss equation was used to calculate soil loss. This takes rainfall erosivity, soil erodibility, slope, land cover, and management into account. These were informed by global datasets, as well as a range of studies from the East African region and beyond (Angima *et al.,* 2003; Leh *et al.,* 2013; Hamel *et al.,* 2017; Bagstad *et al.,* 2020; Fenta *et al.,* 2020, Karamage *et al.,* 2017). This service was quantified as the amount of soil loss avoided for each one-hectare pixel, in m^3 per year. The avoided sedimentation of downstream rivers, reservoirs, and lakes was assumed to be fully demanded, and was valued using the replacement cost method. This assumed that in the absence of the service, sedimentation problems would need to be addressed through the construction of sediment check-dams, the cost of which was obtained from Mekonnen *et al.* (2015a). The physical modeling methods are described in more detail in Appendix 3.

WATER QUALITY AMELIORATION

Water quality amelioration was estimated using the InVEST 3.8.7 nutrient delivery ratio model, which uses a nutrient mass balance approach to quantify nutrient export to downstream aquatic systems. It combines measures of nutrient input across the landscape, with land-cover-specific retention and connectivity properties of pixels belonging to the same downstream flow path. In addition to the basic inputs for the seasonal water yield model described above, the model also required inputs on the nutrient (nitrogen and phosphorous) loads generated in kg/ha/year by each land cover class and the nutrient retention capacity, which were estimated based on the available literature (Freeman & Omiti, 2003; Ariga *et al.*, 2006; Namaazi, 2008; Leh *et al.*, 2013; Bagstad *et al.*, 2020), and a runoff proxy, for which we used mean annual precipitation data for 1970-2000 from World Clim (Fick & Hijmans, 2017). These methods are described in more detail in Appendix 3.

CARBON STORAGE

Carbon storage was valued using the most up to date global datasets available on carbon stocks, including above- and below-ground biomass and soil carbon (see FAO & ITPS, 2018; Spawn & Gibbs, 2020). The carbon retention value of these stocks was valued in terms of the avoided losses of economic output by the different countries in the landscape as well as the rest of the world, using recently published estimates of the global and disaggregated country-specific damage effects of climate change (see Ricke *et al.,* 2018, Table 58). These damage estimates are called the "social cost of carbon" and are expressed as U.S. dollars per ton of CO_2 emissions, in net present value terms. Thus, carbon stocks were first converted to the equivalent quantity of CO_2.

The social cost of carbon is estimated as a net present value of climate change impacts over the next 80 years of one additional ton of carbon emitted into the atmosphere today (Ricke *et al.*, 2018). The net present value is the discounted sum of the costs borne every year for 80 years. The effect of discounting is to downweight future values. A social rate of discount is used, which is a relatively low discount rate that does not downweight the future costs as much as a typical discount rate, *e.g.*, one used by an investor. This is because social planners take future generations into account. Even so, different countries use slightly different social discount rates based on factors such as income and life expectancy (Addicott *et al.,* 2020). For this study, we expressed the social costs of carbon in annual terms to be comparable with the other ecosystem services values.

Table 4. The country-specific social cost of carbon and social discount rates for each of the six countries used to estimate carbon storage value

COUNTRY	COUNTRY-SPECIFIC SOCIAL COST OF CARBON (US$ PER TON OF CO_2)	SOCIAL DISCOUNT RATE (%)
Kenya	0.61	6.52
Tanzania	1.04	4.86
Rwanda	0.16	5.41
Burundi	0.04	0.99
Uganda	0.84	4.04
South Sudan	0.53*	4.36*

*Note that the study did not have an estimate for South Sudan so an average across the other five countries was used to generate an estimate. Source: Ricke *et al.,* 2018 and Addicott *et al.,* 2020.

CROP POLLINATION

The contribution of natural habitats to crop production through pollination by wild pollinators was estimated on the basis of recently published empirical studies carried out in Tanzania and Kenya, which find that the proportion of natural habitat in the land surrounding cultivated fields has a positive effect on crop yields (Kasina *et al.*, 2009; Tibesigwa *et al.*, 2019). Tibesigwa *et al.* (2019) estimated the contribution of wild pollinators to crop revenues for smallholder crop farms in Tanzania, based on detailed plot-level panel data on production and surrounding land cover. A production function was created with the following inputs: plot-level agricultural data (Tanzania National Panel Survey); plot revenue from crop farming; wild pollination services (*i.e.*, percent share of natural habitat); production inputs; plot and household characteristics; and weather characteristics. To capture the relationship between crop productivity, foraging distance, and frequency of pollination, buffers of 100 m, 250 m, 500 m, 1,000 m, 2,000 m, and 3,000 m distance were placed around each of the farm plots, and the percentage share of natural habitat (forest) within each of the buffers was determined. For estimating the value of wild pollination services associated with each wildlife landscape and in determining the impacts of a loss of natural vegetation, we adopted a benefit transfer approach, drawing on this work. For the purposes of this study, we assumed that production in our study areas contained similar food crop species mixes, and similar pollinator ratios to those in the Tibesigwa *et al.* (2019) study.

Using the landcover map, all the pixels within and around the wildlife landscapes that were classed as cultivation were buffered by 1,000 m. The proportion of natural land cover in the buffer was then calculated and used to predict crop revenues by applying the panel regression model. This was compared to the result obtained when the proportion of natural land cover was set to zero. The difference was taken to be the contribution of the wild pollinators from the surrounding natural vegetation. This value was then mapped to the natural habitats in the buffer area.

FORAGE FOR LIVESTOCK PRODUCTION

Livestock farming is an important livelihood activity across most of the region, and East Africa has by far the largest livestock population in sub-Saharan Africa (Otte & Chilonda, 2002). Smallholder farmers dominate the agricultural landscape, with most smallholder farmers engaging in pastoral or agro-pastoral production systems. While most livestock are kept extensively, some areas support more intensive stall-fed production systems (*e.g.*, for dairy).

The ecosystem service described here is the contribution of natural systems to production. We valued the benefit in terms of contribution to GDP from extensive livestock production in the wildlife landscapes. This was quantified in physical terms as the amount of production (in terms of large stock units) supported, based on spatial data on livestock stocking rates. We used the latest (2010) density estimates for ruminant livestock (*i.e.*, cattle, sheep and goats) at 10 km resolution (Gridded Livestock of the World database, Gilbert *et al.,* 2018, Figure 6). For each landscape, the annual production in each country was valued using the average GVA per LSU,[2] which was derived from the national accounts. It should be noted that this map is generated from wildlife data provided by the different countries. This has the advantage of being consistently mapped across country boundaries, but the accuracy of the spatial patterns is unknown and may vary.

[2] LSUs provide a way of standardizing the quantification of different types of ruminant livestock. 1 LSU is ~ 1 cow or ~ 6 sheep/goats. More precisely, an LSU is equivalent of one head of cattle with a body weight of 450 kg and gaining 500 g per day (Meissner et al. 1983).

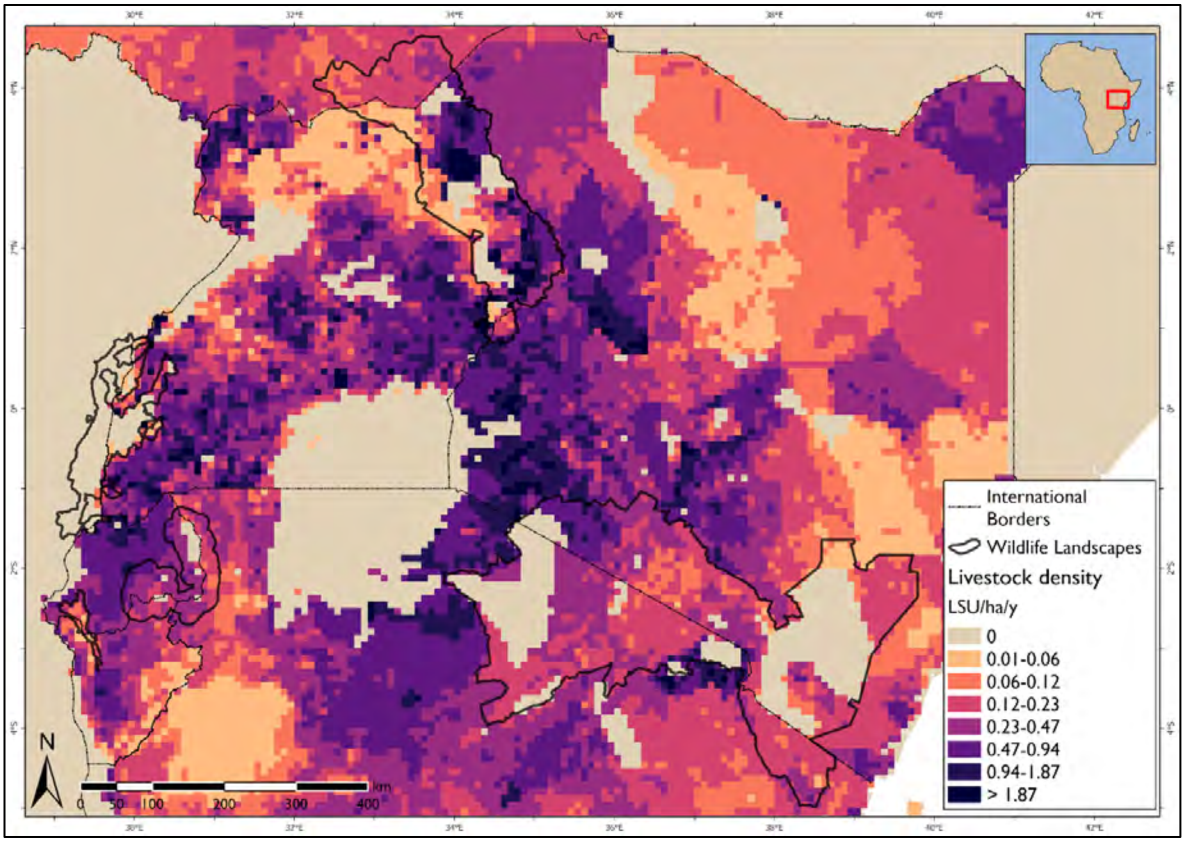


Figure 6. Density of livestock (LSU/ha/y) across the East Africa region

PROVISION OF HARVESTED WILD RESOURCES

Millions of people throughout East Africa harvest wild plant and animal resources for nutrition, health, energy, and raw materials, particularly where there are limited economic opportunities. The capacity of the landscape to supply different types of wild resources is related to vegetation type and condition, availability of water, and other factors. However, a number of other factors determine their use and value, and these vary in space and time. The accessibility of wild resources is determined by regulations such as land tenure and harvesting rights, social norms, and informal agreements, by geographic features such as topography and rivers, and by human-made features such as roads. The demand for wild resources is influenced by the socio-economic circumstances of households and the prices of alternatives. Due to data constraints, few, if any, studies have modeled these factors comprehensively. In this study, we have devised relatively simple estimates of capacity, accessibility, and demand. This study focuses on small-scale use of wild biomass resources. It does not include estimates of legal commercial harvesting of wild resources or illegal commercial-scale poaching of high value, endangered species.

The valuation of provisioning services involved estimating and mapping the stocks of each resource group based on habitat characteristics, estimating and mapping the demand for the resources based on household characteristics, livelihoods, and population density, and then estimating use based on the patterns of demand and availability of resources. The habitat characteristics were based on a map of habitats derived from land cover, vegetation type and biomass data, and stock densities were estimated based on the literature, with assumptions interpolated among habitat types as necessary. Household characteristics were obtained from census data and reports and assigned spatially based on maps of urban vs. rural areas. Data on livelihoods and use of resources were then assigned to households based on their location and characteristics. The use of resources was estimated with the spatial model of Turpie *et al.* (2020). Detailed data sources and assumptions are given in Appendix 4.

EXPECTATIONS UNDER A BUSINESS-AS-USUAL SCENARIO

OVERVIEW

Having provided a description of the baseline situation as in 2018, the next task was to consider what changes might be expected over the next three decades (to 2050), under a "business-as-usual" (BAU) scenario. BAU means no change in policy or management. This was informed by a review of information on past trends and/or available projections of population, land use, climate, and biodiversity. We also took into account the reality of the COVID-19 pandemic that took hold in East Africa during 2020. The aim of this part of the study was to provide a qualitative assessment of the direction and potential magnitude of the effects of a variety of pressures on wildlife habitats and wildlife, and the implications for the supply of ecosystem services and for society. The conceptual framework used in the assessment of the BAU is shown in Figure 7. By definition, the governance/policy driver is assumed to be constant. The methods for assessing the remaining drivers are outlined below.

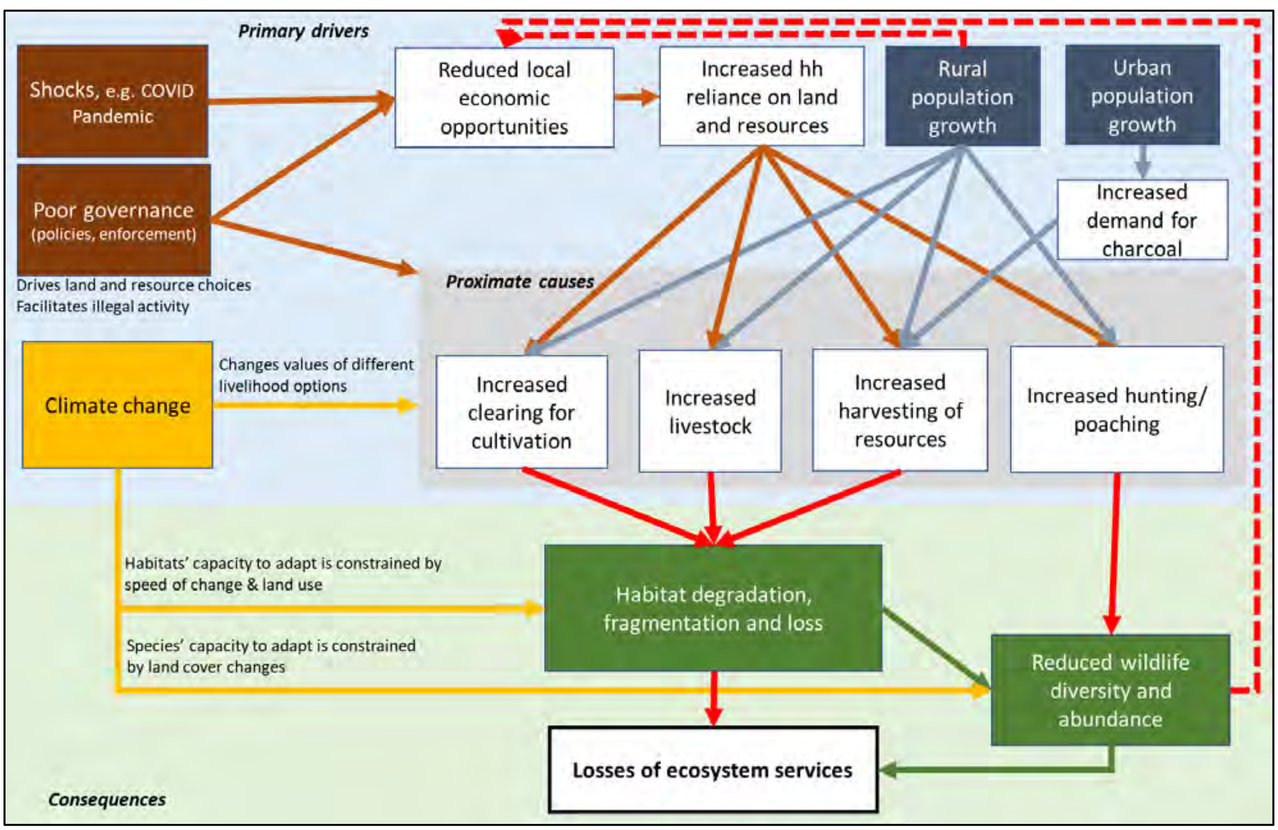


Figure 7. Conceptual model of the potential impact pathways leading to loss of ecosystem services over time under a BAU scenario

POPULATION CHANGES

Population projections up to 2050 were obtained from the United Nations (UN) (2019) and Worldometer (2020). These projections take rates of urbanization into account. Thus, the rates of change in the rural and urban populations are quite different. Both have an impact on the wildlife landscapes however, as urban population growth drives up demand for resources such as charcoal.

LAND COVER CHANGES

To assess historical trajectories and predict future habitat loss and degradation, we drew upon quantitative estimates of land cover change, as well as qualitative accounts of land cover change trends. Numerous studies have quantified changes in land cover in various parts of East Africa. However, these have invariably been done at different spatial extents and scales to our study landscapes, ranging from fine-scale studies at the sub-catchment level to national-level estimates of land cover change.

To acquire estimates of future land cover change for our study regions specifically, we explored the possibility of modelling future land cover changes using available software. However, the accuracy of available land cover proved to be a significant obstacle. While the 100 m Copernicus land cover (Buchhorn *et al.*, 2020) used extensively in the rest of the study appears to be reasonably accurate for the region, the earliest version of this product only goes back to 2015. A comparison with this 2015 layer suggested that this time period was too short from which to acquire meaningful land cover change trends. While other East African land cover change studies conducted at smaller scales generally create their own fine-resolution land cover datasets, the size of the study landscapes, time constraints, and the impossibility of ground truthing precluded this.

The only other land cover datasets that encompass the whole region and cover a sufficiently large time period were the coarser resolution 300 m ESA CCI land cover (European Space Agency, 2018) and 500 m MODIS land cover (Friedl & Sulla-Menashe, 2019). However, comparison of these datasets with satellite imagery of our study regions indicated some accuracy concerns, particularly with the MODIS data. Due to this and the coarse resolution of the datasets, it was felt that they would not be appropriate for land cover change modelling, especially as any initial inaccuracy would be compounded during the modelling process. However, historical changes in the area of cultivation and settlements derived from the 300 m ESA CCI land cover (European Space Agency, 2018) are reported. To improve the accuracy of these estimates, a correction factor was applied to the 300 m land cover. This was obtained by comparing the cultivated and built-up areas in the 300 m land cover and 100 m land cover for the year 2018, and proportionally altering the area estimates derived from the 300 m land cover accordingly. Nevertheless, these figures are likely to have a fair degree of inaccuracy and should be taken as only a rough guide to trends occurring in the landscape.

Given these limitations in regional land cover data, we used alternative data sources to acquire trends in land cover change for the landscapes, and make projections on the basis of these for modelling changes under a BAU scenario. For the Albertine Rift Forests and Rweru-Mugesera-Akagera Wetlands, Global Forest Change 2000-2019 data was used as the primary indicator of land cover change. The methods used by this dataset are described in Hansen *et al.* (2013). As natural habitats in the Albertine Rift and natural terrestrial habitats surrounding the Rweru-Mugesera-Akagera generally have moderate to high tree cover, the use of deforestation data provided a usable measure of land cover change in this case. In the Great East African Plains landscape, the trend in cultivated area between 2015 and 2018 in Copernicus 100 m land cover did align with changes reported in the literature. Hence, it was possible to simply use the land cover data to predict future expansion of cultivation under a BAU scenario in this landscape. The Northern Savannas landscape proved to be the most challenging for predicting land cover change, as tree cover is generally too low for use of the Global Forest Change dataset (Hansen *et al.*, 2013) as an indicator of land cover change, while all regional land cover products explored reported declining or static cultivated areas for the region. In contrast, increased cultivation was widely reported by the literature. This included studies that developed their own land cover data for portions of the landscape, some of which reported extreme increases in agriculture of three times (Nakalembe, Dempewolf & Justice, 2017) and even ten times (Egeru *et al.*, 2014c) in just over a decade. In the end, the more moderate rate of expansion reported by Osaliya, Wasonga & Kironchi (2019) was used, as this appeared to provide a more realistic intermediate between the extremes of no change and very rapid change derived from other sources.

CLIMATE PROJECTIONS

Baseline climate data for the study area were obtained from WorldClim historical data, which provides average monthly climate data for mean and minimum temperature and for precipitation for the period 1960–1990 at a spatial resolution of about 1 km^2 (a resolution grid of 30 arc-sec; Hijmans *et al.*, 2005). Seven global climate models sourced from the Coupled Model Inter-Comparison Project (CMIP5) and considered to best simulate the climate in Africa (Conservation International, 2018), were used to project the climate for the period 2040-2060, under the representative concentration pathway (RCP) 8.5 scenario, at the same spatial resolution as the baseline data. The RCP 8.5 emissions scenario is considered the worst-case scenario (business-as-usual) by the IPCC (Sanford *et al.*, 2014). Given that the climate change we experience over the next few decades will primarily be caused by past emissions (Glick, Stein & Edelson, 2011), the RCP 8.5 scenario was used in this study. This is also the scenario that observed changes in climate appear to be tracking most closely. Furthermore, because near-term projections of climate change scenarios tend to have a higher degree of certainty, a relatively short time horizon of 2050 was used.

It should be noted that while rainfall and temperature projections are available, there are no reliable projections on the frequency or severity of extreme events such as droughts and floods. In East Africa, these are strongly linked to the El Niño-Southern Oscillation (ENSO), through its influence on the inter-tropical convergence zone (Hulme *et al.*, 2001; Ummenhofer, Kulüke & Tierney, 2018), but this is complicated by the influence of the Indian Ocean sea surface temperature. These influences are not well simulated in global climate models (Hulme *et al.*, 2001), but there is consensus that a warmer climate will lead to more ENSO events in total with the potential for more extreme events in both directions—flooding and droughts.

CLIMATE CHANGE IMPACTS ON SUITABILITY FOR CROPS

The potential impact of a change in climate on the suitability of crop production for key crops across the different landscapes was evaluated using the UN Food and Agriculture Organization's (FAO's) EcoCrop analytical tool and the FAO database for climate and maturation input variable thresholds (FAO, 2010). The EcoCrop model can be used to estimate climate suitability scores for a certain area, which, when run using both historical and future climate data, can be used to estimate the potential change in suitable area for production of the specific crop in question. For each crop in each study area, the model was run using climate data at a resolution of about 1 km^2 to yield a suitability score for each 1 km^2 cell. The FAO EcoCrop analytical tool is a mechanistic model that has been used to predict suitability of various crops under different climatic conditions. It uses temperature and rainfall ranges, as well as time to maturity for a crop, as input variables to determine the climate thresholds of a crop and then produces a suitability score as output (Ramirez-Villegas, Jarvis & Läderach, 2013). Suitability scores range from 0.0 (not suitable) to 1.0 (highly suitable). The results were mapped for visual interpretation of potential impacts of climate change on agricultural livelihoods and rates of land cover change.

CLIMATE CHANGE IMPACTS ON HABITATS AND SPECIES

Climate change is a major threat to biodiversity, affecting both individual species and overall ecosystem functioning (Scheffers *et al.*, 2016). To survive a shift in suitable climate, species may need to either adapt to their changed environment or relocate to more suitable areas in order to avoid extinction (Moritz & Agudo, 2013). However, opportunities to move may be restricted by anthropogenic or natural barriers such as cultivated land, mountain ranges, or water bodies. Shifts in species ranges as a result of climate change have already been observed (Parmesan & Yohe, 2003; Chen *et al.*, 2011; Mason *et al.*, 2015), and further shifts in suitable range are predicted in response to future climate changes (Araujo *et al.*, 2005). Shifts in plant species can entirely change the ecosystem structure and functioning of areas, altering existing habitats completely (Gonzalez *et al.*, 2010). These ecological shifts will exacerbate the challenge of conserving these species and ecosystems in the Great East African Plains landscape.

The vulnerability and expected response of species and habitats to climate change has received considerable attention in the scientific community, with correlative or "climate envelope" modelling having received the most attention (Dawson, 2011). This approach maps where species are likely to migrate to as they attempt to stay within their specific climatic niche (Hijmans & Graham, 2006). These species distribution models (SDMs) are based on the climate niches of their historical distributions. Hannah *et al.* (2020) have estimated changes in distribution by 2070 of about 20,000 animal species globally. This study has made use of their output maps for more than 1,000 species of amphibians, reptiles, mammals, and birds found in the East African region. It should be noted that the SDMs are climate niche models and do not consider habitat, natural or anthropogenic barriers to range changes, hydrology, or inherent potential for adaptation to climate change. The results of the regional level analysis are shown in Appendix 5, and their implications are discussed for each of the landscapes.

THE INFLUENCE OF THE COVID-19 PANDEMIC

The reaction to the COVID-19 pandemic has left an impact that will last for years and possibly even decades to come. The primary mitigation measure for reducing the spread of the virus has been to limit human movement, which has in turn resulted in a decline in economic activity. The United Nations (2020) estimates a 1.1 percent real growth rate in Africa's GDP under its most optimistic scenario, down from 3.1 percent previously. The most pessimistic scenario forecasts a 2.7 percent real decline in GDP. This has major implications for employment. In this region, the key impacts are likely to be on livelihoods centered around tourism, a critical sector of the economy. The decline in foreign tourists, in particular, due to lockdowns on travel worldwide is projected to affect protected areas and the wildlife within them. For example, roughly 68 percent of Tanzania's protected areas rely on income generated from trophy hunting. In Kenya, nearly 50 percent of the Kenya Wildlife Service's (KWS's) annual budget is derived from tourism (Lindsey *et al.*, 2020). The shutting down of international travel to these destinations is likely to have severe consequences at least in the short-term. In the Great Eastern Plains region, the Masai Mara community conservancies may lose roughly US$7.5 million per year without normal tourism numbers. This may, according to Lindsey *et al.* (2020), lead to a lower incentive to manage land sustainably. Aside from the decline in local benefits from nature-based tourism, the pandemic has resulted in a general increase and food insecurity and poverty due to losses of jobs and income and reduced local conservation benefits. These drive up reliance on natural resources, leading to increases in wildlife and non-wildlife crime, which threatens biodiversity both inside and outside protected areas (Lindsey *et al.*, 2020).

Reduced tourism has already had knock on effects for effective management of protected areas with some areas temporarily closing, reduced park staff on duty, and reduction in routine maintenance (IUCN ESARO, 2020). This effect will be compounded by government funds being transferred to other sectors such as bolstering healthcare facilities. In Uganda, for example, key informants note that the pandemic has affected budgetary allocations for conservation, which has severely affected operations in national parks. In Rwanda, tourist numbers reportedly fell by 75 percent in 2019, with an 85 percent reduction in tourism revenues. At the same time, they noted funds for conservation have been diverted to healthcare due to the pandemic emergency. The Rwandan key informant expected recovery of tourism to begin from 2023. Rwandan key informants also noted that the loss of tourism revenue led the government to halt revenue sharing agreements with communities living around national parks, prompting concern that this could lead people to start poaching and extracting resources again. Key informants from the Great East African Plains landscape similarly noted that COVID-19 has drastically reduced the flow of nature-based tourism benefits to local communities. This has reportedly resulted in the KWS suspending revenue sharing, while conservancies have been unable to pay dividends to community members due to the absence of tourists (Ondicho, 2021).

Any increased losses in biodiversity resulting from the impacts of the COVID-19 pandemic may affect future visitor numbers to wildlife areas, which could drive a positive feedback loop of continued degradation without new, resilient conservation models. For the assessment of the four study areas, it is assumed that the pandemic will have increased the level of dependence on land and resources relative to what would have been expected under normal trajectories of changes in population and income levels.

OVERALL IMPACTS ON ECOSYSTEM SERVICES AND WELLBEING

Descriptions of the expected nature and severity of outcomes for ecosystems and wildlife and for the capacity of wildlife landscapes to deliver benefits under a BAU scenario note the uncertainties involved in projecting forward to 2050.

Projections of demand for natural resources were made on the basis of current human population trends and the associated changes in urban land cover, while future livestock numbers were predicted from current trends in livestock numbers.

Changes in nature-based tourism were estimated based on current trends of tourism growth over the period 1995-2018 (actual) and 2018-2027 (forecast) from the WTTC data gateway. Based on estimates of decreased tourism during 2020 as a result of COVID-19, the value of tourism was assumed to decline by 70 percent in 2020, recovering slowly over a five-year period from 2021 to 2026. This was based on a review of similar global impacts such as the Ebola virus disease in West Africa in 2014/15 and the global recession in 2008. Projected changes up to 2050 for each landscape were made using data on changing wildlife numbers, information on changes in tourist numbers and rates of congestion in protected areas, and impacts of rising populations on protected area encroachment, livestock numbers, and changing cultivation.

Changes in hydrological services (flow regulation, soil erosion control, water quality amelioration) were based on projections of land cover change by 2050 for the different landscapes, and comparison of this with an extreme (unrealistic) scenario where all arable areas are converted to cultivation. This extreme scenario represents the full loss of hydrological services that would occur if all natural habitat were lost. The predicted change in cultivated area by 2050 under a BAU scenario was then calculated as a proportion of the change in cultivated area in the extreme scenario. This proportion was applied to the estimates of hydrological service loss in the extreme scenario, to obtain the estimated change in hydrological services by 2050 under a BAU scenario.

Changes to carbon storage (both above and below ground) were calculated using a combination of projected deforestation trends derived from the Global Forest Change data (Hansen *et al.*, 2013) and population trends (as a proxy for urban/built-up land cover change). The calculations of changes to carbon used the tons of carbon per land cover type in 2018 and adjusted these based on the projected changes in land cover relevant to each site. Losses in carbon storage mean increased emissions, which incurs costs globally in terms of global climate change.

VALIDATION AND STAKEHOLDER INPUTS

The study was presented to stakeholders at a series of online workshops throughout the region from early- to mid-2021. Based on their feedback, semi-structured interviews were then carried out with a number of key informants relating to each different sector in each country of each region to obtain further information to fill the gaps that had been identified. The inputs made were qualitative in nature and were used to amend or add details to the descriptions and interpretations of data in the study. The information received was then integrated into the final report.

REGIONAL CONTEXT OF THE STUDY AREAS

OVERVIEW

East Africa is a region of exceptional climatic, topographic, and ecological diversity. While the precise extent of the East African region is variably defined, this study is confined to the EAC countries of Burundi, Kenya, Rwanda, South Sudan, Tanzania, and Uganda, with a focus on selected transboundary landscapes within these nations. Vegetation across these countries ranges from forest in the mountainous areas of the Albertine Rift and the humid Indian Ocean coast, to arid shrubland across large areas of northern Kenya, with a range of woodland and savanna types in between. The region has the longest history of human habitation of anywhere in the world, given its central role in the evolutionary history of modern humans. East Africa is renowned for its wildlife, most notably the exceptional herbivore populations in the grassland plains of Tanzania and Kenya, as well as the lesser-known large wildlife populations of South Sudan's grasslands and savannas. Meanwhile, the forests of Uganda, Rwanda, and Burundi are best known for their populations of mountain gorilla (Uganda and Rwanda only) and other large primates.

EXTENT, TOPOGRAPHY, AND DRAINAGE

As can be seen in the map of topography and drainage (Figure 8), much of East Africa is relatively high-lying, due to a series of plateaus of varying elevations (Maxon, 2009). The African Rift Valley is a major topographic feature, with its western and eastern arms forming two pronounced troughs. The western arm, or Albertine Rift, extends across the western regions of Uganda, Rwanda, Burundi, and Tanzania and neighboring areas of the DRC. Strong relief features are associated with both arms of the Rift Valley, with steep escarpments and a series of lakes in the deep valley floors. From the Virunga volcanoes northwards, drainage in the Albertine Rift occurs in a northerly direction into the Nile Basin. This includes Lakes Albert, Edward, and George. South of the Virunga volcanoes, the Albertine Rift drains into the Congo Basin, which includes Lakes Kivu and Tanganyika.

The eastern arm of the African Rift Valley, also known as the Gregory Rift, extends across western Kenya and central Tanzania. The Gregory Rift lacks an outflow, hence drainage occurs internally only. The Turkana Basin, centered on Lake Turkana, is a drainage basin of the system, encompassing northwest Kenya and southern Ethiopia, as well as small areas of eastern Uganda and southeast South Sudan. Smaller internal basins drain into the more southerly lakes of the Gregory Rift, including Lakes Baringo, Naivasha, and Natron.

Land between the two arms of the Rift Valley mostly consists of a large central plateau, which underlies much of Uganda, southwest Kenya, central and western Tanzania, and eastern Rwanda and Burundi. This area mostly drains into Lake Victoria, the largest lake in Africa. The Lake Victoria Basin encompasses a large part of the region, draining west to southwest Kenya, northwest Tanzania, southern Uganda, and most of Rwanda and northern Burundi. Lake Victoria is also the source of the White Nile, which exits from the northern part of the lake.

East of the Gregory Rift Valley, most of northern and eastern Kenya as well as eastern Tanzania are underlain by a lower-lying eastern plateau. The various river basins in this region drain eastwards into the Indian Ocean.

Other prominent relief features of the broader East African region include several large isolated volcanic mountains, Mount Kilimanjaro being the most famous example. The Eastern Arc Mountains form a prominent mountain range in eastern and central Tanzania and southeastern Kenya.

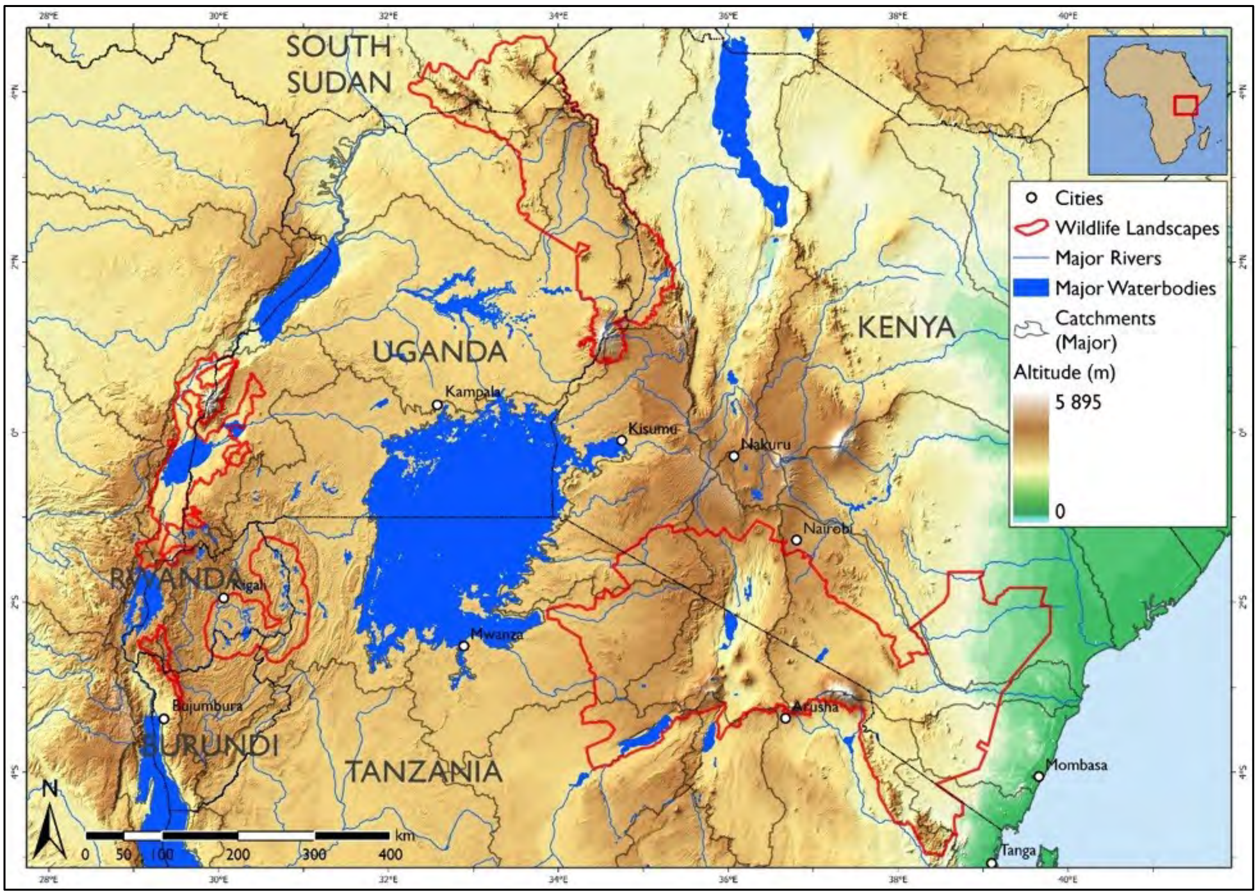


Figure 8. Topography and drainage of East Africa

Data sources: Rivers and waterbodies, "African Water Bodies"; "Africa Rivers" 2014; Catchment areas, Lehner & Grill, 2013; Elevation, https://elevation.arcgis.com/arcgis/services/NED30m/ImageServe)

CLIMATE

East Africa is a climatically diverse region. High rainfall zones (over 1,000 mm mean annual precipitation) are mostly found in the western part of the region (Figure 9). This includes most of Uganda, Rwanda, and Burundi, as well as southwestern Kenya and northwest Tanzania. Southern Tanzania and the Indian Ocean coastal belt also receive high rainfall. Away from these zones, localized areas of high rainfall are associated with relief features. For example, certain areas on the slopes of Mounts Kilimanjaro and Kenya receive 2,000 mm mean annual precipitation or more.

Away from these wetter regions, large areas of East Africa are anomalously dry compared to other equatorial regions (Camberlin, 2018). The northern and eastern parts of Kenya are particularly dry, with mean annual precipitation below 500 mm over most of this region. A relatively dry corridor with <700 mm annual rainfall also extends over southeast Kenya, and northeast and central Tanzania. These drier regions of East Africa also experience highly variable rainfall, both in terms of the overall amount and timing of precipitation. This adds to the challenges of local people in these already semi-arid to arid regions.

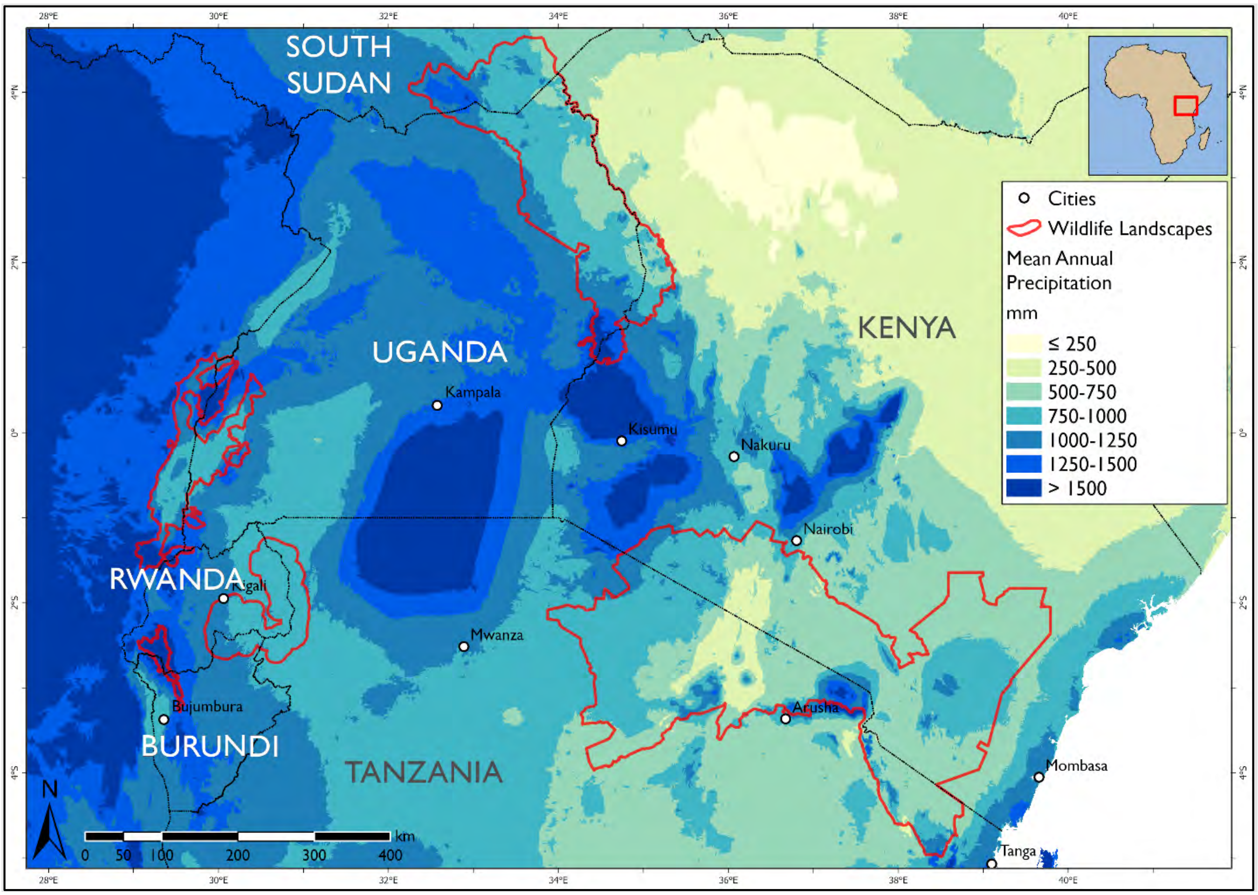


Figure 9. Mean annual precipitation across East Africa

Data source: Fick & Hijmans 2017

The seasonality of precipitation also varies across East Africa. Most of the region has a double peak rainfall regime, where rainfall peaks twice during the transitional (spring/autumn) seasons. These peaks occur around March-May and September-November. The drier areas of Kenya and Tanzania experience a prolonged dry season outside of these peak seasons (Maxon, 2009). Areas around the Lake Victoria Basin experience a third rainfall period around June-August, thus reducing the length of the dry season (Marchant *et al.*, 2018). The pattern changes moving away from the equator. Further north, South Sudan and northern Uganda have a wet season during the northern hemisphere summer, with a dry season during the northern hemisphere winter. Southern Tanzania exhibits the opposite trend, with peak rainfall during the southern hemisphere summer and a dry season over the southern hemisphere winter.

Temperature in the region is closely connected to altitude, while cloudiness also has some influence (Figure 10). Due to relatively high elevations, much of equatorial East Africa is fairly cool for its latitude. Mean annual temperatures below 20°C occur over much of Burundi, Rwanda, and southwestern Uganda, as well as in the higher lying areas of Kenya and Tanzania. On the upper slopes of the region's highest mountains, mean temperatures drop below 0°C. Due to a combination of reduced cloudiness and lower elevation, the highest temperatures are experienced in the dry regions of northern and eastern Kenya, where the annual mean ranges between 25 and 30°C (Daron, 2014). The Indian Ocean coastal region is also warm, often with mean annual temperatures in excess of 25°C. Aside from the southern highlands of Tanzania, seasonal variation in temperature is small, as is typical for equatorial regions (Camberlin, 2018).

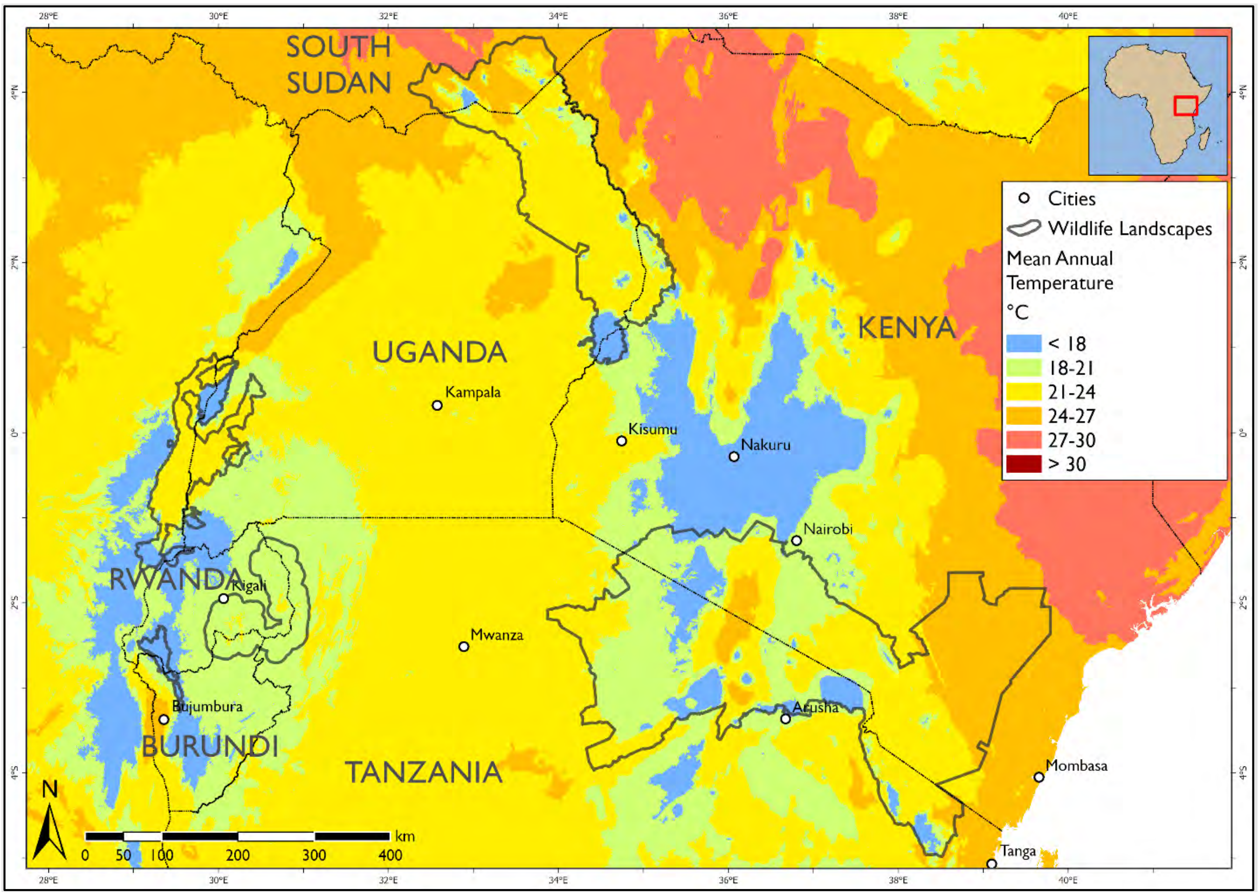


Figure 10. Mean annual temperatures across East Africa

Data source: Fick & Hijmans, 2017

NATURAL VEGETATION, LAND USE, AND LAND COVER

The extraordinary range of natural habitats across East Africa reflects the climatic and topographical diversity of the region (Figure 11, Figure 12). A belt of coastal rainforest occurs in the warm, wet Indian coastal regions (Marchant *et al.*, 2018). Moving inland, *Acacia-Commiphora* bushland and wooded grassland is dominant over much of northern, eastern, and southern Kenya and northern Tanzania. Patches of Afromontane forest also occur in mountainous parts of this generally dry region, due to localized increases in rainfall. Ericaceous and afro-alpine vegetation are found on the upper slopes of Mounts Kenya and Kilimanjaro. Further south, *Acacia-Commiphora* bushland gives way to miombo woodland, which dominates central and southern Tanzania, while semi-desert shrubland and desert occur in the driest parts of northern Kenya.

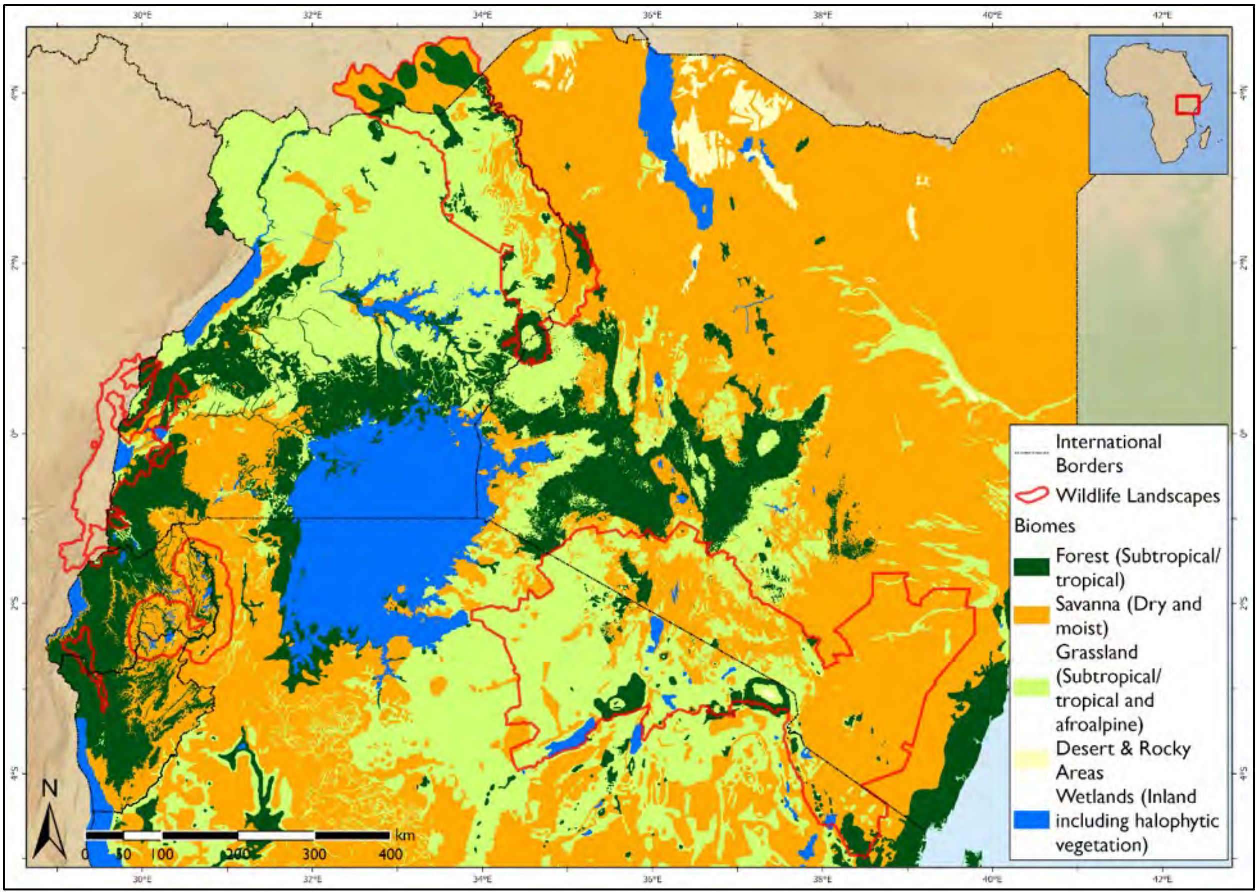


Figure 11. Biomes of East Africa

Data source: Potential natural vegetation map for eastern and southern Africa (van Breugel *et al.,* 2015) grouped according to broad IUCN habitats classification scheme

With increased rainfall over the central highlands and Lake Victoria Basin regions of Kenya, *Acacia-Commiphora* bushland gives way to evergreen bushland, forest, and moist Combretum wooded grassland. However, due to more favorable climatic conditions for agriculture, most natural habitats here have been converted to cultivation. The Lake Victoria Basin region of northwest Tanzania is similarly heavily cultivated. Moving further west into Uganda, rainforest was the dominant natural vegetation type across much of the wetter northern and western regions of Lake Victoria. However, most of this forest had already been lost by the end of the 20th century due to extensive habitat conversion to agriculture (Struhsaker, 1987). Numerous wetlands also occur in this region, although many have similarly been degraded or converted.

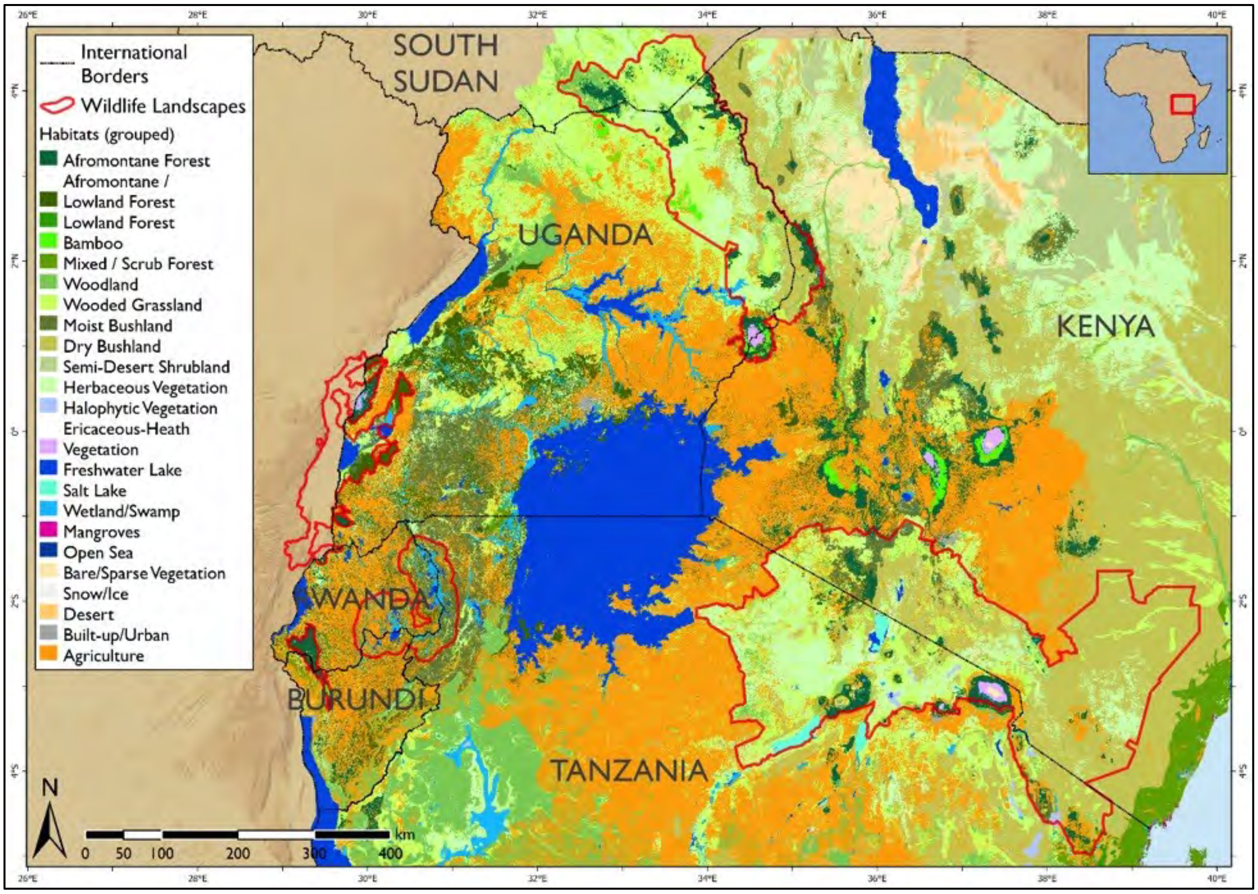


Figure 12. Derived habitat map of the East Africa study area.

The map shows the 72 habitats combined into 23 more broad groupings for brevity and legend display purposes. The habitat types were derived from a combination of vegetation map, land cover, and NDVI change spatial data. See section on delineation of ecosystem types for a full description of how habitat types were mapped using these data layers, and Appendix 2 for the full list of habitat types.

To the north of Lake Victoria, woodland and wooded grasslands replace forest as the dominant natural vegetation types over northern Uganda and southern South Sudan. However, patches of forest do occur at higher elevations where average rainfall is increased. In the driest regions of northeast Uganda, *Acacia-Commiphora* bushland also occurs. To the west of Lake Victoria, evergreen bushland is the dominant natural vegetation type over much of southern Uganda, eastern Rwanda, northeast Burundi, and the northwest corner of Tanzania. Several large wetlands also occur in this region, with the Akagera wetlands along the Rwanda/Tanzania border being the most extensive.

Rainfall increases again further west into the Albertine Rift region. As a result, Afromontane forest replaces evergreen bushland as the dominant natural vegetation type over western Rwanda and Burundi, as well as southwest Uganda. Again, high population densities across this region mean very little forest remains outside of protected areas. From southern Burundi, miombo woodland replaces Afromontane forest, and remains the dominant vegetation type throughout western Tanzania. Similarly, forest is replaced by woodland and wooded grassland in the drier northwest parts of Uganda.

DEMOGRAPHIC AND SOCIOECONOMIC STATUS

East Africa is one of the poorest regions in the world. However, most countries in the region have experienced rapid economic growth in recent years. This is particularly the case for Rwanda, Tanzania, Kenya, and Uganda, all of which had annual GDP growth rates of around 5 percent or more from 2014 to 2018 (UNECA, 2020; Table 5). Despite economic growth, East African countries rank poorly on the Human Development Index (HDI), particularly Burundi and South Sudan, both of which are in the bottom five countries of the world (UNDP, 2019). All but Kenya have an HDI below the average for sub-Saharan Africa.[3] Thus, the region continues to face steep developmental challenges. At the same time, rapid population growth is occurring across the region, with East African countries among the fastest growing globally, adding to the pressure on natural resources and economies. Population growth is among the highest in the world, having increased from 35 million in 1960 to 185 million in 2018, a five-fold increase in 60 years. Rwanda and Burundi are the two most densely populated countries in mainland Africa (UN, 2019; Figure 13). Although rapid urbanization is occurring, the regional population remains predominantly rural. With 87 percent of its population still rural, Burundi is the least urbanized country in the world (UN, 2018). Proportionally, Tanzania is the most urbanized country in the region, although even here, 66 percent of the population is rural.

Table 5. Summary of demographic and socioeconomic statistics for each of the six East African countries

COUNTRY	GDP (2018, US$ BN)	GDP GROWTH RATE (%)	POPULATION (MILLIONS)	POP DENSITY (PEOPLE PER KM^2)	POP GROWTH RATE (%)	% OF POP RURAL	% LIVING IN POVERTY
Burundi	3.04	1.6	11.2	435.2	3.2	87	71.8
Kenya	87.91	6.3	51.4	90.3	2.3	73	36.8
Rwanda	9.51	8.6	12.3	498.7	2.6	83	55.5
South Sudan	12.00	-10.8	11.0	18.1	0.6	80	42.7
Tanzania	58.00	5.4	56.3	63.6	3.0	66	49.1
Uganda	27.46	6.2	42.7	213.1	3.7	76	41.7

Based on estimates from 2015-2017 unless otherwise indicated. South Sudan GDP and GDP growth estimate from 2015. Burundi poverty estimate from 2013. South Sudan poverty estimate from 2009 (pre-independence). All other data is for 2018.

There is evidence for progress toward improving social wellbeing. For example, life expectancy in the region increased at an unprecedented rate from 2008-2018, ranging from a four-year increase in South Sudan to eight-year increases in Tanzania, Rwanda, and Uganda, reflecting increased immunization and improved access to healthcare (UNECA, 2020). On a less positive note, food insecurity is a persistent issue, with worsening malnutrition in Uganda and Kenya from 2004 to 2018. Drought and other extreme weather conditions have contributed to these declines. Additionally, rapid population growth

[3] HDI 2018: Kenya 0.579, Rwanda 0.536, Tanzania and Uganda 0.528, Burundi 0.423, South Sudan 0.413 (sub-Saharan African average – 0.541; world average - 0.731).

has led to reductions in per capita food production across most East African countries (UNECA, 2015). Progress toward poverty reduction remains mixed, with significant poverty reductions in countries like Uganda, but stagnant or even increasing poverty levels in Kenya (UNECA, 2015). The region is also experiencing rapid growth in numbers of refugees and internally displaced persons, presenting a serious barrier to economic growth and social development (UNECA, 2020). With 1.2 million refugees, Uganda hosts the third-largest refugee population globally.

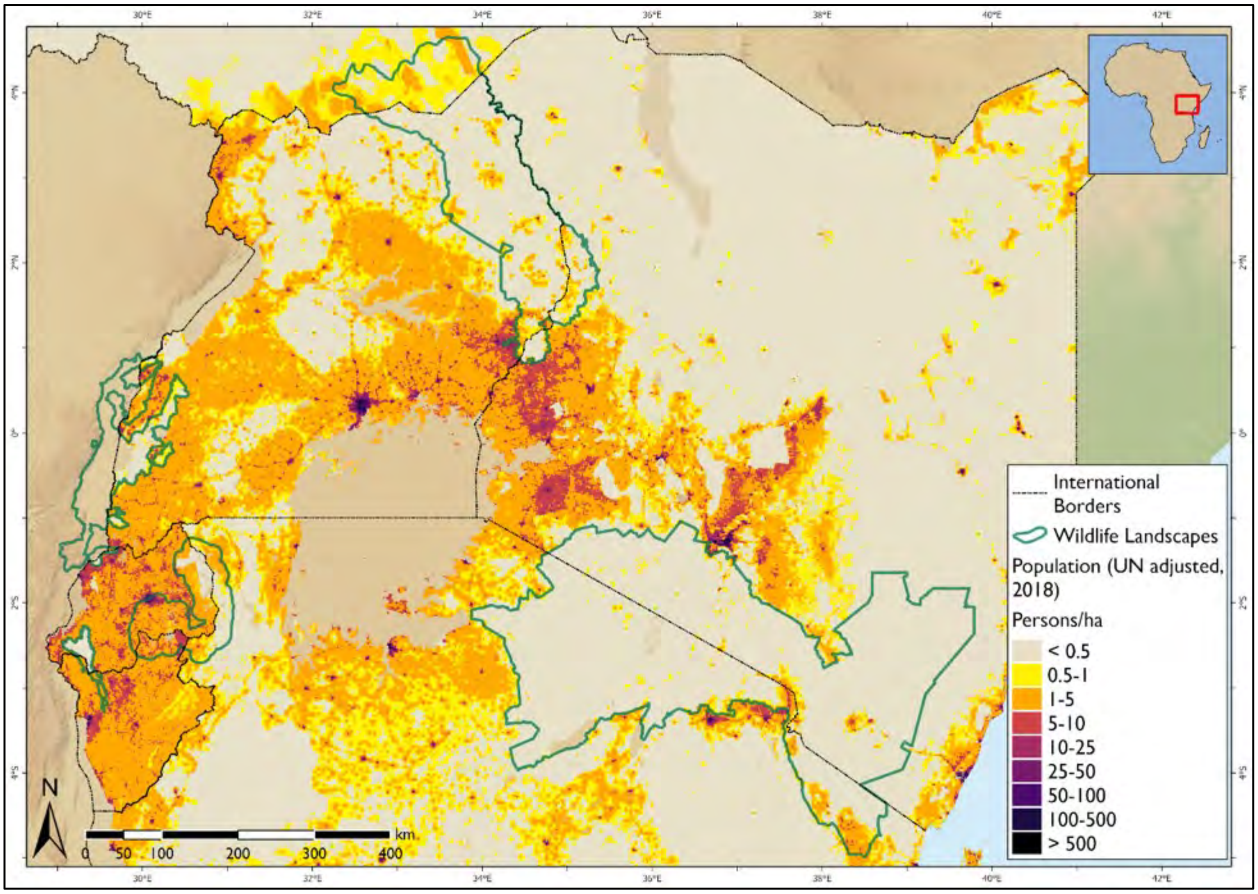


Figure 13. Population density (people per ha) across East Africa

Data source: WorldPop (www.worldpop.org - School of Geography and Environmental Science, Department of Geography and Geosciences & Departement de Geographie, Universite de Namur) and Center for International Earth Science Information Network (CIESIN), 2018).

PROJECTED CHANGES IN CLIMATE AND POPULATION

All parts of the planet are likely to experience changes to their current climatic conditions over the next few decades due to natural and anthropogenic climate change. The predicted changes can be modelled using one or more of several sophisticated models that are used for different climate and greenhouse gas emissions scenarios. Figure 14 shows the potential changes between historic and future mean annual temperature changes in the region as well as the mean temperature anomaly using an averaged value from the CMIP5 models. Overall, the trend is that the region becomes 1-3°C hotter, with the western regions (Albertine Rift Forests and Rweru-Mugesera-Akagera Wetlands) and the northern regions

(Northern Savannas) warming the most, and the coastal areas with the lowest mean change. The implications of this for people and wildlife are discussed in the final sections of the following chapters.

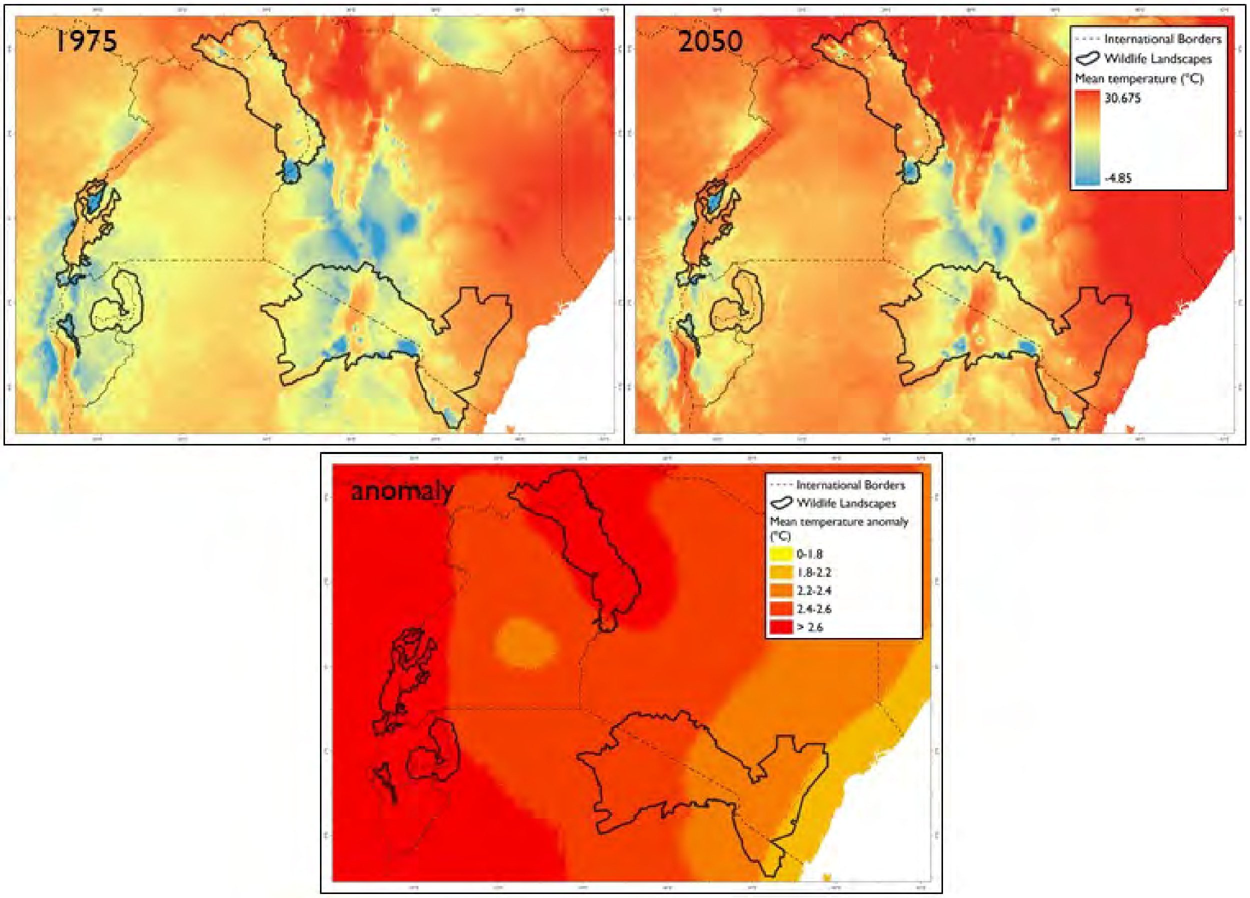


Figure 14. Historic (1960 – 1990) and projected (2040 – 2060) mean annual temperature across the study area and surrounds, as well as the mean temperature anomaly.

Source: Based on data from WorldClim Version2 and CMIP5.Predicted

The study area is also projected to continue experiencing considerable population growth, with most of this in urban areas. By 2050, the region is expected to have 379 million people, with urban populations almost 3.8 times the number in 2018, and with rural populations having increased by 40 percent overall (Figure 15, Table 6). Rural populations are projected to begin stabilizing around 2040 in South Sudan, Uganda, and Kenya. Uganda and Tanzania's urban population is expected to overtake the number of people living in rural areas before 2050, while the same is expected in South Sudan and Kenya. The implications are discussed in the following chapters.

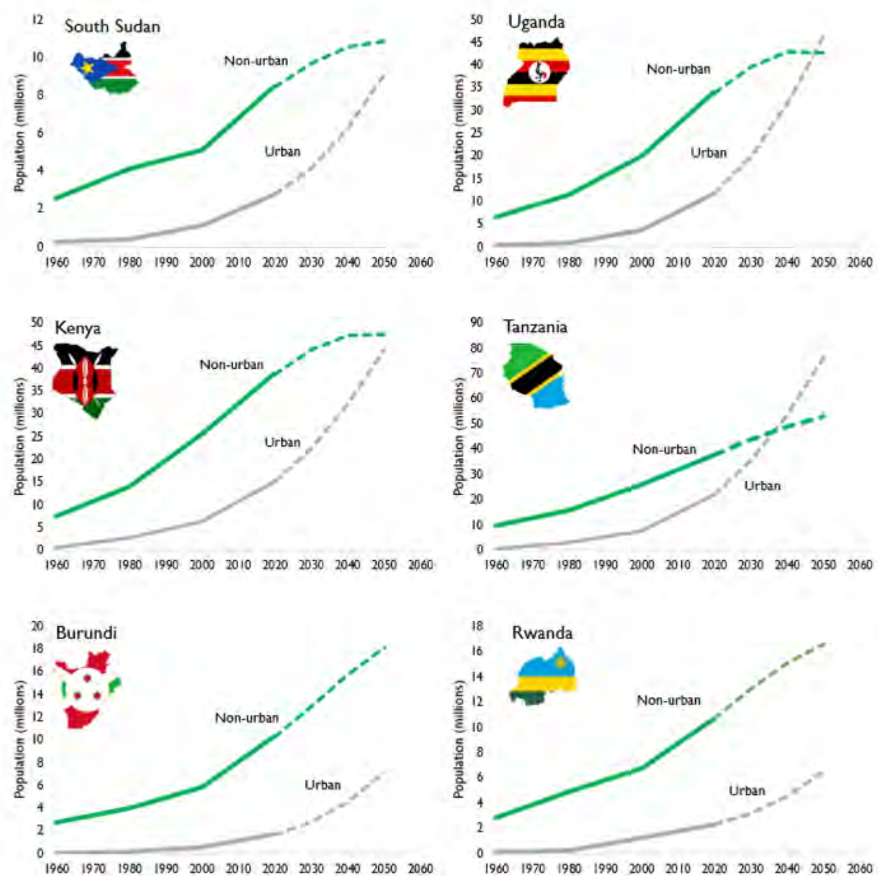


Figure 15. Population changes and projections for the six East African countries.

Data source: UN (2019) and Worldometer (2020).

Table 6. Projected population growth to 2050

COUNTRY	PROJECTED 2050 POPULATION (MILLIONS)	2050 URBAN POPULATION AS MULTIPLE OF 2018	2050 RURAL POPULATION AS MULTIPLE OF 2018
Kenya	92	3.21	1.26
Tanzania	129	3.84	1.45
Burundi	25	4.91	1.87
Uganda	89	4.43	1.33
Rwanda	23	3.01	1.63
South Sudan	20	3.60	1.28

Source: Based on U.N. (2019) and Worldometer (2020).

THE GREAT EAST AFRICAN PLAINS

FEATURES AND LOCAL CONTEXT

WILDLIFE AND WILDLIFE HABITAT

The Great East African Plains support the largest wildlife populations on Earth. This area encompasses some of the most famous protected areas in Africa (Figure 16), including the Mara, Amboseli, and Tsavo in Kenya, and the Serengeti and Ngorongoro Crater in Tanzania, which before the COVID-19 pandemic drew in more than a million visitors a year.

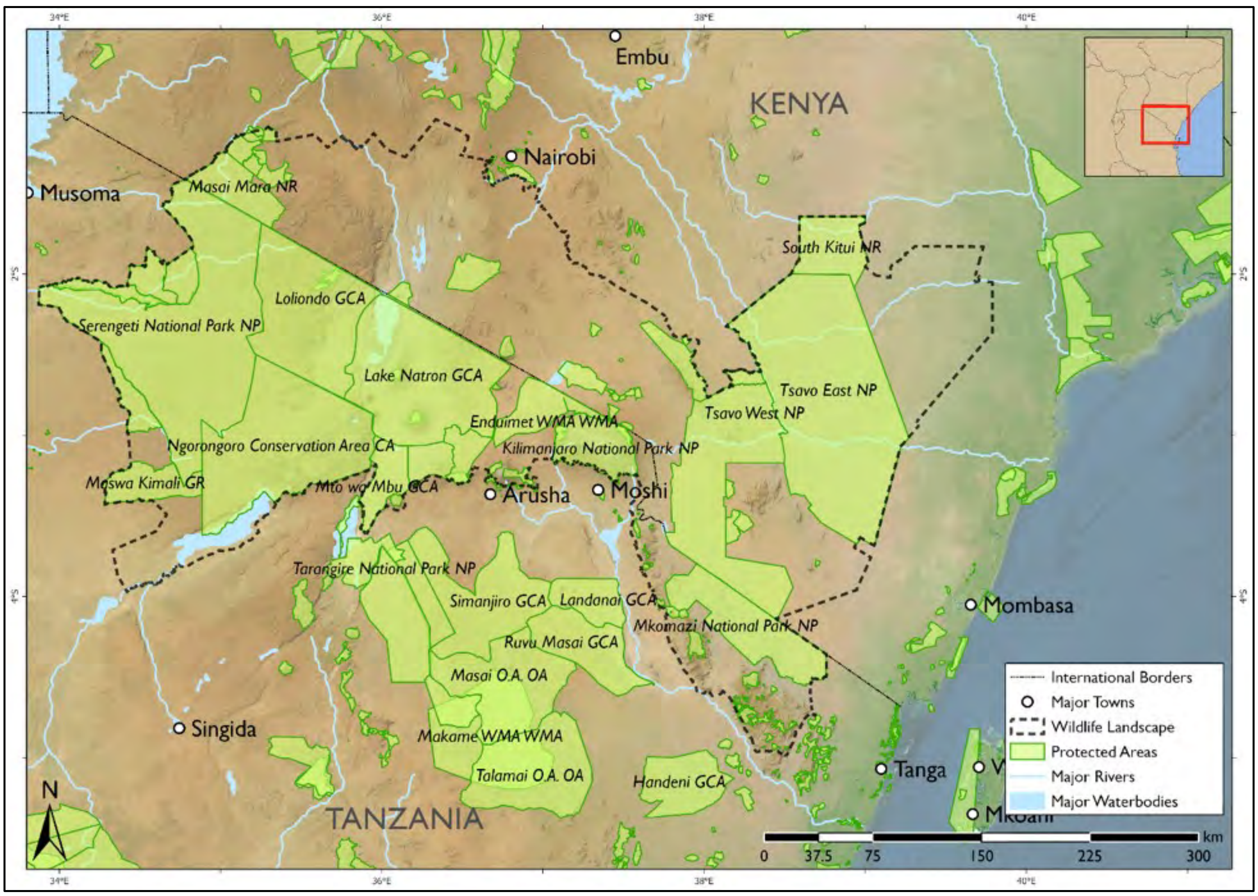


Figure 16. The Great East African Plains wildlife landscape and associated protected areas

The terrain rises gradually moving westward from the Indian Ocean toward the Great East African Plains wildlife landscape. Multiple large mountains produced by volcanic activity rise out of the plateaus of this region. These include Mount Kilimanjaro, the highest mountain in Africa at 5,895 m, as well as Mount Meru. These mountains are the headwaters of the Pangani River, the major catchment for the far northeast regions of Tanzania. Tanzania's Pare and Usambara Mountains are located to the southeast of Kilimanjaro, and also contribute water to the Pangani Basin. Further north, the Chyulu and Taita Hills are other prominent relief features, and are a source for several springs and rivers in an otherwise dry area of Kenya (Kenya Water Towers Agency, 2018). To the west of Mounts Kilimanjaro and Meru, the Eastern Plains study area is bisected by the eastern arm of the Great Rift Valley, also known as the Gregory Rift. Topography is varied in this region, which includes a deep trough along the Kenya/Tanzania border, and the prominent Ol Doinyo Lengai volcano and Ngorongoro Crater Highlands to the south of this. West of the eastern Rift Valley, slopes become gentler upon reaching the relatively flat plains of the Serengeti-Mara region.

The Great East African Plains region is largely semi-arid to arid, with vegetation ranging from the productive, largely treeless short-grass associations of the Serengeti Plains to wooded grassland, bushland, thicket, Acacia woodland, and montane forest (Figure 17; Homewood, Trench & Brockington, 2012). The Mara and northern Serengeti regions in the northwest of the study area are relatively wetter, with vegetation a mix of wooded grassland, evergreen bushland/thicket, and montane forest (Homewood *et al.*, 2012). Elsewhere, isolated areas of montane forest occur in higher lying areas where rainfall is heavier, including parts of the Ngorongoro Crater Highlands, the slopes of Mounts Kilimanjaro and Meru, and the Taita and Chyulu Hills. Ericaceous and Afroalpine vegetation occur at higher elevations on Mounts Kilimanjaro and Meru. Away from mountainous areas, *Acacia-Commiphora* bushland dominates large areas of the drier rangelands to the east of the Serengeti-Mara region, interspersed with grassland.

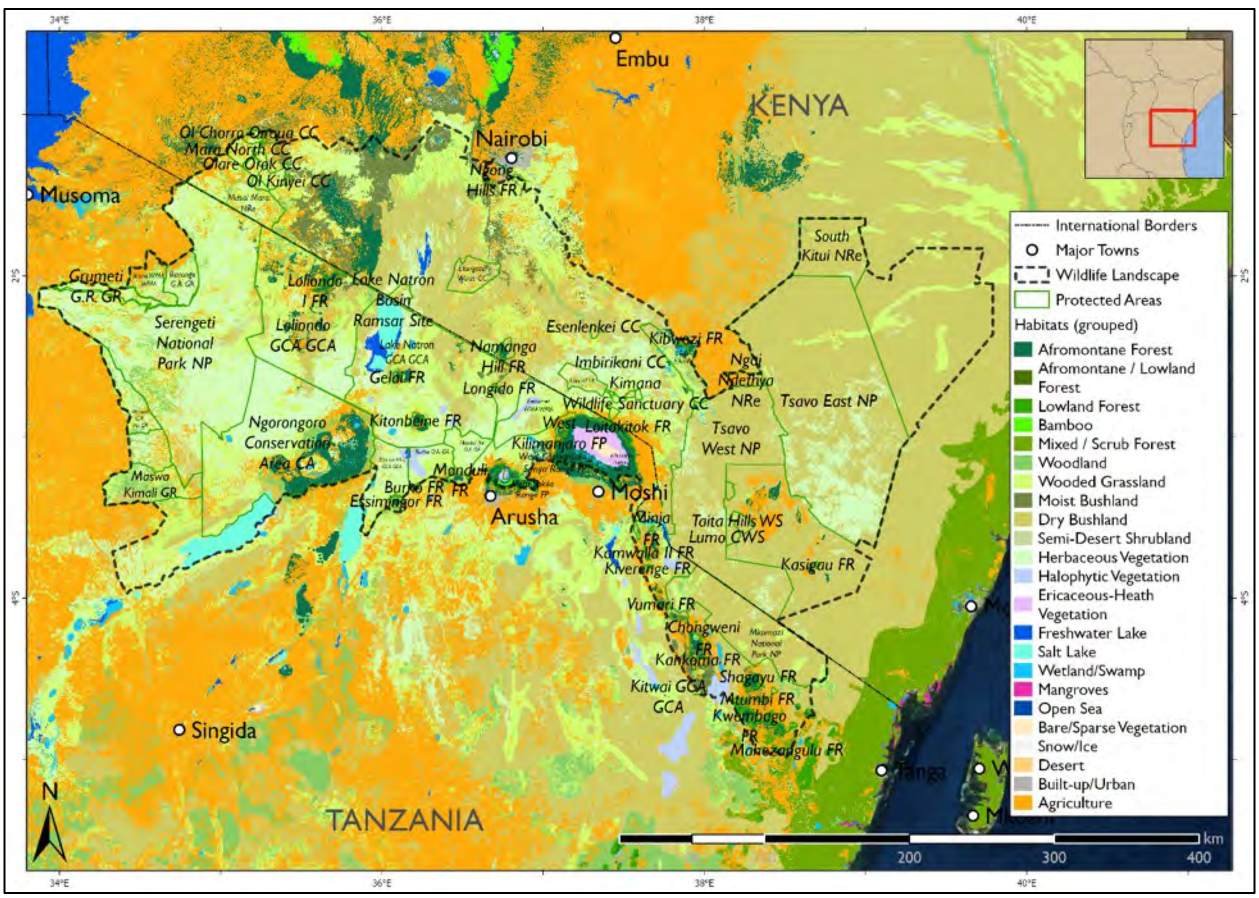


Figure 17. Land cover and natural habitat types of the Great East African Plains wildlife landscape

People in this area have historically depended on a mixture of livestock herding, farming, hunting, and gathering. Specialized pastoralism has been on the decline for some decades, with the majority of households diversifying toward agro-pastoralism or non-farm activities (Homewood, Kristjanson & Trench, 2009). This has been accompanied by a significant shift in land tenure, as formerly communal rangelands have been subdivided into private holdings, converted to commercial agriculture, or set aside for conservation. Traditionally, both people and wildlife moved across the landscapes seasonally to optimize the quality and quantity of grazing (Homewood *et al.*, 2009). However, the shift toward land privatization and fencing of formerly communal rangelands has constrained movement and access to key resources for people, livestock, and wildlife. It has also driven some households to adopt cultivation as a means of staking claim to land in response to privatization, as has occurred in the Mara region (Thompson *et al.*, 2009). At the same time, rapid population growth has occurred across the region, driving further expansion of cultivation and increased livestock numbers. Land privatization and population growth have together eroded traditional rotational grazing systems in the region, resulting in increased sedentarization and more intense use of grazing lands (Bartels, 2016). Degradation and loss of important wildlife habitats in the region has resulted from intensified livestock impacts and from the expansion of both subsistence and commercial agriculture.

Significant declines in wildlife populations in the Mara appear to have resulted from these anthropogenic impacts, with 50-80 percent declines in numbers of most species since the 1970s, including a 90 percent reduction in the resident wildebeest population (Ogutu *et al.*, 2011, 2016; Damania *et al.*, 2019). Wildlife populations in Tanzania are somewhat more stable, however declines of a number of ungulate species have also occurred here (Stoner *et al.*, 2007). For example, buffalo numbers in the Serengeti underwent a sharp decline in the late 1970s and 1980s due to increased poaching in a time of economic turmoil, though numbers have recovered in recent years (Metzger *et al.*, 2010; Mduma *et al.*, 2014). As it encompasses some of Africa's premier safari destinations, the study region generates substantial tourism revenue, which makes an important contribution to the economies and foreign currency reserves of Kenya and Tanzania. Declines in wildlife populations could thus have serious economic consequences for these countries.

PROTECTED AREAS AND NATURAL RESOURCE MANAGEMENT

A significant portion of this wildlife landscape is under some form of protection, forming extensive landscapes for the movement of wildlife. However, the level of protection and management differs across each type of protected area. A variety of state protected areas are found in the Tanzanian portion of the landscape. The national parks are managed by Tanzania National Parks Authority (TANAPA). These areas permit only non-consumptive use of wildlife and do not allow cultivation or grazing of livestock. The Ngorongoro Conservation Area is managed by a standalone entity called the Ngorongoro Conservation Area Authority (NCAA). Livestock grazing is permitted here but cultivation is not. All other protected areas, including game reserves, game controlled areas (GCAs), and wildlife management areas (WMAs) are managed by the Tanzania Wildlife Management Authority (TAWA). Both non-consumptive and consumptive (hunting) use of wildlife takes place in these areas. As with national parks, cultivation and grazing of livestock are not permitted within game reserves. GCAs are areas declared for conservation of wildlife outside village land that seek to avoid activities damaging to wildlife. However, unlike in game reserves, livestock grazing and cultivation are permitted. WMAs are community-based conservation areas formed on village land and where villagers set aside land purposely for sustainable conservation and utilization of wildlife resources. These areas have lower levels of protection when compared to national parks. Open areas represent the lowest level of protection, as they have no formal conservation status. District councils are important partners in conservation in Tanzania, particularly in WMAs. Hunting is not permitted in Tanzanian national parks but is allowed in GCAs, WMAs, and open areas.

In Kenya, the national parks and sanctuaries are managed by Kenya Wildlife Services (KWS). National reserves are managed by county governments, including the Masai Mara National Reserve, which is managed by Narok County Government. These areas do not permit cultivation or grazing of livestock. In addition to state protected areas, the Kenyan portion of the wildlife landscape has numerous community- and privately owned conservancies that provide additional habitat for wildlife and can play an important role in providing migratory corridors and buffer areas around state protected areas. Typically, conservancies are divided into three zones: 1) a core wildlife conservation zone that is strictly protected; 2) a zone for human settlement; and 3) a prescribed livestock grazing area (Oduor, 2020). Collection of firewood, medicinal plants, and honey for subsistence use may be permitted outside the core zone. In contrast to Tanzania, trophy hunting is not allowed in Kenya in any protected area category. The dominant protected areas within the Great East African Plains landscape and their defining features are described in Table 7.

Table 7. The dominant protected areas within the Great East African Plains landscape and their defining features

PROTECTED AREA	SIZE (HA)	MANAGEMENT	DEFINING FEATURES
Arusha National Park	11,136	Established 1960, managed by TANAPA	Dominated by Mount Meru, the fifth-highest free-standing mountain in Africa. Has a variety of habitats and is home to buffalo, warthog, zebra, waterbuck, giraffe, and colobus monkeys. Important birdwatching area.
Kilimanjaro National Park	184,791	Established 1973, managed by TANAPA	The highest mountain in Africa, and world's tallest free-standing volcano. Montane forest, glaciers, and waterfalls. The park is home to 140 species of mammals including 7 primate species and 179 bird species. Hiking and climbing to summit, paragliding, crater camping, and mountain biking are some of the activities.
Mkomazi National Park	341,187	Upgraded to a NP in 2006, managed by TANAPA	Shares a border with Tsavo in Kenya, important migratory route for herds of elephant, oryx, and zebra during the wet season. Diversity of wildlife including black rhino, gerenuk, wild dogs, lions, and cheetah. Also 450 bird species.
Ngorongoro Conservation Area	825,014	Established 1959, managed by NCAA	World Heritage Site. Ngorongoro Crater is the largest unflooded and unbroken caldera in the world. Multiple land use area where pastoralism, conservation of natural resources, and tourism are the three main components given equal consideration in policy decisions. Olduvai Gorge is an important archaeological site. Flamingos are common to the lakes of the NCA.
Serengeti National Park	1,303,605	Established 1951, managed by TANAPA	World Biosphere Reserve and World Heritage Site. Home to the great migration and to the world's largest populations of wildebeest, zebra, eland, lion, cheetah, hyena, and gazelles. Spectacular landscapes and unique bird assemblages.
Amboseli National Park	38,741	Established 1974, managed by KWS	One of the best places in Africa to view large herds of elephants. A variety of habitats such as Lake Amboseli, wetlands with sulfur springs, savannas, and woodland areas support numerous wildlife species including leopard, cheetah, wild dogs, buffalo, elephant, giraffe, zebra, and lion. Prolific birdlife with more than 600 species.

PROTECTED AREA	SIZE (HA)	MANAGEMENT	DEFINING FEATURES
Masai Mara National Reserve	151,785	Established 1974, managed by Narok County Government	The most famous and popular safari destination in Kenya. The rolling hills, open grassland, and acacia woodland support an incredible profusion and variety of wildlife including lion, cheetah, and leopard, which can be seen up close throughout the year. The great migration takes place between August and October when more than 1 million wildebeest move into Kenya from Tanzania.
Tsavo East National Park	1,339,708	Established 1948, managed by KWS	Tsavo East forms part of the largest protected area in Kenya. Notable features include the Galana River and the 300-km long Yatta Plateau. Home to elephant, rhino, buffalo, lion, leopard, hippo, crocodile, waterbuck, lesser kudu, and gerenuk. A total of 500 bird species have been recorded.
Tsavo West National Park	703,024	Established 1948, managed by KWS	Tsavo West is a rugged wilderness with open grasslands, scrublands, Acacia woodlands, belts of riverine vegetation, and rocky ridges. Wildlife include elephant, rhino, hippo, lion, cheetah, leopard, and buffalo. There are diverse plant and bird species too, including the threatened corncrake and near threatened basra reed warbler.
Chyulu Hills National Park	74,100	Established 1983, managed by KWS	Verdant rolling hills and spectacular views characterize the scenery of this park, with diverse habitats ranging from grassland to montane forest. Large mammals include buffalo, elephant, bushbuck, and giraffe, along with a wealth of reptiles and insects.

Source: TANAPA, TAWA, KWS, NCAA

Conservancies in Kenya and WMAs in Tanzania represent efforts to involve communities in management of wildlife and natural resources. Large wildlife populations are found in these community-managed areas in both countries. In addition to offering game viewing opportunities, these conservancies and WMAs are used as reserve areas for livestock grazing during the dry season. Numerous conservancies are located within the Kenyan portion of the landscape, including Eselenkeni, Kitenden, Kimana, Tawi, Osupuko and several others in the Amboseli-Kilimanjaro region, Shompole and Elengata Wuas conservancies in the Lake Magadi region, and several conservancies adjoining the northern border of the Maasai Mara National Reserve (Mara North, Olare Orok, and Naboisho Conservancies, to name just a few). These areas include both community and group conservancies. Community conservancies are established on communal land and allow local inhabitants to retain the right to use the land in accordance with activities permitted under the conservancy's constitution (Oduor, 2020). In contrast, group conservancies involve land owners leasing out parcels of land to a registered company, which manages wildlife and generates revenues from ecotourism. These revenues are used to fund conservation activities and to compensate land-owners (Oduor, 2020).

The main WMAs in the Tanzanian portion of the landscape are Ikona, which is located along the northwest border of the Serengeti National Park, and Enduimet, which is located between Lake Natron and Kilimanjaro National Park. Communities in these areas benefit from the sharing of revenues generated by WMAs, as 30 percent of revenues go to district councils and 20 percent go directly to member villages. Community informants also noted that WMAs have benefitted them through improving and securing pasture for livestock while enhancing protection of wildlife. Some community members also reported they benefit through employment as guides and village game scouts. On the negative side, some community informants said that the revenues generated from these areas were lower than they had hoped, reducing their community development potential. Additionally, a tourism operator noted that having too many livestock in WMAs compromised their potential for nature-based tourism investments, as the pristineness of the landscape is reportedly compromised. This highlights the need to carefully balance conservation and livestock in WMAs. It was reported that communities tend to value grazing land more than the conservation of wildlife habitats in WMAs, which could represent a threat to the future success of these areas in attracting wildlife tourism.

In addition to partnering with communities in conservation of the landscape, government conservation agencies are assisted by a number of NGOs and private sector conservation partners. These help to supplement conservation efforts by government agencies, whose work is inevitably limited by financial constraints. There is also significant transboundary collaboration between Kenya and Tanzania in management of the wildlife landscape. According to key informants, KWS and TANAPA conduct joint patrols in the Tsavo-Mkomazi, Amboseli-West Kilimanjaro, and Serengeti-Mara ecosystems to combat poaching. Notably, these joint patrols are not anchored in any formal policies or laws but are dependent on mutual respect and agreements between the relevant enforcement agencies. The joint patrols have reportedly achieved laudable success in controlling poaching. In addition to joint patrols, there is also a cross-border rhino security program and transboundary initiatives for management of the Mara River Basin. The latter was formalized by the signing of a memorandum of understanding (MoU) by the two countries in 2015. Kenya and Tanzania also signed a MoU to support joint management of the transboundary waters of Lakes Chale and Jipe and the Umba River ecosystems in 2015. The two countries are also currently developing the Southern Kenya Northern Tanzania program, a transboundary conservation effort to assist the two governments implement their respective wildlife corridor and human development initiatives, which should further improve coordination of cross-border conservation activities.

Future opportunities to invest in and expand conservation in the landscape, as identified by key informants, include the expansion of community conservancies in Kenya and WMAs in Tanzania. These represent opportunities to expand conservation of wildlife habitats in the region in a manner that is potentially mutually beneficial to both local communities and wildlife. There are also opportunities for more private land owners to set aside their land for conservation through contractual agreements, in return for lease payments and other benefits from nature-based tourism. However, the attractiveness of these conservation options will be heavily affected by the level to which tourism recovers from the impacts of the COVID-19 pandemic, as this has greatly curtailed opportunities to generate revenue from nature-based tourism in the region.

PEOPLE AND LIVELIHOODS

The people that live in the areas surrounding the plains wildlife landscape are diverse in their livelihood activities. Rainfall varies considerably across this region, and as a result so does the type and intensity of agricultural activities undertaken. In the central and eastern parts of the study region in Kenya, the majority of the population is semi-nomadic Maasai and livestock production is the main source of income for households. However, an increasing shift to cultivation is occurring, with agriculture often viewed as more profitable than pastoralism (Okello, 2005). This has put increased strain on land and water resources in the region. For example, cultivation and irrigation have caused substantial losses of swamps in the Amboseli/Chyulu Hills area, compromising these important habitats for wildlife and people (Okello & Kioko, 2011). Crops in this region are mostly cultivated on a small scale and generally produce low yields ("SSEBop Evapotranspiration Products," n.d.). However, large-scale cultivation also exists, notably the large-scale maize and wheat farming in close proximity to the Mara (Gicheru *et al.*, 2012). In the western parts of the landscape, rainfall is far more erratic. People farm crops for sale and own consumption and also keep cattle, sheep, and goats. In Tanzania, Maasai pastoralists are found across the central areas and rely solely on livestock as their main source of income. To the west, in the areas that lie adjacent to the Serengeti National Park, most households grow crops for own consumption as well as for sale at market. Cotton is farmed and is a source of employment for many in this region. Around the Kilimanjaro-Meru area, livelihoods are more diverse. People grow a variety of crops such as maize, coffee, and plantains. Tourism in this area provides employment opportunities for some. In this study region, there is a total population of just under 9 million people, with more than two-thirds of these people living in Tanzania (Table 8). The population in Kenya is almost entirely rural (97 percent), whereas in Tanzania there are some built-up areas resulting in a population that is 88 percent rural.

Table 8. Population statistics for the Great East African Plains study region

COUNTRY	TOTAL POPULATION	NUMBER OF RURAL HOUSEHOLDS	AVERAGE HOUSEHOLD SIZE	% RURAL
Kenya	2,297,121	522,513	4.2	97
Tanzania	6,700,721	1,141,476	5.1	88
Total for study region	8,997,842	1,663,988	4.7	93

ECOSYSTEM SERVICES

NATURE-BASED TOURISM

Tourism is estimated to account for 8.2 percent of Kenya's total economy and 10.7 percent of Tanzania's (WTTC, 2020b, 2020d). In both countries, tourism is a leading sector in terms of foreign exchange earnings and contributes significantly (8.5 percent and 11.1 percent, respectively) to total employment, especially in rural areas where economic opportunities are limited. It provides more than 1.5 million jobs (Okello, 2014; Tanzania MFP, 2016; Damania *et al.*, 2019; WTTC, 2020b, 2020d). Nature-based tourism, in particular wildlife viewing, is the backbone of the tourism industry in both countries. Indeed, wildlife contributes significantly to the economies of Kenya and Tanzania, and wildlife tourism is seen as key contributor to socio-economic development and a valuable source of income. Tourism numbers and expenditure have been increasing steadily over the last 20 years (Valle & Yobesia, 2009; Okello, 2014; Price, 2017). A recent report by Sanghi *et al.* (2017) on the economic assessment of Kenya's tourism industry found that safari tourism generated greater economic growth than the other forms of tourism (business, beach, and other), addressed poverty problems, and created rural economic opportunities. When compared to the other forms of tourism, safari tourism was found to generate the highest GDP, as well as significantly greater household income. This is important, considering the limited economic opportunities in rural areas surrounding most of the protected areas in this region.

Wildlife viewing safaris, in which tourists visit protected areas to observe wildlife in natural habitats, is the main tourism activity in both Kenya and Tanzania (Okello, Wishitemi & Lagat, 2005; Okello &

Yerian, 2009; Tanzania NBS, 2017; Damania *et al.*, 2019). The UN World Tourism Organization (UNWTO) estimates that wildlife watching represents 80 percent of the total annual sales of trips to Africa (UNWTO 2015). In Tanzania, wildlife tourism revolves around the protected areas within the "Northern Circuit" (Serengeti, Kilimanjaro, Lake Manyara, and Arusha National Parks, and Ngorongoro Conservation Area), which account for over 80 percent of the tourists visiting the country (Okello & Yerian, 2009). Likewise, in Kenya, wildlife tourism revolves around the Masai Mara National Reserve, bordering the Serengeti National Park, and Amboseli National Park, which are the most popular safari destinations in Kenya. Consumptive wildlife tourism in the form of hunting is also significant in the Tanzanian portion of the landscape, where it is permitted in game reserves, GCAs, and WMAs. According to key informants, this has resulted in significant investment interest in game reserves and GCAs where hunting is permitted. No hunting takes place in the Kenyan portion of the landscape due to a national prohibition.

The Serengeti-Mara wildlife landscape, with its rolling hills and open grasslands, supports an incredible profusion and variety of wildlife, including big cats and herds of elephants, which can be seen up close throughout the year. The great wildebeest migration takes place between July and October each year and is the world's largest terrestrial migration of wildlife, where more than 1 million wildebeest travel from the Serengeti National Park to the Masai Mara National Reserve. This landscape is also home to the world's largest populations of zebra, eland, lion, cheetah, hyena, and gazelles. However, wildlife across the region is in dramatic decline (Ogutu *et al.*, 2016; Damania *et al.*, 2019). Data from Kenya's Department of Resource Surveys and Remote Sensing estimate that in the past three decades, Kenya has lost more than half of its wildlife biomass (Damania *et al.*, 2019). Wildlife habitats have become fragmented and wild herd numbers have shrunk considerably, declining on average by 68 percent between 1977 and 2016 (Ogutu *et al.*, 2016). Most concerning is that the recent monitoring assessment by Ogutu *et al.*, (2016) suggests that the long-term declines in wildlife numbers are occurring at the same rates within Kenya's protected areas as outside of them. Critically, wildlife depends as much on surrounding adjacent land for continued viability as it does on protected areas. Therefore, pressures from outside of the parks are having an impact on wildlife within the parks (Damania *et al.*, 2019). Population growth, the expansion of agriculture, fencing, poaching, a changing climate, and intrusive infrastructure have been identified as the main causes for the decline in wildlife numbers across Kenya. Declining wildlife populations and loss of habitat quality present a serious threat to the future attractiveness of the landscape for wildlife tourism. Furthermore, the mass tourism model promoted by Kenyan and Tanzanian tourism policies and strategic visions has ultimately resulted in overcrowding in the most popular parks, especially in the Masai Mara National Reserve, where tourism visitor density and the concentration of tourism lodges is extremely high.

As noted earlier, there are revenue sharing mechanisms in place that seek to increase the flow of benefits from nature-based tourism to local communities in the landscape. In Tanzania, 30 percent of nature-based tourism revenue from WMAs is shared with districts, while 20 percent is shared directly with communities. Tourist facilities located within WMAs also pay concession fees to communities. Additionally, district councils receive 25 percent of proceeds from hunting blocks, which is meant to be put toward conservation and community development. While laudable in theory, some key informants felt that too much revenue were captured by corrupt leaders, meaning money did not necessarily reach the grassroots level as intended. Some key informants also reported that revenues generated by WMAs were lower than was hoped for when the concept was first proposed. Nevertheless, significant total revenue was generated by WMAs in the landscape prior to the COVID-19 pandemic, particularly by

Ikona, which is attractively positioned along the northwest boundary of Serengeti National Park. From 2016/2017 to 2017/2018, Ikona generated around US$1.7 million in revenue while Enduimet generated US$250,000 (URT, 2018). This includes revenue from both photographic tourism and hunting. Communities also receive a share of concession fees for any lodges developed within WMAs.

In Kenya, the KWS does not have any formal mechanisms for sharing a specific portion of nature-based tourism revenue with communities. However, some conservancies do support local development projects such as schools and healthcare facilities and development of road and water supply infrastructure. Even so, some community informants expressed dissatisfaction here too, stating these benefits were not sufficient. Communities in the Kenyan portion of the landscape also benefit from revenue raised by conservancies. For example, in the case of the group conservancies leased out to ecotourism operators around the Maasai Mara National Reserve, land owners benefit from lease fees, which reportedly ranged from around US$18 to US$50 per hectare per month prior to the COVID-19 pandemic (Oduor, 2020). In Mara North, which appears to be a particularly lucrative conservancy, the average landholding per conservancy member is 60.7 ha and average monthly revenue is US$50, resulting in annual lease fees of around US$3,000 for the average landholder. Apart from revenue sharing, community key informants noted that they benefit from wildlife-based tourism through payments for cultural tours and selling crafts to tourists. Notably, one key informant noted that species associated with HWC, such as carnivores and crop-raiding species like elephant and buffalo, are only tolerated by communities due to the tourism benefits associated with these species. This highlights the importance of maintaining the flow of nature-based tourism benefits to local communities who bear the costs of HWC.

Park visitor numbers across the wildlife landscapes in Kenya increased steadily over the period 2015–2018, with the four major parks experiencing a 19-24 percent annual growth in visitor numbers over this time (Figure 18). In 2018, the Masai Mara National Reserve experienced a 69 percent increase in visitor numbers compared to the year before, increasing from 172,700 to 291,200 visitors. Tsavo East also experienced significant growth in numbers in 2018, from 120,500 to 167,700 visitors, an increase of 39 percent. In total, 710,200 tourists visited these parks in 2018 compared to 496,017 tourists in 2017, an increase of 43 percent.

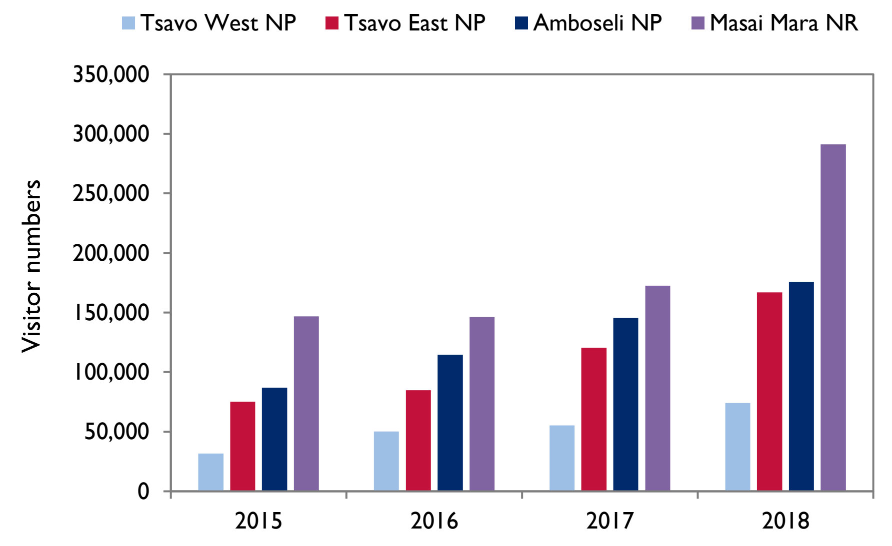


Figure 18. Total number of visits to Amboseli, Tsavo West, Tsavo East National Parks and Masai Mara National Reserve in Kenya from 2015-2018
Source: KNBS, 2020

The growth in visitor numbers to Tanzanian parks has been less significant (Figure 19). These data represent visitor numbers across all the national parks in Tanzania and not just those within the Great East African Plains wildlife landscape. Data on individual parks were not available. However, given that the national parks within this region account for over 80 percent of the tourists visiting the country, this provides a good understanding of the total number of tourists to these parks each year and how numbers have changed over time. Visitor numbers remained relatively stable over the period 2011-2016, with an average annual growth rate of just 2 percent. In 2018, visitor numbers increased to just under 1.2 million, an increase of 11 percent on the previous year.

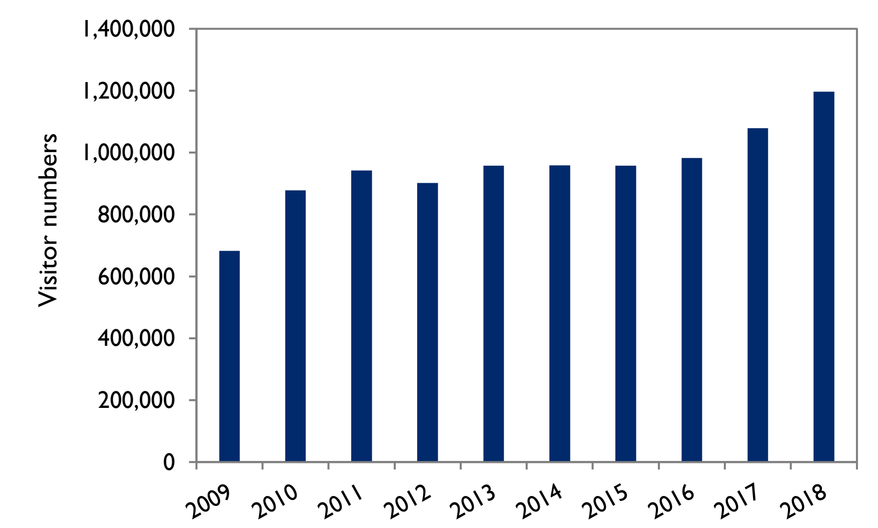


Figure 19. Total number of visits to Tanzanian National Parks from 2009-2018

Source: TANAPA, 2019

Holiday tourists represent most of the tourists in both Kenya and Tanzania (Table 9). These tourists have the highest daily spending and generate the most revenue (Tanzania NBS, 2017). Their spending is entirely attributed to visiting the region's attractions. It is therefore unsurprising that in both Kenya and Tanzania, the total attraction-based tourism value represents such a high proportion of the tourism direct contribution to GDP. The percentage of leisure tourists is high when compared to other East African countries, such as Uganda and Rwanda, where tourists on holiday represented only 22 percent and 7 percent, respectively.

The total attraction-based tourism value in 2018 for Kenya was estimated to be US$1.69 million and for Tanzania US$1.71 million (Table 10). These values were spatially allocated in proportion to photo density (from the InVEST Recreation Model) to generate an estimate of the value of the wildlife landscape, *i.e.*, the proportion of the total attraction-based tourism value associated with the natural areas within the Great East African Plains study region. This represents the nature-based tourism value of these landscapes (Table 10, Figure 20). The total nature-based tourism value of the Great East African Plains landscape was estimated to be US$1.2 billion in 2018: US$508 million in Kenya and US$707 million in Tanzania. In Kenya, this represents 30 percent of the total national attraction-based spend in the country, and in Tanzania it represents 41 percent (Table 10).

Table 9. Typology of tourists to Kenya and Tanzania in 2018

PURPOSE OF VISIT	KENYA (%)	TANZANIA (%)
Holiday	74	64
Visiting Friends and Relatives (VFR)	7	16
Business	13	9
Other	6	11

These values align well with the results from a study in 2009 that estimated that the southern circuit in Tanzania receives 300,000 tourists per year on the 300 km stretch between Arusha and Serengeti, generating a total inbound tourism expenditure of US$500 million per year, more than half of Tanzania's foreign exchange earnings from tourism (UNWTO 2014).

Table 10. The estimated total attraction-based tourism value for Kenya and Tanzania in 2018 and estimated nature-based tourism value of the Great East African Plains landscape; all values in 2018 US$ millions

COUNTRY	TOURISM DIRECT CONTRIBUTION TO GDP	LEISURE SPENDING AS A PROPORTION OF TOTAL SPENDING (%)	TOTAL ATTRACTION-BASED TOURISM VALUE PER COUNTRY	TOURISM VALUE OF WILDLIFE LANDSCAPE	% OF NATIONAL VALUE
Kenya	$2,983 m	64	$1,693 m	$507.8 m	30
Tanzania	$2,762 m	84	$1,712 m	$707.2 m	41

The map of tourism value (Figure 20) clearly highlights the importance of the protected areas in attracting tourists and generating revenues for Kenya and Tanzania. The Serengeti National Park and Ngorongoro Conservation Area in Tanzania have the highest tourism value, generating US$251 million and US$222 million per year (Table 11). The Maasai Mara National Reserve in Kenya generates slightly less at US$218 million per year. However, it has the highest per hectare tourism value, with a value of US$1,439 per hectare per year, compared to US$192 per hectare per year for the Serengeti National Park and US$269 per hectare per year for the Ngorongoro Conservation Area. Amboseli National Park, Kilimanjaro National Park, and Arusha National Park all have a per hectare tourism value that is greater than US$500. The parks to the far east of the wildlife landscape (Tsavo East, Tsavo West, and Mkomazi) have the lowest per hectare values, ranging from US$4 in Mkomazi to US$31 in Tsavo East to US$52 in Tsavo West. These nine parks represent just under 80 percent of the total tourism value for the Great East Africa Plains wildlife landscape. Nature-based tourism in the landscape also generates an estimated $1.5 billion in net benefits (consumer surplus) to international visitors.

Table 11. Tourism value of selected protected areas within the Great East African Plains landscape

PROTECTED AREA	COUNTRY	TOURISM VALUE (US$ MILLIONS/Y)	TOURISM VALUE (US$/HA/Y)
Serengeti National Park	Tanzania	250.7	192
Ngorongoro Conservation Area	Tanzania	222.0	269
Maasai Mara National Reserve	Kenya	218.3	1,439
Kilimanjaro National Park	Tanzania	109.8	594
Tsavo East National Park	Kenya	41.9	31
Tsavo West National Park	Kenya	36.2	52
Amboseli National Park	Kenya	34.2	884
Arusha National Park	Tanzania	6.2	557
Mkomazi National Park	Tanzania	1.4	4

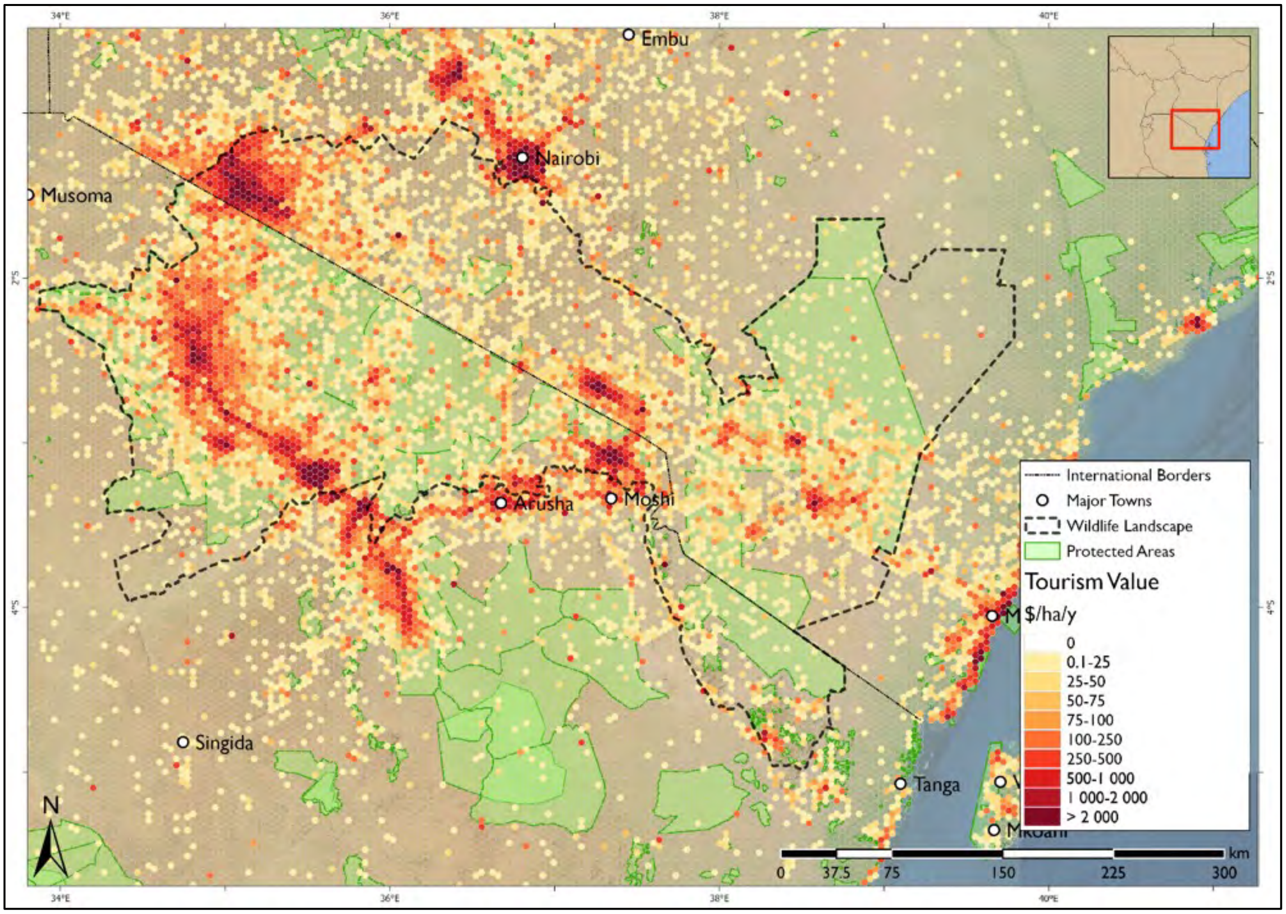


Figure 20. Tourism value (US$/ha/y) for 2018 across the Great East African Plains wildlife landscape, based on the distribution of geo-referenced photographs uploaded to Flickr

FLOW REGULATION

Natural ecosystems regulate seasonal surface flows through infiltration of rainfall into groundwater flows, and in so doing reduce the seasonal variation in flows by slowing down water through the landscape and contributing to river base flows during the dry season. This reduces the size of reservoirs that are needed to meet water demands, as well as affecting the availability of water to people who draw water directly from streams and rivers. In this study, the flow regulation service was evaluated as the difference in the contribution to baseflow (*i.e.*, water that reaches a stream) between current land cover and a scenario in which all land cover is converted to bare ground.

The Great East African Plains landscape was estimated to have an average baseflow contribution of 830 m^3 per hectare per year, the lowest of all the wildlife landscapes studied. Recharge, and thus contribution to baseflow, is generally higher in areas under natural vegetation and higher rainfall, although soil characteristics and relief are other moderating factors. The highest local recharge values in the modelled region were often associated with high rainfall forested areas, such as Mount Kilimanjaro and the Mau forests (Figure 21).

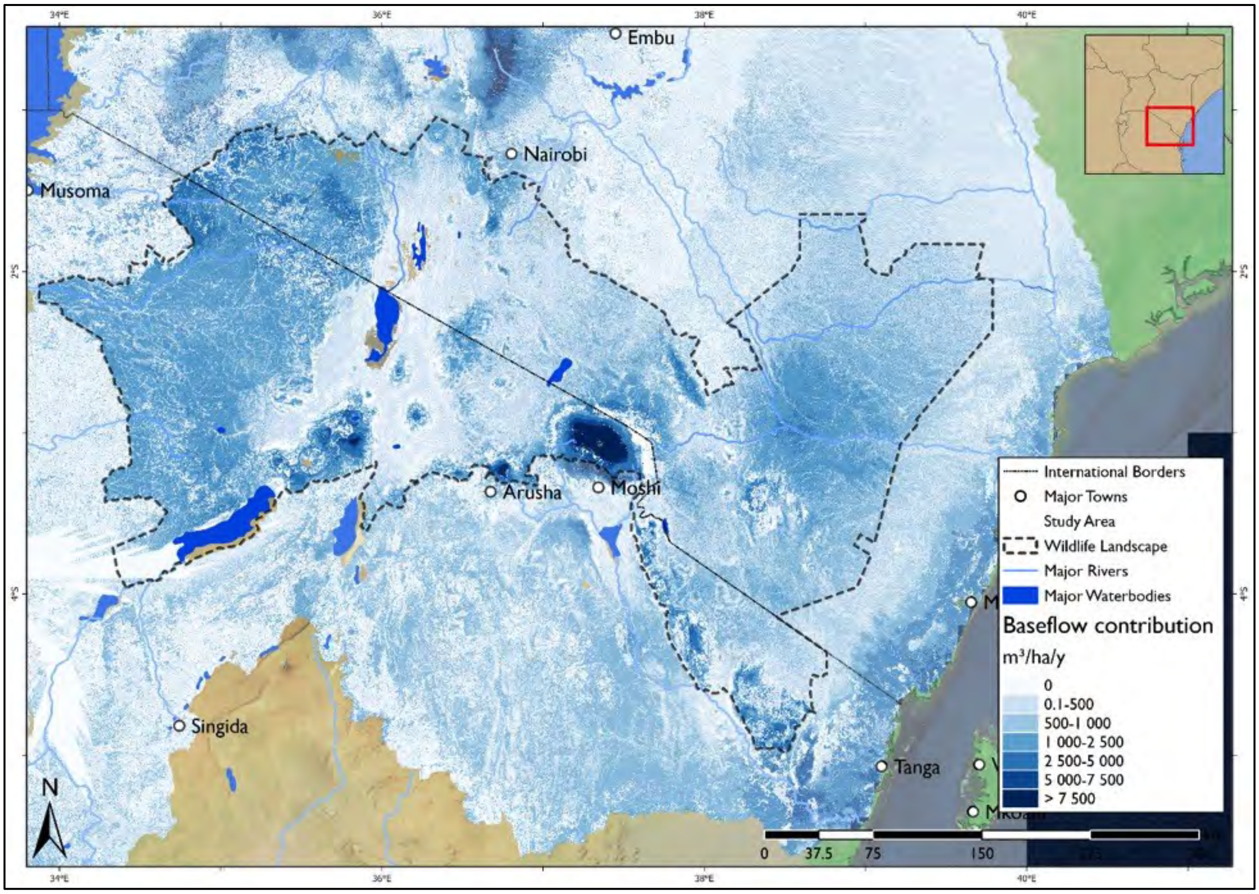


Figure 21. Baseflow contribution (m^3 per ha per year) by ecosystems of the Great East African Plains landscape relative to a barren landscape

Much of the relatively wet Mara and northwest Serengeti regions of the wildlife landscape also make a high contribution to local baseflows. In contrast, the dry Rift Valley regions make little to no contribution to baseflows. Despite moderate to high rainfall, baseflow contributions are low across much of the cultivated land to the north and west of the study region, suggesting that the clearance of natural habitats has resulted in declined dry season flows in these regions. Contributions to baseflow are generally higher in the eastern part of the wildlife landscape, particularly around Tsavo East, where a localized area of elevated rainfall exists from around the Taita Hills northwards. The moister coastal belt to the east of the wildlife landscape also has generally high values for baseflow contribution.

We estimated the total baseflow contribution (*i.e.*, recharge) across the Great East African Plains landscape to be 12,391 million m^3 per year, retaining 8,986 million m^3 per year when compared to a barren landscape. The average water retention was estimated to be 621 m^3 per hectare per year in Kenya and 580 m^3 per hectare per year in Tanzania. If these flows were not being infiltrated and retained by the landscape, the cost of having to construct storage infrastructure that would be needed to capture these additional flows was estimated to be approximately US$1.0 billion per year, US$549 million per year in Kenya and US$454 million in Tanzania.

WATER QUALITY AMELIORATION

Levels of nutrient export to watercourses are strongly linked to cultivation, with much higher nutrient export from agricultural lands due to the application of manure and inorganic fertilizers (Odada *et al.*, 2004). Due to limited cultivation, estimated nutrient export is low across much of the Great East African Plains habitat where natural vegetation remains (Figure 22). This contrasts sharply with much higher nutrient levels outside the wildlife landscape, most notably in the heavily cultivated areas in the Kenyan highlands and Lake Victoria regions to the north of the wildlife landscape, as well as the Lake Victoria Basin regions of Tanzania west of the wildlife landscape. These trends suggest a high level of nutrient export into Lake Victoria from these extensive cultivated areas, with the wildlife landscape contributing much lower nutrient loads. Nutrient export is particularly high from cultivated land on the Kenyan side, as Kenyan farmers on average use more fertilizer than their Tanzanian counterparts. Nutrient export is generally low over much of the eastern part of the wildlife landscape and surrounding areas. This reflects the much lower levels of cultivation and human population densities in this dry region, relative to the higher potential agricultural areas to the west.

For valuation of the nutrient retention service, we focused on the parts of the wildlife landscape situated within the Lake Victoria Basin. This encompasses most of the Serengeti-Mara region. Based on the outputs of the InVEST model, the Lake Victoria catchment area of the Great East African Plains landscape generates nutrient loads on the order of 682 tons of phosphorus per year. This is of particular concern as it contributes to the eutrophication of the lake (Figure 22). We estimated both the active and passive nutrient retention service provided by this part of the wildlife landscape. The active service refers to the current retention of nutrients by the wildlife landscape. If vegetation in the Lake Victoria portion stopped retaining phosphorus, we estimated the replacement cost of the retention service to be on the order of US$512,796. The value of the active service is fairly small, as most of the cultivated areas exporting large loads of phosphorus are located downstream of the wildlife landscape. Since nutrient loads are generally low in the wildlife landscape, the amount retained by vegetation will in turn be low. However, if this portion of the wildlife landscape were converted to agriculture, phosphorus loads entering Lake Victoria would be much higher, as demonstrated by the high nutrient export values in cultivated areas along the western boundary. The nutrient export avoided by maintaining natural vegetation, at the expense of cultivation, is the passive service provided by the wildlife landscape. For the portion of the wildlife landscape falling within the Lake Victoria Basin, we estimated the replacement cost of the passive service would be on the order of US$871 billion.

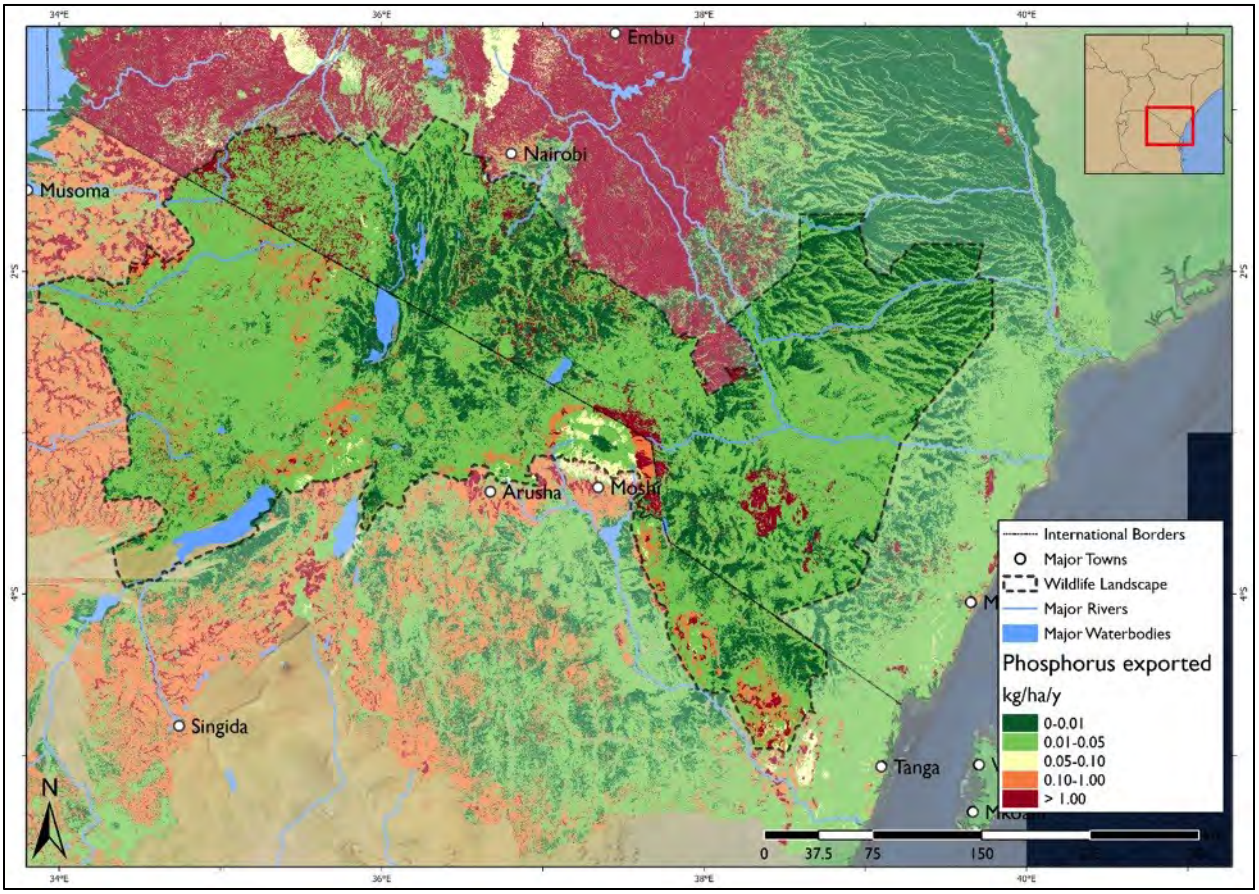


Figure 22. Average phosphorus exported (kg per ha per year) by ecosystems of the Great East African Plains landscape

EROSION CONTROL

Natural habitats reduce soil erosion and transport of sediment to downstream habitats. This can occur through both *in situ* retention of soil due to vegetation cover as well as through the trapping of sediments that have been eroded from elsewhere in the landscape. It reduces the negative impacts of excess sediment loads in watercourses, such as reduced water quality and loss of reservoir storage capacity. In this study, the sediment retention service was evaluated by the difference in sediment export between current land cover and a scenario in which all land cover is converted to bare ground. This difference provides a measure of the amount of sediment being retained by the landscape.

Areas with the highest values for sediment retention are associated with regions with high potential soil loss, which are most strongly linked to steep slopes and high rainfall (Figure 23). This can be seen around Mount Kilimanjaro, the Ngorongoro Highlands, and the Usambara Mountains in the southeast corner of the wildlife landscape. Notably, the headwaters of the Pangani Basin drain from Mounts Kilimanjaro and Meru, while part of the Usambara Mountains also drain into the Pangani further downstream. Given the high erosion risk in these steep, high rainfall areas, maintaining natural vegetation cover makes a crucial contribution to reducing sediment export to the Pangani River system, an area of great importance for agricultural and domestic water needs in northeast Tanzania.

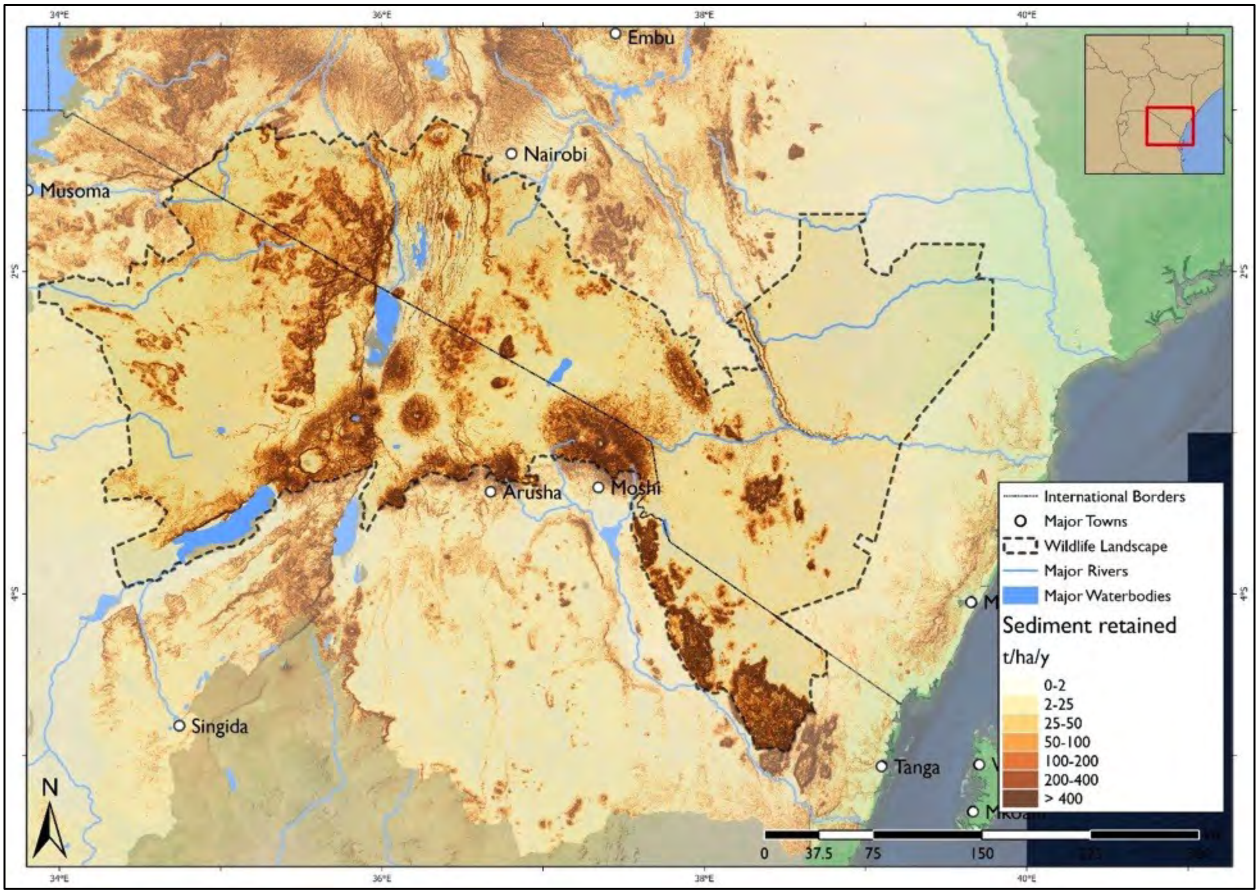


Figure 23. Average sediment retained by ecosystems in the Great East African Plains landscape (tons per ha per year) relative to a barren landscape

By reducing sediment build up in dams, this service is also highly beneficial for hydropower generation facilities in the Pangani Basin, such as Tanzania's Nyumba ya Mungu Reservoir. Another important region for sediment retention is the Mara-Serengeti portion of the wildlife landscape. Although sediment retention per hectare is not as high as in the more mountainous parts of the wildlife landscape, much of the Mara-Serengeti ecosystem is situated in the upper reaches of the Lake Victoria Basin. As a result, natural vegetation in these protected areas makes a significant contribution to avoiding sediment export into Lake Victoria. Large areas with low sediment retention values extend across much of the eastern part of the wildlife landscape, where moderate to low rainfall and gentle slope mean potential soil loss is relatively low. Nevertheless, the sizeable areas of natural vegetation in this region do make a valuable contribution to reducing sediment export into the lower Athi River, which flows through Tsavo East National Park.

In total, the model estimated that current land cover retains 1.8 billion tons of sediment (124 t/ha/y) per year, relative to a scenario in which all land cover is converted to bare ground (Table 12). Most of this retention falls within Tanzania (73 percent) where the average retention is 195 tons per hectare per year, compared to Kenya where the average retention was much lower at 61 tons per hectare per year.

If this sediment were not being retained by the landscape, the replacement cost of this service, in terms of the construction of sediment check-dams, was estimated to be US$2.2 billion, per year.

Table 12. Total sediment retained, mean sediment retained per hectare per year, and the total annual cost of sediment retention (US$ million/y) for the East Africa Plains wildlife landscape

COUNTRY	TOTAL SEDIMENT RETAINED (MT/Y)	MEAN SEDIMENT RETAINED (T/HA/Y)	TOTAL ANNUAL VALUE (US$ MILLION/Y)
Kenya	484	61	594.1
Tanzania	1,329	195	1,632.6
Total	**1,813**	**124**	**2,226.7**

CARBON STORAGE

Natural ecosystems make a significant contribution to global climate regulation through the sequestration and storage of carbon. About half of all vegetative biomass comprises carbon. In addition to accumulation in woody biomass, carbon accumulates in soils and peat as a result of the accumulation of leaf litter and partially decayed biomass. Degradation of vegetated habitats releases carbon and contributes to global climate change with impacts on biodiversity, water supply, droughts and floods, agriculture, energy production, and human health, whereas restoration or protection of these habitats mitigates or avoids these damages, respectively. The conservation and restoration of natural systems thus helps to reduce the rate at which greenhouse gases accumulate in the atmosphere and the consequent impacts of climate change.

The landscape is dominated by grassland, wooded grassland, and moist and dry bushland. While there are some pockets of intact Afromontane forest, they cover a relatively small area. Although grasslands are less carbon-dense than bushland and forest, where above-ground vegetation makes up only a small proportion of the total carbon pool, they play an important role in mitigating climate change through the sequestration of soil carbon (Dlamini, Chivenge & Chaplot, 2016). Indeed, grasslands are estimated to contain up to one-third of above- and below-ground carbon stocks globally (Tessema *et al.*, 2020). Furthermore, the soil organic carbon pool in grasslands is critically important for soil fertility and plant productivity, contributing to flow regulation. Soil organic carbon is an important indicator of grassland productivity, and deep-rooting African grasses have been shown to be highly productive in sequestering carbon (Tessema *et al.*, 2020). The perennial nature of grasslands allows for the continuous input of carbon from above-ground vegetation to the subsoil via extensive root systems, to depths of several meters. As a result, soil carbon contributes more than two-thirds of the ecosystem carbon that is found in grasslands (Dlamini *et al.*, 2016). However, grasslands are threatened by overgrazing (amongst other factors, but this is the main contributor to grassland degradation, which leads to soil compaction) and evidence suggests that the impacts of overgrazing on soil organic carbon can be significant, with global losses in grassland soil organic carbon stocks of between 1.2 percent and 4.2 percent as a result (Dlamini *et al.*, 2016; Tessema *et al.*, 2020).

Based on global datasets derived from satellite data (see FAO & ITPS 2018; Spawn & Gibbs 2020), it was estimated that approximately 4.6 billion tons of carbon are stored within the vegetation and soils of the

Great East African Plains wildlife landscape, approximately 57 percent in Kenya and 43 percent in Tanzania (Figure 24, Table 13).

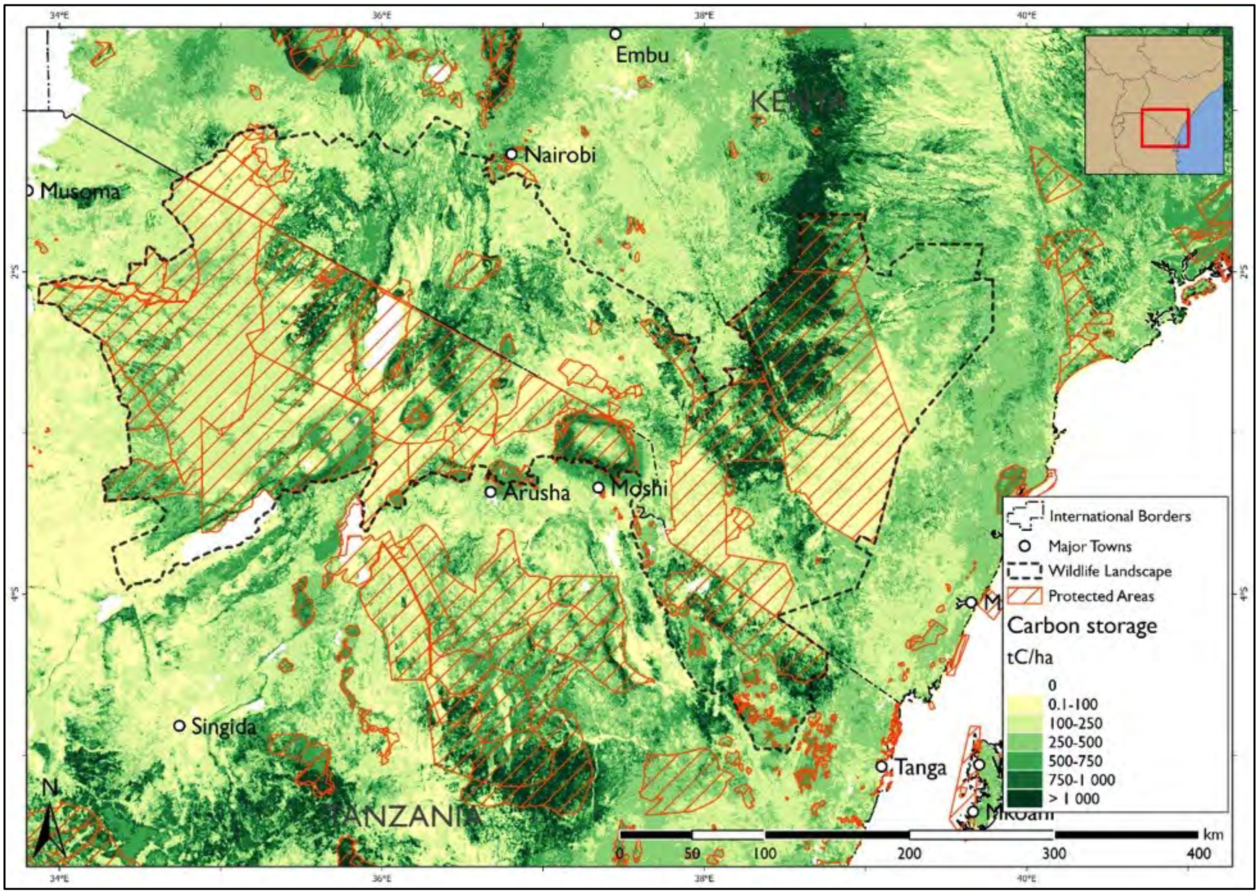


Figure 24. Total carbon storage (metric tons/ha) across the Great East African Plains wildlife landscape

The minimum, maximum, and mean values for carbon stored per hectare are very similar across the two countries, ranging from as little as 5.7 tons per hectare to as much as 1,540 tons per hectare. Expectedly, the mean of 336 to 382 tons per hectare is much lower than the mean recorded in the Albertine Rift Forests landscape. Densities are highest in the northeast of the landscape where bushland dominates, and in areas of Afromontane forest, such as around Mount Kilimanjaro. It has been estimated that a ton of carbon released into the atmosphere will cause global damages on the order of US$417 (net present value over 80 years), of which Kenya's share is US$0.61 per ton and Tanzania's share is US$1.04 per ton (Ricke *et al.*, 2018). The total global damage costs avoided by retaining the total stock of biomass carbon is significant, at almost US$400 billion per year (Table 14). The avoided damage cost to Kenya is estimated to be US$290 million per year, while the avoided damage costs to Tanzania are about US$500 million per year.

Table 13. The total amount of carbon stored within the Great East African Plains landscape and summary statistics (tons carbon per hectare) per country, in metric tons

COUNTRY	TOTAL STOCK OF CARBON (TONS)	MEAN T/HA	MIN T/HA	MAX T/HA
Kenya	2,623,486,060	335.69	8.31	1,539.82
Tanzania	1,974,801,929	381.98	5.73	1,334.65

Table 14. The total global damage costs avoided by retaining the total stock of biomass carbon and the avoided damage cost to each country (US$ million/y)

	KENYA	TANZANIA	REST OF THE WORLD
Carbon storage value (damage costs avoided, US$ million/y)	290.1	497.8	397,136

POLLINATION OF CROPS

Pollination services are widely recognized as critical for human wellbeing and survival given their role in ensuring food security. However, the value of wild pollinators remains unclear. This is concerning for sub-Saharan Africa, a region highly dependent on subsistence agriculture as a main source of livelihood (Tibesigwa *et al.*, 2019). The presence of wild pollinators is directly linked to natural vegetation (Kremen *et al.*, 2004), which plays an essential role in certain life cycle stages of pollinator species, such as through the provision of nesting sites or forage at certain times of year. Insects are responsible for 80-85 percent of all pollinated commercial crops, which represents about one-third of global food production (Allen-Wardell *et al.*, 1998; Klein *et al.*, 2007).

Outside protected areas, smallholder agriculture plays an important food security role, even though crops are cultivated on a smaller scale and generally produce low yields compared to some of the other study regions. This is due to the semi-arid nature of this region and the associated erratic rainfall. The main crops grown include maize, cassava, beans, and some vegetables. In the Kilimanjaro-Meru region, communities are able to grow a variety of crops such as coffee, mangos, cashews, and plantains. In this area, crop yields are significantly higher than on the far western side of the landscape. While not all of these crops require insect pollination (*e.g.*, maize), the majority that do (*e.g.*, vegetables, fruits, coffee, beans, groundnuts) experience reduced yields (of up to 90 percent) in the absence of wild pollinators or show a reduction in seed/breeding yield without wild pollination (*e.g.*, cassava and cocoyams).

The distribution and value of the pollination service is shown in Figure 25. The value of wild pollination service (contribution to production from crops in smallholder cultivated land) is summarized in Table 15. Based on the percentage share of natural vegetation within a 1,000 m buffer distance of all cultivated land surrounding the wildlife landscape, we estimate the value of wild pollination services to nature-dependent smallholder cultivated land in this study region to be US$592 million per year.

Close to 60 percent of this total value falls within Tanzania, contributing US$338 million to crop production each year, and in Kenya slightly less at US$254 million per year. The mean per hectare value of US$131 (and range US$0–985/ha) aligns with estimates of pollination value in the Kakamega region of western Kenya (US$32–2,430/ha; Kasina *et al.,* 2009). The pollination service appears to be most valuable in the north- and southwestern sections of the landscape surrounding the Masai Mara National Reserve and the Serengeti National Park, the north central areas, and the area around the Usambara Mountains in the southeast. The natural vegetation in these areas is extremely important for wild pollinators, contributing significantly to crop production.

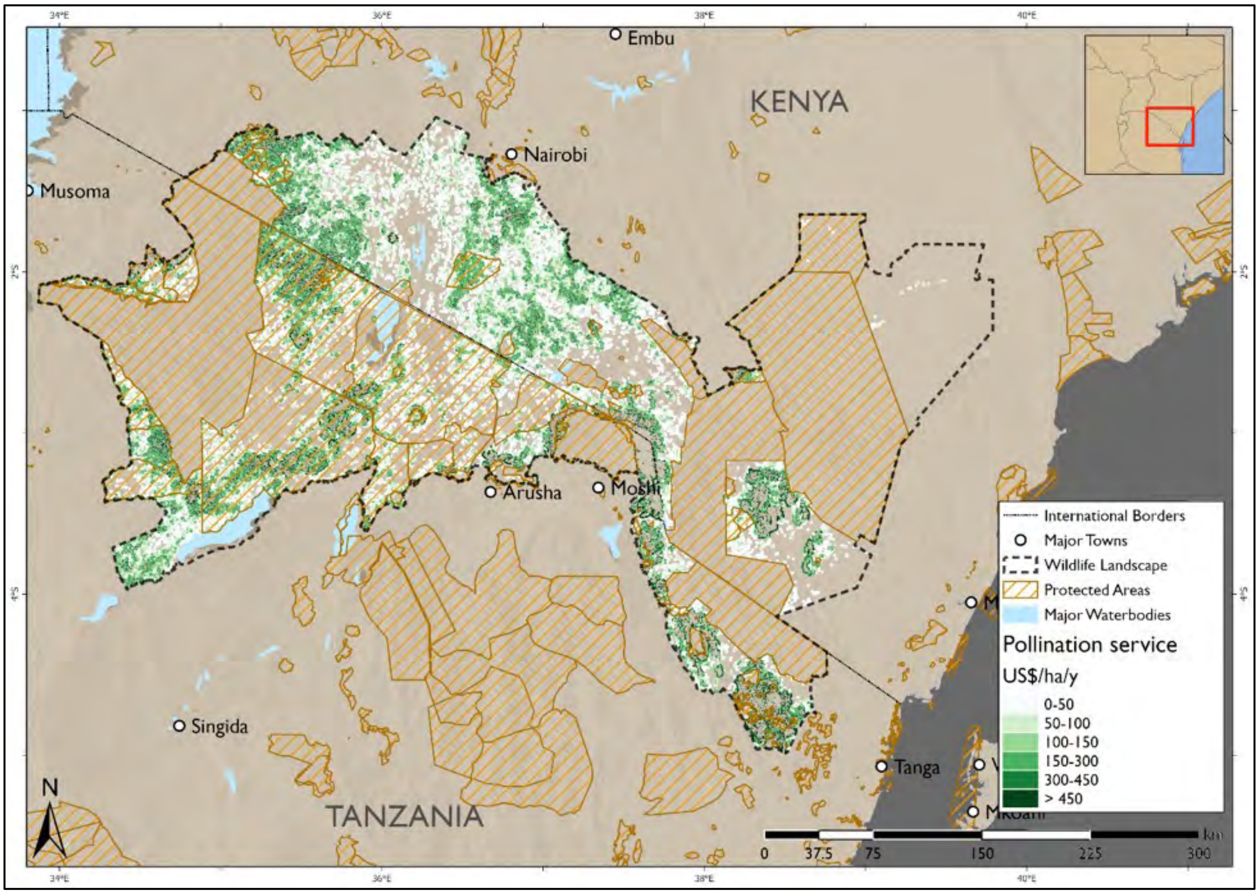


Figure 25. Contribution of pollination services from natural habitats to smallholder farmer revenues in the Great Eastern Plains wildlife landscape (US$/ha/y)

Table 15. The value of the wild pollination service within the Great East African Plains wildlife landscape (2018 US$)

	TOTAL HA OF NATURAL VEG IN BUFFERS	TOTAL POLLINATION SERVICE VALUE (US$)	MAX VALUE (US$/HA)	MEAN VALUE (US$/HA)
Kenya	2,261,621	253,897,511	974	112
Tanzania	2,240,453	338,069,407	985	151
Total	**4,502,074**	**591,966,919**	**985**	**131**

Note that the minimum value in all cases was zero.

FORAGE FOR LIVESTOCK PRODUCTION

The Great East African Plains are home to pastoral communities (Maasai pastoralists) who are dependent on extensive livestock production, mainly cattle, with some sheep and goats. Pastoralism or nomadic herding is an extensive livestock production system that is generally practiced in arid and semiarid lands where pastoralists move across a landscape with their animals in search of grazing lands and water. Livestock is the most important source of livelihood and food security for these communities. Indeed, it has been estimated that more than 75 percent of cattle herds in Kenya and 90 percent in Tanzania are kept by pastoralists, who are responsible for supplying the bulk of meat that is consumed in both countries (Nyariki & Amwata, 2019). Notably, much cross-border movement of livestock takes place, especially cattle. This is driven by the search for water and forage as well as markets. Key informants reported that livestock tend to come into conflict with wild animals during these cross-border movements, as well as with disease-causing pests such as tsetse fly and ticks. However, a framework to regulate the cross-border movement of livestock has not been established. Thus, a joint livestock program between Kenya and Tanzania was recommended to manage the movement of livestock and control the cross-border spread of livestock diseases. Key informants also made various other recommendations for improving livestock productivity in the landscape, including the use of high quality and drought-adapted livestock breeds, diversifying livestock species and meat consumption, and implementing fodder conservation practices and grazing management plans.

It is estimated that there are 2.3 million LSUs across the Great East African Plains wildlife landscape, with 1.2 million LSUs in Kenya and 1.1 million LSUs in Tanzania. The density of livestock was estimated to be 0.21 LSUs per hectare in Kenya and 0.16 LSUs per hectare in Tanzania (refer to Figure 9).

Livestock production contributes some US$3.4 billion to Kenya's GDP and US$4.3 billion to Tanzania's GDP (in 2018 prices; Kenya National Bureau of Statistics, Tanzania National Bureau of Statistics). The value of livestock production was mapped based on 2010 density estimates for cattle, sheep, and goats at 10 km resolution (refer to Figure 9). Within the Great East African Plains wildlife landscape, it is estimated that the natural rangelands contribute US$557.5 million per year to livestock production: US$247.8 million in Kenya and US$309.6 million in Tanzania, representing 7.3 percent and 7.1 percent of national output, respectively. Based on the FAO livestock distribution map, the value of this service was highest in the semi-arid western parts of the wildlife landscape adjacent to the Serengeti National Park and northeast of the Maasai Mara National Reserve.

HARVESTED RESOURCES

In addition to their conservation value, natural resources play an important role in supporting the livelihoods of people. Wild plant and animal resources are harvested for food, animal feed, medicine, energy, and raw materials, particularly where there are limited economic opportunities. The capacity of the landscape to supply different types of wild resources is related to vegetation type and condition, availability of water, and other factors. However, several other considerations determine their use and value, and these vary in space and time. The accessibility of wild resources is determined by regulations such as land tenure and harvesting rights, social norms and informal agreements, geographic features such as topography and rivers, and human-made features such as roads. The demand for wild resources is influenced by the socio-economic circumstances of households and the prices of alternatives.

Wildlife habitats usually require full, no-take protection, not only because of the risk associated with overharvesting that changes the nature and functioning of wildlife habitats, but also because of the disturbance that it can cause, especially affecting the shier and more vulnerable wildlife species. The people that live around these wildlife habitats are largely dependent on wild resources, particularly during times of economic stress. Examples of such stressors include crop disease, drought or floods that are likely to only worsen with climate change, and international pandemics such as COVID-19. During these times, people fall back on nature to fill livelihood needs. However, this is a potentially vicious cycle of unsustainability as more people rely on nature for food and raw materials and stocks become depleted. The stocks of resources protected within parks and reserves help to maintain the stocks utilized outside of these protected areas. The more resources harvested unsustainably, the fewer there will be available in the future and the less people can rely on nature to fill this need. As resource stocks outside of wildlife habitats become degraded, there will be a higher demand for the resources on the edge of these landscapes as well as on the inside.

THE DEMAND FOR NATURAL RESOURCES

The people that live in the areas bordering the Great East African Plains wildlife landscape belong to a wide array of ethnic groups that undertake different livelihood activities. These activities are to a great extent governed by rainfall, with large parts of this region being arid or semi-arid and not suitable for the cultivation of crops. Across the central parts of this landscape, in both Kenya and Tanzania, the semi-nomadic Maasai pastoralists form the majority of the population (Figure 26). They rely exclusively on livestock production as their main source of income. In these areas, there is minimal crop production, and where crops are grown, yields are very low. Livestock and livestock products are sold for cash in order to buy food crops from markets. During times of economic stress, bush products are collected and sold (FEWS NET, 2010a). The cultivation of crops for household consumption and for sale is the dominant livelihood activity for the people living in the western areas adjacent to the Serengeti National Park. The main cash crops grown are cotton, sweet potato, and rice, and maize and cassava are grown by most households for own consumption. Wealthier households supplement their income through the sale of livestock and livestock products and poorer households through the sale of poultry (FEWS NET, 2008). To the east in the areas surrounding Kilimanjaro and Meru, rainfall is more reliable, and soils are more fertile, allowing for the production of a wide variety of fruit and vegetable crops. Livestock production is of less importance in this area. In the south-central area of the wildlife landscape adjacent to Lake Eyasi, the Hadzabe (Hadza ethnic group) are true hunter-gatherers that rely on the collection of wild fruits, roots, and bush meat for food (FEWS NET, 2008).

Virtually all households in the study region depend on firewood or charcoal as an energy source (Table 16). Of these, the vast majority use firewood; studies from across the region consistently reporting that firewood is used in 90-100 percent of households (Hosier, 1984; Jensen, 1984; Mung'ala & Openshaw, 1984; Emerton, 1996; Giliba et al., 2010; Schmitt, 2010; Bär & Ehrensperger, 2018). Estimates of consumption are more varied, with estimates of per capita annual fuelwood consumption ranging from 355-897 kg (Jensen, 1984; Mung'ala & Openshaw, 1984; Biran, Abbot & Mace, 2004; Wiskerke et al., 2010) to 1,679-4,190 kg annual firewood consumption per household (Hosier, 1984; Wiskerke et al., 2010; Bär & Ehrensperger, 2018). Charcoal use occurs in from 3 to 16 percent of households (Hosier, 1984; Schmitt, 2010; Bär and Ehrensperger, 2018). For households that do use charcoal, estimates of annual consumption in the region range from 380-600 kg (Hosier, 1984; Bär & Ehrensperger, 2018). A positive relationship between rural consumption of both firewood and charcoal with wood availability has been reported (Mung'ala & Openshaw, 1984). This may underlie variability in the consumption of wood fuels across the study region, where vegetation varies from grassland to forest patches. Several households may also harvest fuelwood to produce charcoal for income generation, rather than subsistence use.

According to Drigo *et al.* (2015), the Kenyan counties in the study area have sufficient wood stocks to sustain local rural consumption when considered at the overall county level. However, due to its relatively accessible wood resources and proximity to urban areas, Drigo et al.'s model predicts heavy commercial fuelwood harvesting in Kajiado, resulting in significant overexploitation of wood stocks when both rural subsistence consumption and commercial harvesting are considered. The ban on commercial harvesting of wood for charcoal in Kenya has simply driven the activity underground as people still need and rely on charcoal as a fuel source (K. Mutu, pers. comm.).

Figure 26. Top: The Maasai people in Kenya and Tanzania are semi-nomadic pastoralists and cattle are their main source of income. Bottom: The Hadzabe ethnic group in Tanzania grow no food and raise no livestock; they rely exclusively on wild resources for their food and energy needs.

Credit: Alan Harper (top); Kiwiexplorer (bottom left); Amanda Jane Fletcher (bottom right).

Wood is also widely used as a construction material in the study area, both for building houses as well as other structures like fences or bomas for livestock (Table 16). Wood use for construction of houses and bomas among the Maasai in general has been estimated to be 0.52 m^3 per year (Chamshama, S.A.O. Kerkhof & Singunda, 1989). A more locally specific estimate was made by Jensen (1984), who gauged wood consumption for construction to be 101 kg per person among the Maasai of the Amboseli region.

Table 16. Proportion of rural households harvesting woody resources for wood fuel and raw materials within each country in the Eastern Plains study region and the estimated demand per average household per year

COUNTRY	FIREWOOD		CHARCOAL		POLES & WITHIES		TIMBER	
	% RURAL HH	M^3/HH/Y	% RURAL HH	M^3/HH/Y	% RURAL HH	M^3/HH/Y	% RURAL HH	M^3/HH/Y
Kenya	78	3.0	15	0.2	70	0.3	3	0.1
Tanzania	87	3.8	25	0.4	64	0.3	3	0.1

The use of grasses for thatching is highly variable across the study region (Table 17). In Kenya, the use of grasses for thatching of roofs ranges from as low as 1 percent of households in Makueni County to 21 percent in Narok County and 35 percent in Tana River County (Kenya National Bureau of Statistics (KNBS), 2019). In Tanzania, grass thatching is highest in the Mara and Simiyu regions (24 percent) and much lower in the regions of Arusha and Kilimanjaro (4 percent) where most roofs are constructed using corrugated metal sheeting. However, estimates of quantities harvested are limited. In the Pangani Basin in Tanzania, Turpie *et al.* (2005) estimated that on average 34 grass bundles were harvested each year per household.

Figure 27. Houses in the Mara region adjacent to the Serengeti National Park are constructed out of mud, wooden poles, and grass thatching. Right: A typical Maasai homestead where houses are made from mud and branches.

Credit: Jay Galvin (left); John Atherton (right).

Reeds and sedges are not an important resource in this study region as they are not particularly common or abundant (Table 17). There was no evidence to suggest that reeds and sedges are harvested in any of the Kenyan counties of the study region. However, in Tanzania they are known to be harvested in the Pangani Basin (some parts of the Arusha and Kilimanjaro regions) and also in certain parts of the

Mara region where the Mara River drains into Lake Victoria. In the Pangani Basin, around 7 percent of households harvested reeds and sedges, taking an average of 38 bundles per household per year (Turpie *et al.*, 2005). Palm leaves are even less common than reeds and sedges in the study region, with very few households harvesting palm leaves for use in thatching and the making of household items such as mats and baskets.

Table 17. Proportion of rural households harvesting non-woody raw materials within each country in the Great East African Plains study region and the estimated demand per average household per year

COUNTRY	PALM LEAVES		REEDS AND SEDGES		THATCHING GRASS	
	% RURAL HH	KG/HH/Y	% RURAL HH	KG/HH/Y	% RURAL HH	KG/HH/Y
Kenya	2	0.9	-	-	12	4.0
Tanzania	1	0.7	3	7.6	12	4.4

Communities in this region are heavily reliant on traditional plant medicines, which are used by an estimated 70 percent of Kenyans (Odera, 1997) and 80 percent of Tanzanians (Walter, 2001). Traditional medicine is widely practiced even in large urban areas, fueling commercial harvesting of medicinal plants from rural areas for sale in towns (Cunningham, 1997). Local estimates for harvesting of medical plants in the study region are particularly high (Table 18). For example, Emerton (1996) found that 85 percent of households harvested medicines from Oldonyo Oruk forest along the Kenya/Tanzania border, while 95 percent of households were found to harvest medicines from the Chyulu Hills (Kenya Water Towers Agency, 2018). The wooded nature of these areas may underlie the high levels of medicinal plant harvesting, as it may result in increased variety of medicinal species.

Herbs and wild fruits are commonly consumed by rural households in the area, though few studies have quantified this. In Kajiado County, at least 60 percent of households regularly consumed wild fruit when available, while at least 70 percent regularly used wild herbs in cooking (Oiye *et al.*, 2009). Harvesting of wild foods was slightly lower around Oldonyo Oruk, where Emerton (1996) reported that 49 percent of households undertook the activity.

Table 18. Proportion of rural households harvesting wild plants foods and medicines within each country in the Great East African Plains study region and the estimated demand per average household per year

COUNTRY	WILD PLANT FOODS		MEDICINES	
	% RURAL HH	KG/HH/Y	% RURAL HH	KG/HH/Y
Kenya	35	21.7	63	3.8
Tanzania	19	11.9	35	2.1

Honey harvesting varies significantly across the study region (Table 19) and appears to be related to woody cover. In the Serengeti region where grassland dominates, only 2 percent of households were estimated to harvest honey (Schmitt, 2010). Giliba et al. (2010) reported that 40 percent of households harvested honey from the Nou Catchment Reserve in the Manyara Region just south of the study region. Household participation in honey harvesting was higher still around Oldonyo Oruk (64 percent) (Emerton, 1996) and Chyulu Hills (68 percent) (Kenya Water Towers Agency, 2018). All three of these regions contain areas of woodland or forest, which may account for the higher harvesting levels.

Significant quantities of bushmeat are consumed in the study region (Table 19). High bushmeat consumption across much of the study region is driven by a number of factors, including its lower price and often greater availability than domestic meat, preferences for the taste of wild meat, and belief that it is healthier than domestic meat (TRAFFIC, 1997; Loibooki *et al.*, 2002; Ndibalema & Songorwa, 2008; Rentsch & Damon, 2013). Bushmeat may be a particularly important food source during times of economic hardship. Hunting is conducted for both subsistence and commercial purposes, which has the potential to generate substantial incomes for hunters relative to other livelihood options. In the western Serengeti and Kilimanjaro regions, up to two-thirds of hunted meat is traded for income, compared to about one-third in Kenya's Kitui County (TRAFFIC, 1997). Overall, high variability in estimates of bushmeat consumption were found across and even within different regions of the study area. Ethnicity is one important influencing factor (Ndibalema & Songorwa, 2008), with little traditional interest in bushmeat hunting among the Maasai (Homewood & Rodgers, 2004). Bushmeat consumption is much more prevalent in ethnic groups with a stronger tradition of hunting, such the WaSukuma and Walkongo around the southwestern part of the Serengeti (Homewood & Rodgers, 2004), and the Kamba around Chyulu Hills and Tsavo National Parks (TRAFFIC, 1997; Kenya Water Towers Agency, 2018). In addition to ethnic differences, the illegal nature of bushmeat may lead to a fear of disclosing consumption, with many studies noting that involvement is likely under-estimated as a result (Kaltenborn, Nyahongo & Tingstad, 2005; Knapp *et al.*, 2010; Mfunda & Røskaft, 2010; Nuno *et al.*, 2013).

Bushmeat hunting has been well studied in the Serengeti region, especially to the west of the park where bushmeat consumption is particularly high. Studies in this region estimate 8-37 percent of households engage in hunting (Loibooki *et al.*, 2002; Schmitt, 2010; Mfunda & Røskaft, 2011; Nuno *et al.*, 2013; Manyama, 2020). However, since many households buy rather than hunt their own bushmeat, as much as 75-82 percent of households have been estimated to consume bushmeat in communities of the western Serengeti (TRAFFIC, 1997; Loibooki *et al.*, 2002). A lower estimate of 42 percent of households consuming bushmeat was made by Schmitt (2010), while across the three villages surveyed by Knapp *et al.* (2010), the proportion of households in each village admitting to eating bushmeat in the previous year ranged from 36-71 percent. Bushmeat consumption is lower in rural communities east of the Serengeti, where the Maasai are the dominant ethnic group. Schmitt (2010) reported that 27 percent of households in this region admitted to consuming bushmeat. Further east, TRAFFIC (2000) estimated that 68 percent of households consumed bushmeat in the Kilimanjaro region. Around Chyulu Hills, about 68 percent of Kamba households were estimated to consume bushmeat, while consumption was much lower among the Maasai, at around 13 percent (Kenya Water Towers Agency, 2018). High consumption among the Kamba around Tsavo East National Park has also been reported, with 80 percent estimated to consume bushmeat regularly (TRAFFIC, 1997). Quantities of bushmeat consumed are high in the Western Serengeti region, where preferred larger species are readily available, with annual household consumption of 140 kg reported by Rentsch and Damon (2013), and per capita consumption of 11-32 kg across different districts in the area (Ndibalema & Songorwa, 2008). At 169 kg per household (25 kg per capita), consumption was similarly high among the Kamba of Kitui County

(TRAFFIC, 1997). However, in the more densely populated and cultivated Kilimanjaro region where large wildlife species are scarce, a much lower household consumption of 19 kg per year was found (TRAFFIC, 1997). People resort to eating a wider variety of smaller species in such areas where preferred large species have disappeared.

Table 19. Proportion of rural households harvesting wild animal resources within each country in the Great East African Plains study region and the estimated demand per average household per year

COUNTRY	MAMMALS, BIRDS		WILD HONEY		FISH	
	% RURAL HH	KG/HH/Y	% RURAL HH	LITERS/HH/Y	% RURAL HH	KG/HH/Y
Kenya	44	75.1	14	1.0	1	1.7
Tanzania	28	46.7	3	0.2	11	25.6

THE SUPPLY, USE, AND VALUE OF HARVESTED WILD RESOURCES

To briefly recap, the resource use results are the combined product of natural resource stocks, the availability of these resources for harvesting (protected area status), and the local demand for the various resources. Stocks of natural resources per unit area varied according to habitat type and condition. However, the supply of natural resources was also moderated by protected area status, as we reduced the proportional availability of natural resources where they occurred within protected areas. The magnitude of this reduction varied according to the level of protection. Finally, the data for available stocks per hectare was combined with estimated household demand per hectare. Demand is a function of both the average quantity of resources used per household, and the number of households in the area (population density).

Our model estimated low average resource use per hectare for the Great East African Plains wildlife landscape (Table 20). Quantities used were generally substantially lower than for the other wildlife landscapes. This can be attributed to several factors. Firstly, population densities, and thus demand for natural resources, are low across most of the region, although large cities like Nairobi and Arusha and dense rural populations are found just outside the wildlife landscape. There is also a relatively high coverage of large protected areas, particularly in the Tanzanian portion, resulting in reduced availability of natural resources for harvesting. A third factor is that much of the region is covered by habitats with moderate to low stocks of most natural resources, such as *Acacia-Commiphora* bushland. For these reasons, continuous areas of relatively intact natural resources and low to moderate resource use/ha dominate the Great East African Plains wildlife landscape (Figure 28-31). This includes large areas outside of protected areas, such as the Kajiado rangelands of southern Kenya, where limited cultivation means extensive natural land cover remains, albeit not necessarily in a pristine state. The total value of wild harvested resources was estimated to be US$195.7 million across the landscape, US$66.9 million in Kenya and US$128.7 million in Tanzania.

The use of most harvested resources follows a similar spatial pattern of low to moderate use across the wildlife landscape, with localized areas of elevated use where resource stocks and/or population densities are higher (Figure 28-Figure 31). This is often associated with wetter, more wooded areas, like the Taita Hills between Tsavo West and East National Parks in Kenya, areas around Mounts Kilimanjaro and Meru in Tanzania, and Tanzania's Eastern Arc Mountains in the southeast corner of the study region. In contrast, zero use of natural resources was estimated for the interior of Tsavo East. This is because it was the only region where the population data predicted large areas with a population density of zero, reflecting the much lower population densities in this particularly dry region of the wildlife landscape. Fuelwood had the highest average monetary value per hectare of all harvested resources in the Tanzanian portion of the wildlife landscape (Table 20; Figure 28). However, wild plant foods and medicines had a higher average monetary value per hectare than fuelwood on the Kenyan side. Notably, wild plant foods and medicines were the only resource with higher average use per hectare in the Kenyan portion of the wildlife landscape (Table 20; Figure 30). This reflects the substantially greater average household use of these resources in the Kenyan portion of the wildlife landscape (Table 18). Apart from fuelwood and wild plant foods and medicines, all other natural resources had average use values of less than US$1/ha, reflecting low harvesting across this generally sparsely populated wildlife landscape. While most resources considered followed the aforementioned pattern of low to moderate use across most of the wildlife landscape, reeds, sedges, and fish had much more localized distributions (Figure 29 and Figure 31). Both resources were estimated to have a sparse distribution across the wildlife landscape, with the highest use associated with wetlands in Tanzania in the southeastern part of the region.

Table 20. Average quantities, monetary values per hectare, and total value (US$ millions) for subsistence harvesting of wild resources in the Great East African Plains study region

RESOURCE		KENYA			TANZANIA		
	UNITS	USE (UNITS/HA)	US$/HA	TOTAL US$ MN	USE (UNITS/HA)	US$/HA	TOTAL US$ MN
Fuelwood	m^3	0.10	1.94	28.0	0.25	4.64	78.4
Poles and withies	m^3	0.01	0.27	3.5	0.02	0.46	7.5
Timber	m^3	< 0.01	0.35	4.4	< 0.01	0.60	11.0
Thatching grass	kg	0.08	0.03	0.6	0.20	0.08	1.5
Reeds and sedges	kg	0.01	< 0.01	0.1	0.13	0.06	-
Wild plant foods and medicines	kg	2.65	2.64	25.6	2.16	2.30	24.6
Bushmeat	kg	0.80	0.66	4.5	1.05	0.87	5.3
Honey	l	0.02	0.01	0.1	0.01	0.01	0.1
Fish	kg	0.01	< 0.01	0.1	0.05	0.02	0.3

While natural resource use is low to moderate across most of the wildlife landscape, much higher use values occur in more densely populated rural areas surrounding the wildlife landscape (Figure 28-Figure 32). Notable examples include areas west of the Serengeti-Mara protected areas, and areas adjacent to

Kilimanjaro and Arusha National Parks. Due to extensive conversion of land to agriculture, the remaining natural resource stocks are very patchily distributed outside protected areas in these regions, resulting in high use pressures on the small areas of natural habitat that remain. This has given rise to hard protected area edges, particularly in the western Serengeti, where the contiguous areas of natural resources inside protected areas contrast strongly with the much patchier distribution of resources in the adjacent unprotected areas. Given the high demand and limited natural resource stocks along the Serengeti National Park's western border, rapid depletion of natural resources would be expected in the absence of adequate protection. A similar situation occurs around Arusha and Kilimanjaro National Parks, and, to a lesser extent, the Masai Mara National Reserve. Due to the dense surrounding populations, habitats within the boundary regions of these protected areas were also predicted to have relatively high natural resource usage. In contrast, pressure on natural resources surrounding protected areas is generally lower in the dry, far eastern parts of the study region, particularly around Tsavo East National Park.

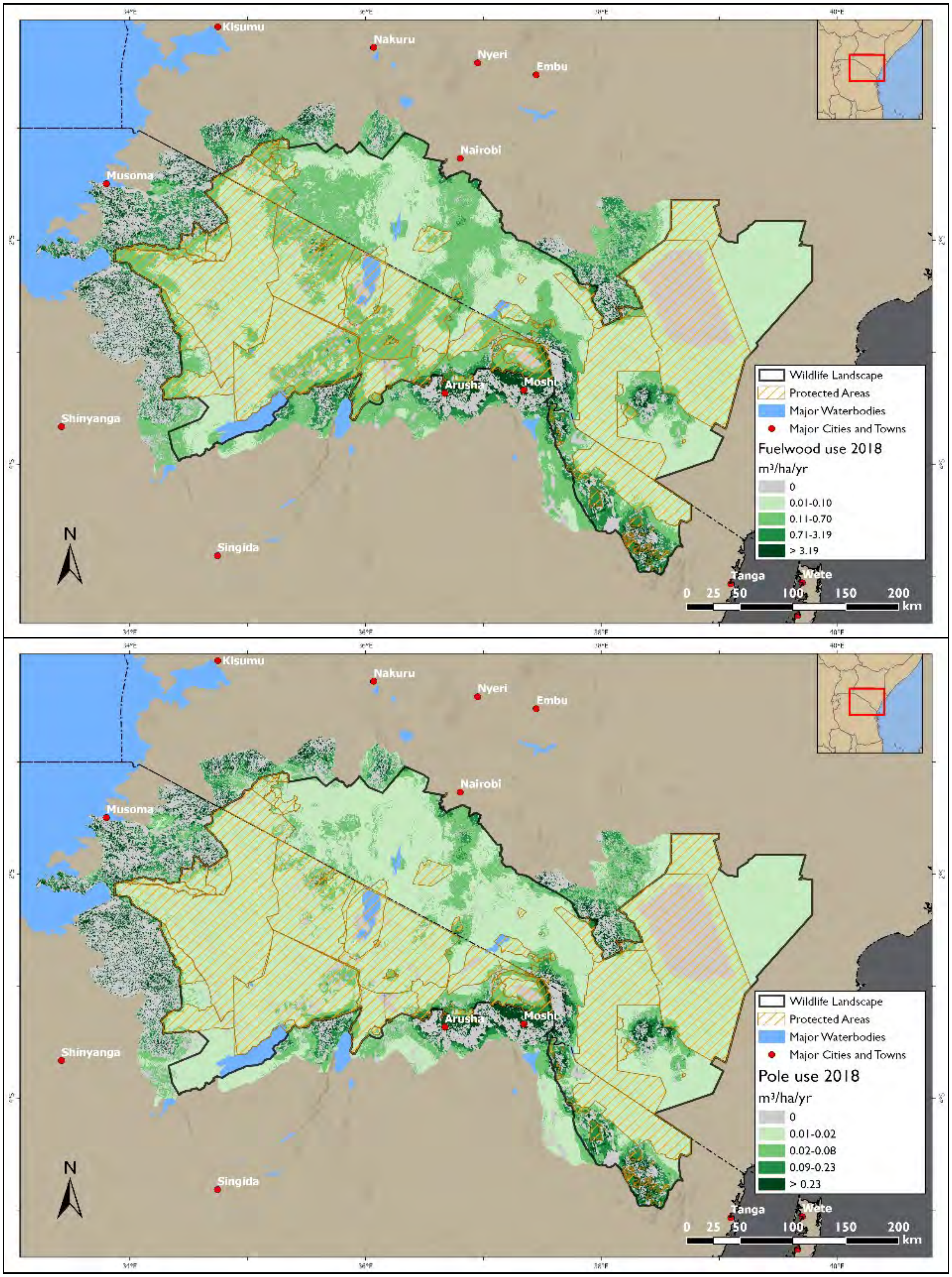


Figure 28. Estimated variation in the subsistence harvesting of fuelwood (top) and poles (bottom) across the Great East African Plains region

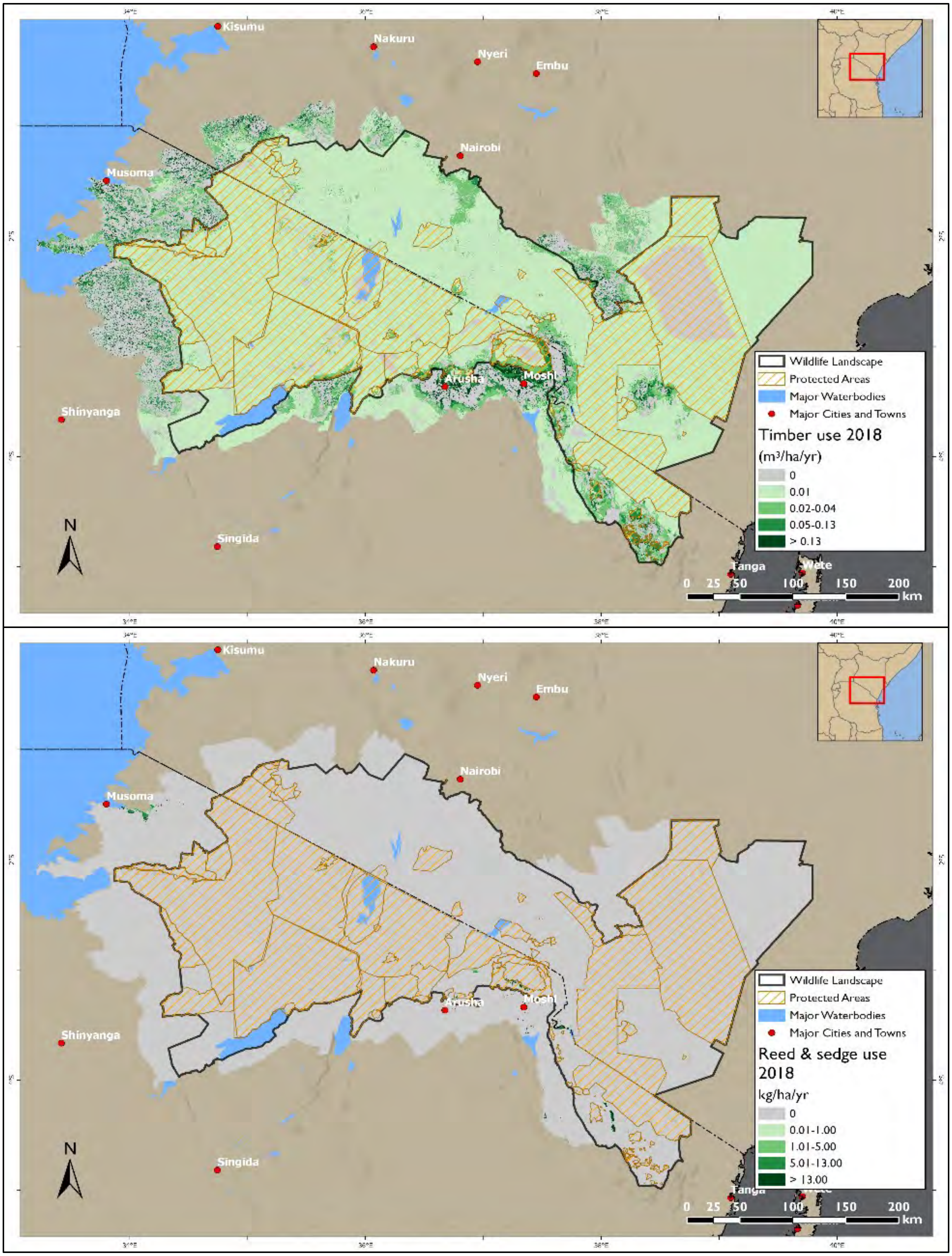


Figure 29. Estimated variation in the subsistence harvesting of timber (top) and reeds and sedges (bottom) across the Great East African Plains region

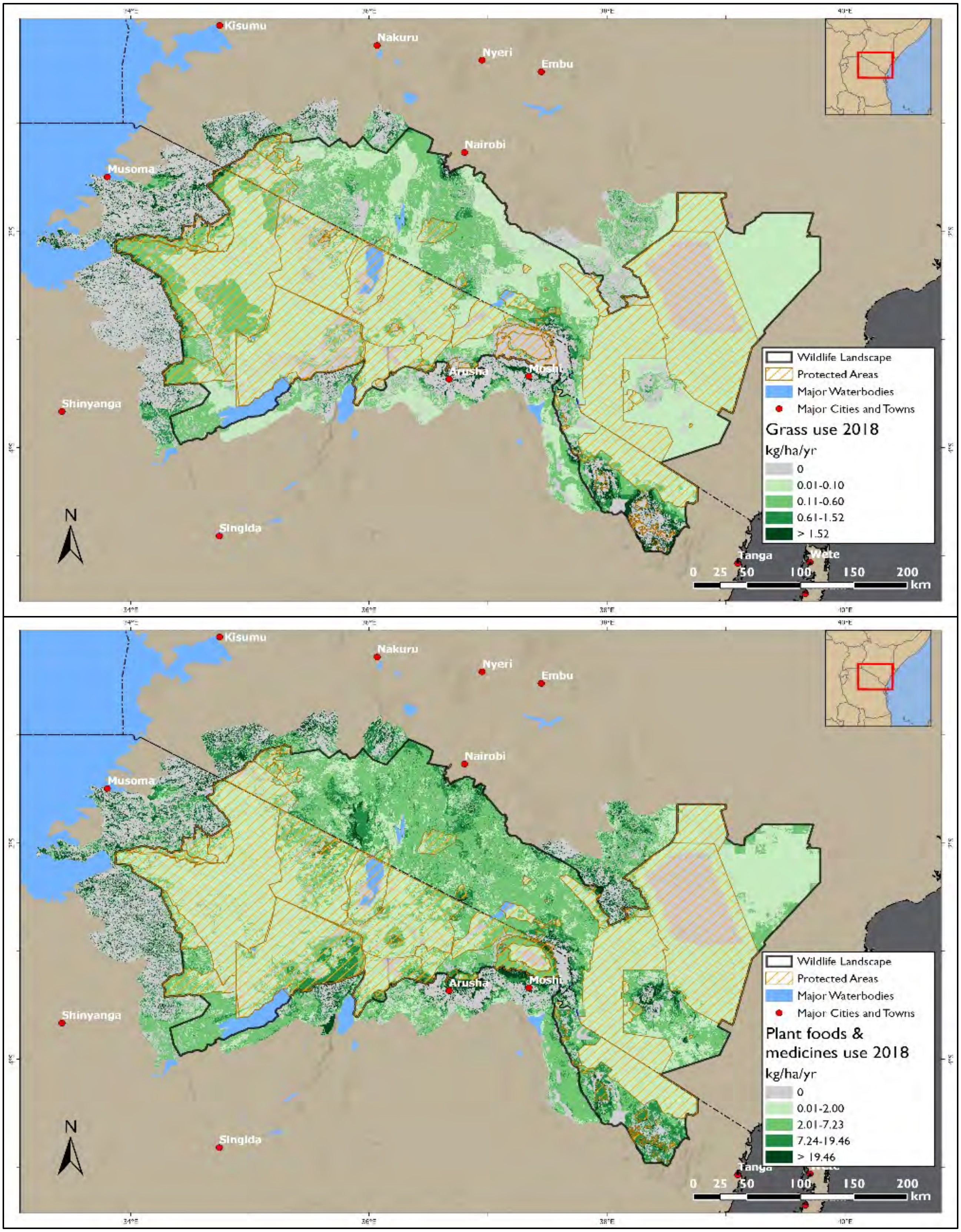
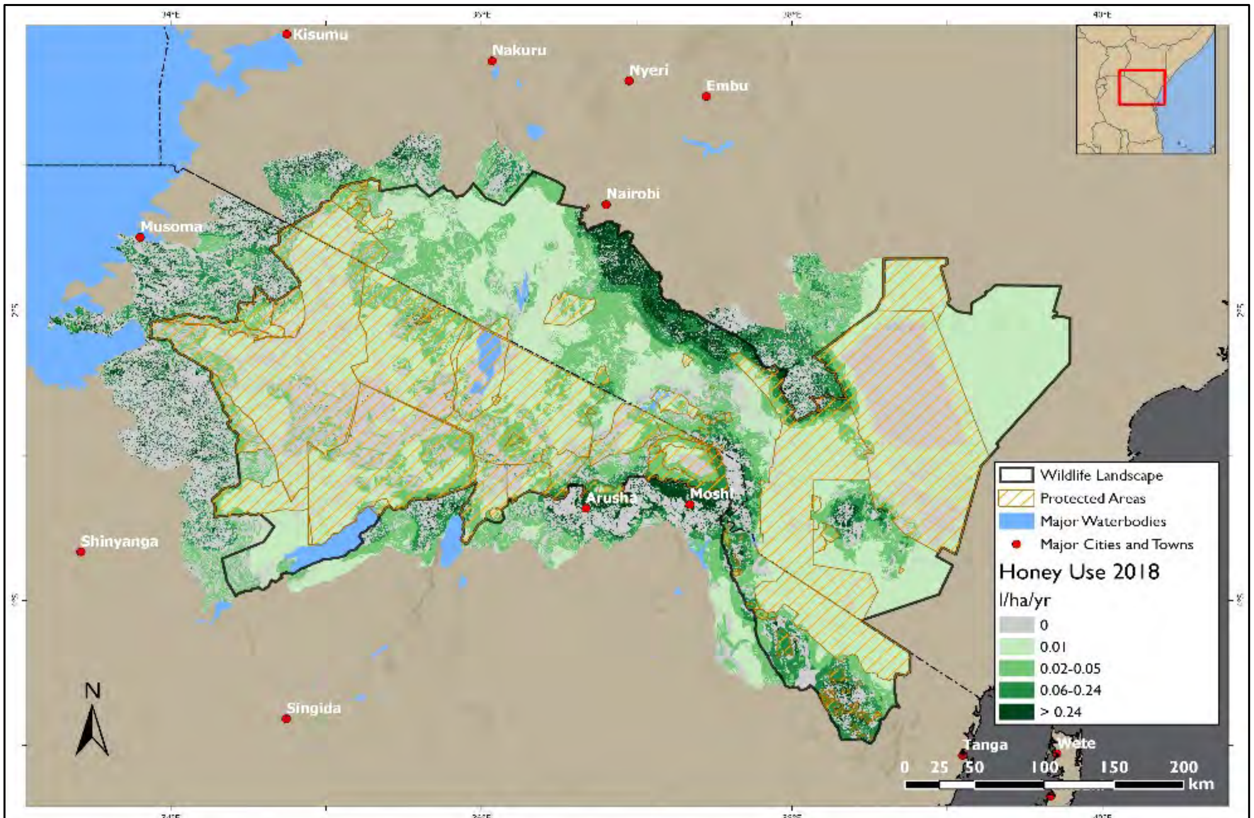


Figure 30. Estimated variation in the subsistence harvesting of thatching grass (top) and wild plant foods and medicines (bottom) across the Great East African Plains region



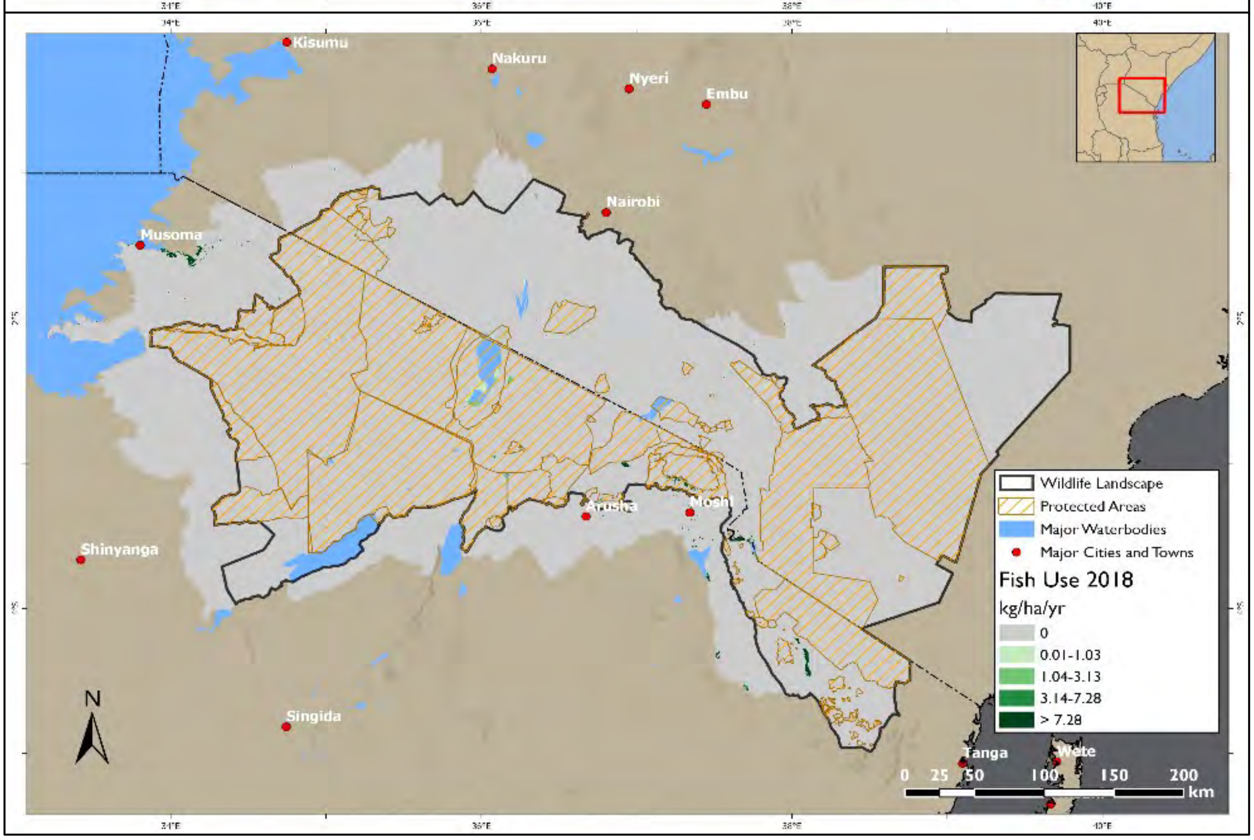


Figure 31. Estimated variation in the subsistence harvesting of honey (top) and fish (bottom) across the Great East African Plains region

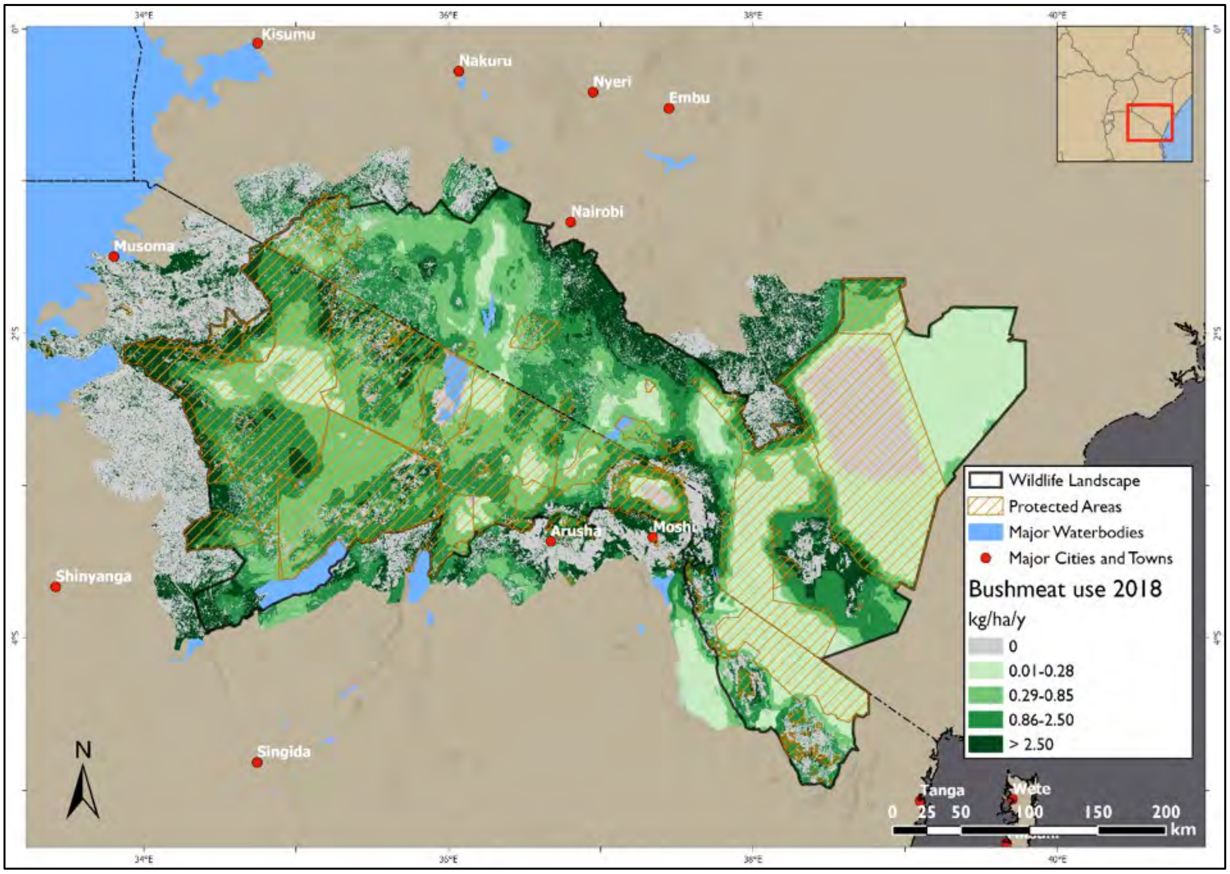


Figure 32. Estimated variation in the subsistence harvesting of bushmeat across the Great East African Plains region

SUMMARY

The Great East African Plains wildlife landscape supports the largest wildlife populations on earth. This area encompasses some of the most famous protected areas in Africa, drawing more than 1 million visitors each year, bringing significant tourism benefits to Kenya and Tanzania. The total direct contribution to GDP of nature-based tourism was estimated to be more than US$1.2 billion in 2018, the highest of the four study areas. Most of this value is associated with the Serengeti National Park and Ngorongoro Conservation Area in Tanzania and the Masai Mara National Reserve in Kenya. These protected areas account for 21 percent of Tanzania's and 11 percent of Kenya's total tourism value, respectively. This tourism also generates an estimated $1.5 billion in net benefits to overseas visitors.

Keeping the wildlife habitats in their current natural condition generates costs savings for the region that could be worth about US$3.2 billion per year, through regulation of hydrological processes and atmospheric carbon. Based on our high-level modelling exercise, these systems contribute an estimated 9 million m^3 in terms of rainwater infiltration and temporary storage, worth US$1 billion per year. They are estimated to retain about 1.8 billion tons (= metric tons) of sediment per year, which would otherwise end up in lakes, reservoirs, estuaries, and coastal environments; this service has a replacement cost value of US$2.2 billion per year. In addition, the wildlife habitats of this landscape that fall within the catchment areas of Lake Victoria are estimated to reduce phosphorous loadings by some

853-4,855 tons per year (depending on what alternative land use it is compared to), which has a replacement cost of up to US$871,000 per year. These estimates should be refined in future with more detailed modelling at finer scales, and with the provision of reliable monitoring data on environmental processes in Kenya and Tanzania. Based on satellite data, the vegetation and soils of this wildlife landscape also store an estimated 4.6 billion tons of carbon, the retention of which, according to the most recent estimates, would avoid local climate change damages of some US$788 million per year. In addition, retention of these carbon stocks avoids damages of almost US$400 billion per year at a global scale, which is twice the 2018 GDP output of the East African region of just under US$200 billion.

The wildlife habitats also contribute to agricultural production within the landscape and around their margins. Wild pollinators in the wildlife landscape were estimated to increase crop production by some US$592 million per year. In addition, the Great East African Plains are home to pastoral communities who are dependent on extensive livestock production. Here the natural rangelands were estimated to support livestock production worth some US$557.5 million per year in terms of contribution to GDP.

Wildlife habitats outside of strictly protected areas (and to some extent within these areas, although not always legally) provide a wide array of wild resources that play an important role in supporting the livelihoods of people. People living in or close to the Great East African Plains wildlife landscape were estimated to harvest some 5.6 million m^3 of firewood, 55,000 tons of wild fruits, vegetables, and medicinal plants, and 406,000 liters of honey, with an estimated total value of US$195.7 million per year. These consumptive use values were twice as high in Tanzania as in Kenya because of the larger population within the landscape in Tanzania. Including a very conservative estimate of the existence value of biodiversity, the wildlife landscape is estimated to be worth at least $508/ha/year on average to East Africa, and over $31,000 per ha globally.

Table 21. Summary of the benefits derived from ecosystem services of the Great East African Plains wildlife landscape (in US$ millions)

	KENYA	TANZANIA	REGION	REST OF WORLD	TOTAL
Nature-based tourism	507.8	707.2	1,215.0	1,544	2,759
Biodiversity existence	0.8	0.6	1.5	5,372	5,373
Flow regulation	548.8	454.1	1,002.9	-	1,003
Erosion control	594.1	1,632.6	2,226.7	-	2,227
Water quality amelioration	0.3	0.3	0.7	-	1
Carbon storage	290.1	497.8	787.9	397,136	397,924
Crop pollination	253.9	338.1	592.0	-	592
Livestock production	247.8	309.6	557.4	-	557
Harvested resources	66.9	128.7	195.6	-	196
Total value $ millions per year	**2,510.6**	**4,069.1**	**6,579.7**	**404,052**	**410,631**
Total value $ per ha per year	**365.3**	**668.5**	**507.6**	**31,169**	**31,676**

IMPLICATIONS FOR THE FUTURE UNDER A BUSINESS AS USUAL SCENARIO

Compared to much of the East African region, the Great East African Plains transboundary landscape still holds exceptional populations of wildlife outside of protected areas and a large, contiguous area of largely natural habitat. However, a range of pressures threaten to further degrade these unique features of the landscape in the future. Alarming declines in wildlife populations in recent decades highlight the precarious state of the region's natural habitats. Furthermore, existing pressures are set to be exacerbated by climate change in the future. Thus, in the following sections, we start by describing some of the expected impacts of a range of existing pressures based on past trends. We then describe some of the expected impacts of future climate change derived from modelling studies. Finally, we draw together the discussion on existing pressures and future climate change impacts to predict the future of wildlife, habitats, and ecosystem services provided by the transboundary landscape under a business-as-usual scenario.

CONVERSION OF HABITATS TO CULTIVATION AND SETTLEMENT

Conversion of natural and semi-natural habitats to cultivation and settlement is a major ongoing threat to the Great East African Plains landscape. This conversion pressure can be expected to increase in the future under BAU, due to both demographic pressures and shifting livelihood strategies. Population growth over recent decades has varied from about 2-5 percent per year across the region (Norton-Griffiths & Said, 2009; Pricope *et al.*, 2013; United Republic of Tanzania, 2013; Ogutu *et al.*, 2016). In the absence of meaningful alternative livelihood strategies, continuing population growth will further increase the area under cultivation and settlement under a BAU scenario, resulting in further degradation of habitats and dispersal areas outside protected areas (Okello & Kiringe, 2004; Ogutu *et al.*, 2016; Veldhuis *et al.*, 2019). Protected areas themselves will likely experience greater encroachment and conversion pressure as land becomes increasingly scarce outside of them. This is particularly noticeable along the western boundary of the landscape, where dense, increasing human and livestock populations have resulted in hard edges between protected areas of the Serengeti-Mara ecosystem and intensively cultivated lands adjacent to the park boundaries.

Ongoing livelihood shifts to a greater reliance on cultivation for subsistence and/or income are another key driver of agricultural expansion in the area, including among traditionally pastoral groups like the Maasai (Okello & Kiringe, 2004; Kiringe & Okello, 2005; Okello, 2005; Homewood *et al.*, 2009; Norton-Griffiths & Said, 2009; Okello & Kioko, 2011; Ogutu *et al.*, 2016). Questionnaire surveys have revealed that local people see cultivation as more economically lucrative than pastoralism, and as a way of diversifying their income sources (Okello, 2005). Norton-Griffiths & Said (2009) clearly demonstrate the powerful economic incentive to convert rangelands in the region, with agriculture producing substantially higher monetary returns per unit area than livestock and conservation, except in very dry regions. Additionally in the Tanzanian rangelands, the desire to strengthen communal property rights against land alienation, through demonstrating that land is in "productive use," is another driver of shifts to cultivation (Sachedina, 2008; Norton-Griffiths & Said, 2009).

Shifts away from large group ranches to individual ownership of small land parcels have also facilitated and encouraged the expansion of agriculture, particularly in the Kenyan portion of the landscape (Okello & Kiringe, 2004; Okello, 2005; Norton-Griffiths & Said, 2009; Gicheru *et al.*, 2012; Ojwang *et al.*, 2012). Significant subdivision of land has also occurred in Tanzania, such as in the densely populated rural areas of the western Serengeti region (Knapp *et al.*, 2015). Small, individual land parcels are less appropriate for pastoralism, encouraging people to turn to cultivation (Okello, 2005; Gicheru *et al.*, 2012). This tenure shift has been accompanied by an increase in fencing of land parcels to exclude both people and wildlife, contributing to the blockage of natural wildlife movements across the landscape (Homewood *et al.*, 2012; Ojwang *et al.*, 2012). Owners of land parcels may also lease them out for cultivation by outsiders, including large-scale commercial farmers, as has occurred in the Mara region (Gicheru *et al.*, 2012; Ojwang *et al.*, 2012). Government policy is another driver of agricultural expansion, with agricultural policy in both countries tending to favor farming over livestock production, with the latter seen as a less productive use of land (Homewood *et al.*, 2009; Ogutu *et al.*, 2016).

Under a BAU scenario, the net result of these factors would be the further loss of wildlife habitat outside of protected areas, and the intensification of agricultural encroachment pressures on protected areas. Since the 1970s, Norton-Griffiths & Said (2009) reported an 8.6 percent annual growth in cultivated area across Kenya's rangelands, almost triple their reported population growth rate of 3.1 percent per year. Substantial further conversion of natural habitat could thus occur by 2050 under the current demographic, socio-economic, and tenure status of this landscape. This is particularly the case for the wetter portions of the landscape, where substantial conversion to agriculture has already occurred (Norton-Griffiths & Said, 2009). Highly threatened wildlife dispersal routes will likely disappear completely with ongoing land subdivision, cultivation, and settlement under business as usual. The link between Amboseli and Tsavo is one example of a highly threatened, important migration route (Ojwang *et al.*, 2012; Ogutu *et al.*, 2014). Due to the small size of Amboseli, linkages to other ecosystems are crucial for maintaining the viability of its wildlife populations, especially in times of stress such as drought (Ogutu *et al.*, 2014). Overall, these threats suggest the severe wildlife declines reported from the landscape, particularly the Kenyan portion (Ogutu *et al.*, 2016), are likely to continue.

Trends in the rate of conversion of wildlife habitats to cropland or urban areas were based an analysis of land cover data available for the period 1992 to 2018 (ESA CCI land cover data at 300 m resolution; European Space Agency, 2018), adjusting to the more accurate land cover dataset used for the Baseline 2018. These data indicate that while area under crops were growing rapidly in the 1990s, there has been a net decrease in cultivated area since then. Areas under human settlement have grown at variable rates, but there was slight negative growth over the last three years to 2018 (Table 22). These results contradict much of the literature, however (see above). Moreover, the Copernicus 100 m landcover data series, which goes back to 2015, suggests that there has been an increase in cropland in the study area from 2015 to 2018 of 126,801 hectares per year. This highlights the potential inaccuracy of land cover data products and the need for ground-truthing. Based on the literature, the increasing cultivation trend of the Copernicus 100 m land cover appears to be more likely.

Table 22. Extent and annual rates of change of land cover classes in the Great East African Plains region from 1992 to 2004 and from 2004 to 2018.

LAND COVER CHANGE	1992 TO 1998	1998 TO 2004	2004 TO 2010	2010 TO 2015	2015 TO 2018
Average annual change in area under crops (ha/year)	998	-4,967	121	-284	-4,218
Average annual change in built-up area (ha/year)	143	407	58	134	-27

Source: Based on ESA CCI Land Cover 300m resolution (European Space Agency, 2018), adjusted

WOOD HARVESTING

Deforestation has been an increasing cause of habitat degradation in parts of the landscape, particularly due to harvesting of wood to meet growing demands for charcoal (Kiringe & Okello, 2005; Macharia & Ekaya, 2005; Kiringe, Mwaura & Warinwa, 2016; Bär & Ehrensperger, 2018; Kyando *et al.*, 2019). This is particularly the case for parts of the landscape in proximity to large urban centers like Nairobi and Arusha (Drigo *et al.*, 2015; Bär & Ehrensperger, 2018), which are found just outside the wildlife landscape. Supplying wood for the charcoal market may also be seen as an attractive income source by poor rural people with limited livelihood options (Macharia & Ekaya, 2005; Mfunda & Røskaft, 2011; Kyando *et al.*, 2019). Given the rapid urbanization occurring in the region, urban charcoal demand will likely put increasing pressure on the landscape's woody resources in the future. This pressure also exists from the longer-standing rural demand for firewood and building materials. The shift toward individual land tenure and sub-division of group ranches in the Kenyan portion of the landscape has also led to increased demand for wood for fencing (Macharia & Ekaya, 2005). Considering the ongoing growth in rural populations occurring across the landscape, harvesting of woody biomass to meet rural demand can also be expected to increase in the future under the BAU scenario.

OVERGRAZING

An excess of livestock and resultant overgrazing has been a long-standing cause of habitat degradation in parts of the wildlife landscape (Macharia & Ekaya, 2005; Okello, 2005). Over much of the traditionally pastoral parts of the landscape, herders were historically semi-nomadic, moving around to take advantage of new foraging resources and allowing grazed pasture time to recover (Ndagala, 1982; Homewood *et al.*, 2009). This included rotating livestock between wet and dry season grazing areas. However, population growth, the proliferation of settlements, shifts to subdivision and individualized land tenure, and the increasing adoption of cultivation have all led to an increase in sedentarization of livestock herders (Osano *et al.*, 2013; Ogutu *et al.*, 2014). Key informants confirmed that traditional rotational grazing patterns have been disrupted due to these factors as rangeland use becomes increasingly intensive. Climate change has reportedly accentuated these challenges through increased drought and flooding when rains do come, resulting in high soil erosion and loss of grass seeds that could support regeneration. The condition of the region's rangeland areas is crucial for wildlife populations, especially in Kenya where the majority of the wildlife population lives permanently on or seasonally uses pastoral areas outside protected areas (Western, Russell & Cuthil, 2009). The intensification of grazing impacts in recent decades has thus reduced forage availability for wildlife too. Hence, competition and displacement of wildlife by livestock continues to be a major cause of wildlife declines outside of protected areas in the landscape (Ogutu *et al.*, 2014). Protected areas themselves have not been immune to these pressures. Excess livestock populations drive people to graze in protected areas, especially during dry periods and droughts (Okello, 2005). In the absence of improved rangeland productivity and alternative livelihood options, these pressures will likely worsen as populations continue to grow under the BAU scenario. The Serengeti-Mara region, where the availability of grazing land outside protected areas has decreased substantially in recent years (Knapp *et al.*, 2015), provides a particularly severe example of the problem. Grazing by livestock and harvesting of natural resources is causing increased degradation of habitats and displacement of wildlife within protected areas in this region (Knapp *et al.*, 2015; Veldhuis *et al.*, 2019). With population growth expected to continue in this region, degradation of the Serengeti-Mara ecosystem will likely worsen, as increased scarcity of land and natural resources leads to greater encroachment into protected areas. Indeed, under current population projections, the complete elimination of available land in the western Serengeti region could occur within a few generations (Knapp, 2009).

INVASIVE SPECIES

Invasive plant species are another cause of habitat degradation in the landscape. Key informants identified *Ipomoea hildebrandtii* as a major invasive species in the area, which has spread rapidly and suppressed indigenous vegetation. Studies from the region have confirmed the impact of the species at reducing grass biomass and soil moisture and nutrient content, thus further compromising rangeland productivity (Mworia, Kinyamario & John, 2008). Other species of concern mentioned by key informants included *Prosopis juliflora* and *Indigofera spinosa,* which both reduce grass production in rangelands, and *Solanum incanum* (locally known as *Ntulelei*), which has invaded wetlands. Some species indigenous to the area were also reported to have become problematic woody encroachers, including *Vacehllia* (formerly *Acacia) seyal* and *Vachellia drepanolobium*. By reducing palatable grass biomass, these species have a negative impact on both wildlife and livestock. A key informant also noted that invasive species have increased human wildlife conflict in some areas, as wildlife move away from invaded areas into those occupied by livestock.

PROJECTED CHANGES IN TEMPERATURE AND RAINFALL

Total annual precipitation across the Great East African Plains landscape for the period 2040-2060 is expected to increase by just over 8 percent relative to historical (1960-1990) precipitation. The wettest months, November through to April, are expected to get wetter, while May through to October are expected to get marginally drier (Figure 33). Mean annual temperature across the Great East African Plains landscape is expected to increase by 2.3°C (11 percent), with April to August (predominantly dry season months) expected to increase by more than September to March (Figure 34). Geographically, the western regions are expected to get relatively wetter and warmer than the rest of the landscape (Figure 35).

Although at the landscape scale the annual rainfall and temperature projections show an increase in total annual rainfall of 8 percent and an increase in mean annual temperature of increase of 2.4°C, the change in climate will differ markedly across the landscape, particularly changes in precipitation. Table 23 provides the projected change in mean annual temperature (°C) and total annual precipitation (mm) for important protected areas in the region. The expected increase in mean annual temperature from present to 2050 ranges from 2.2°C for Tsavo East National Park to 2.5°C for Maswa Kimali Game Reserve, with a strong east-west gradient in the mean temperature anomaly (Figure 35). The change in total annual precipitation ranges from an increase of 3.7 percent for Mkomazi National Park to an increase of 13.6 percent for South Kitui National Reserve. The highest precipitation anomaly between the baseline and 2050 is for Mount Kilimanjaro and the far northwestern areas in the landscape with a small area in the southeast of the landscape predicted to have decreased annual precipitation by 2050. The change in precipitation could render these areas unsuitable for many species, considering they may experience changes in habitat structure, with possibly denser vegetation becoming more prominent under the increased rainfall.

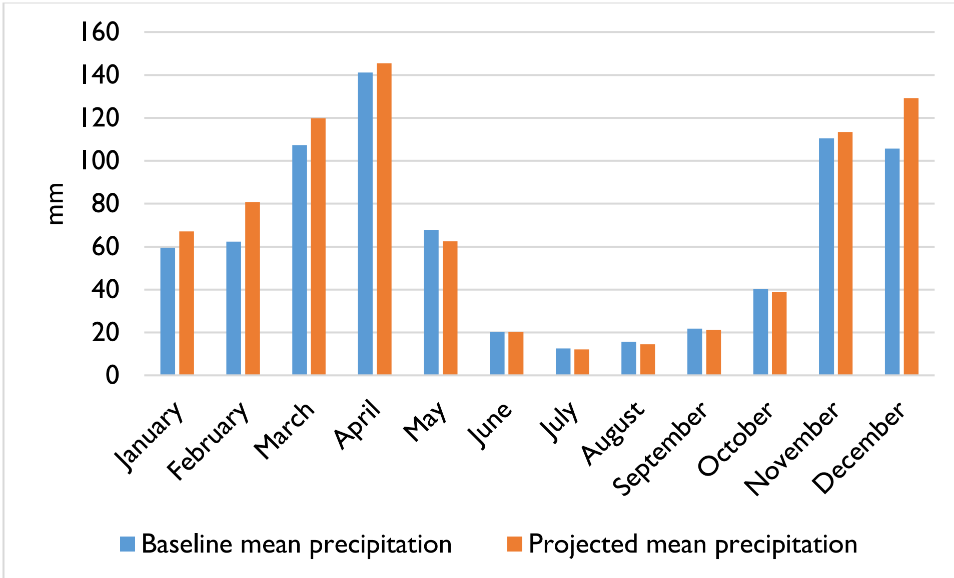


Figure 33. A comparison between historic and projected mean monthly precipitation (mm) for the Great East African Plains landscape

Source: Based on data from WorldClim Version2 and CMIP5

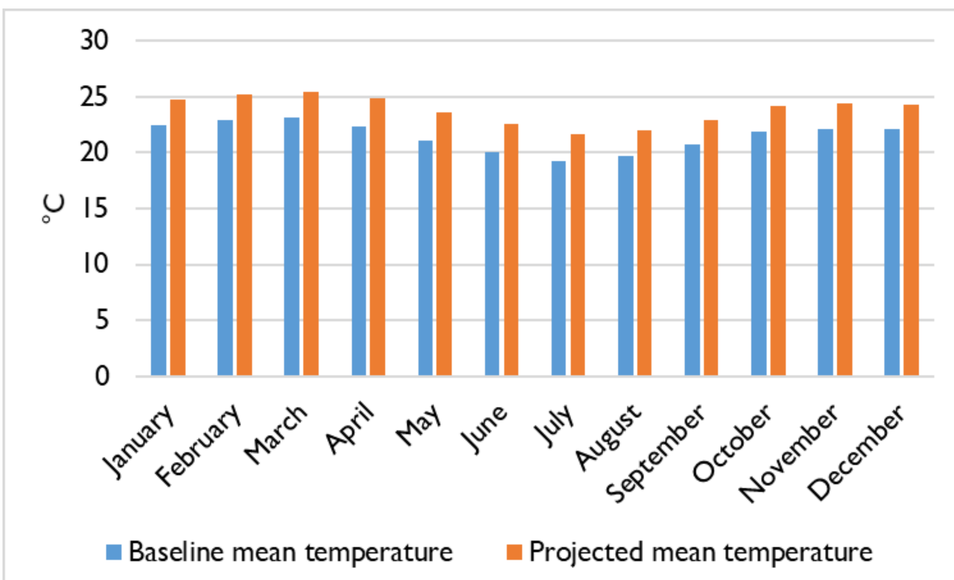


Figure 34. A comparison between historic and projected mean monthly temperature (°C) for the Great East African Plains landscape

Source: Based on data from WorldClim Version2 and CMIP5

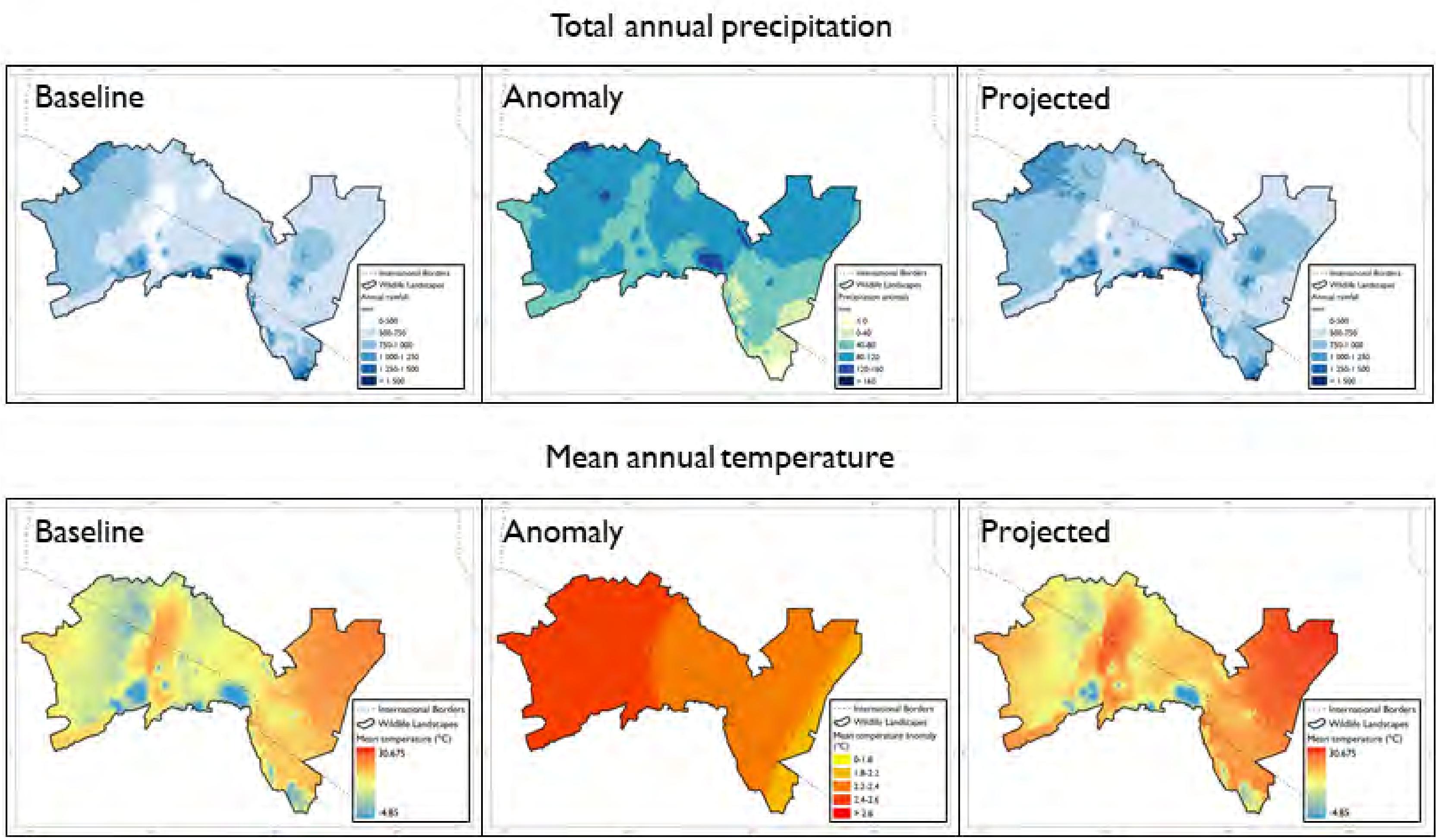


Figure 35. Baseline/historic (1960 – 1990) and projected (2040 – 2060) total annual precipitation (mm) and mean annual temperature (°C) across the Great East African Plains landscape

Source: Based on data from WorldClim Version2 and CMIP5

Table 23. Historic, projected, and percentage changes for mean annual temperature (°C) and total annual precipitation (mm) for key protected in the Great Eastern Plains wildlife landscape

	MEAN TEMPERATURE (°C)			MEAN PRECIPITATION (MM)		
PROTECTED AREA	HISTORIC ANNUAL AVG.	PROJECTED ANNUAL AVG.	CHANGE	HISTORIC ANNUAL TOTAL	PROJECTED ANNUAL TOTAL	% CHANGE
Tsavo East	25.3	27.6	2.2	723	792	9.5
Serengeti National Park	20.3	22.7	2.5	880	948	7.7
Ngorongoro Conservation Area	17.9	20.4	2.5	809	873	7.9
Tsavo West	23.1	25.4	2.3	690	735	6.5
Mkomazi National Park	23.3	25.5	2.2	733	760	3.7
Kilimanjaro National Park	11.7	14.1	2.3	1 320	1 411	6.9
South Kitui	25.7	28.0	2.3	633	720	13.6
Masai Mara	19.5	22.0	2.4	1 035	1 118	8.0
Maswa Kimali	20.7	23.3	2.5	856	917	7.1
Meru	24.5	26.9	2.4	525	596	13.6
Chyulu Hills	20.9	23.2	2.3	769	855	11.1
Amboseli	21.3	23.6	2.3	688	750	9.0
Ngai Ndethya	24.3	26.5	2.3	694	769	10.9
Arusha National Park	15.8	18.2	2.4	1 229	1 287	4.7

Source: Based on data from WorldClim Version2 and CMIP5. Protected areas are listed in descending order of area.

HUNTING PRESSURE ON WILDLIFE

Direct killing of wildlife for meat, trade, and in retaliation for human wildlife conflict will also likely remain a serious threat under a BAU scenario. Bushmeat represents a cheap, readily available protein source across much of the region, where it is often significantly less expensive than meat from domestic animals (Ndibalema & Songorwa, 2008; Mfunda & Røskaft, 2010; Rentsch & Damon, 2013). Hunting may also provide a valuable income source to rural households with limited livelihood opportunities (Loibooki *et al.*, 2002; Mfunda & Røskaft, 2010; Knapp *et al.*, 2015; Kyando *et al.*, 2019). Poverty is thus a major driver of bushmeat hunting and consumption. Furthermore, a preference for bushmeat exists in certain areas, such as the Western Serengeti region, where it is perceived as healthier and tastier than the meat of domestic animals (Ndibalema & Songorwa, 2008). Conversely, cultural taboos against bushmeat consumption limit hunting in parts of the landscape where Maasai pastoralists are the dominant ethnic group. According to community key informants in these areas, most harvesting is carried out by immigrant communities who have moved to the landscape and whose traditions and practices do not prohibit hunting. All else being equal, demand for bushmeat will continue to grow as populations increase, becoming increasingly unsustainable as it does. For example, in the Western Serengeti region, Rentsch & Packer (2014) estimated that 86,000-143,000 wildebeest are harvested each year for local meat consumption. They argue that current per capita consumption will not be sustainable in the future under present population growth rates (~3.5 percent). Indeed, hunting has been cited as the primary threat to the Greater Serengeti Ecosystem for some time (Sinclair, 1995). In the short-term, worsened poverty associated with the COVID-19 pandemic could increase reliance on bushmeat and other natural resources. Similarly, households may increasingly rely on bushmeat hunting in areas characterized by declining livestock productivity or poor crop yields (Knapp *et al.*, 2015). Key informants recommended increased law enforcement and more participatory wildlife conservation programs that involve local communities as potential measures to control unsustainable bushmeat harvesting. For example, a community key informant noted that bushmeat hunting is not a serious issue in Loliondo, where conservation practitioners work closely with communities on anti-poaching operations.

HUMAN WILDLIFE CONFLICT

Human wildlife conflict (HWC) is also a serious problem in much of the landscape. In villages adjacent to protected areas, over 50 percent of households are often affected by some form of HWC every year, ranging from crop raiding, to livestock predation, to attacks on humans resulting in injury or death (Okello, 2005; Schmitt, 2010; Mfunda & Røskaft, 2011; Osano *et al.*, 2013). This imposes a severe cost on people living alongside wildlife, and often drives resentment of protected areas and retaliatory killing of wildlife by local people. Key informants report that HWC is increasing in the landscape, yet communities have limited capacity to deal with it. Generally, community informants expressed higher tolerance for herbivore species, though even these can have a negative impact through crop raiding. However, community key informants said that they do not tolerate lions predating on livestock and would be happy for hyena to disappear from the area. These community perceptions reveal that HWC poses a particularly serious threat to predator species in the landscape. Due to the negative impacts of HWC, some informants reported that community interest in wildlife is gradually eroding. Some key informants also expressed resentment toward conservation authorities, who they felt did not respond adequately to instances of HWC when they occur, but will show up to prosecute communities when they take it into their own hands to deal with problem animals. The spread of diseases from wildlife to livestock was another issue mentioned by some key informants. For example, wildebeest calving in livestock grazing areas can spread malignant catarrhal fever, which is deadly for cattle. According to studies from Kenya, cattle owners can lose up to 10 percent of their herd each year to the disease (Orono *et al.*, 2019), thus representing a serious cost to households in a region where livestock are a central component of livelihoods. Key informants also reported wildlife were problematic for spreading ticks to cattle, in turn passing on various tick-borne diseases.

HWC is likely to worsen as settlement, livestock, and cultivation continue to expand into and around remaining natural habitats, increasing competition between wildlife and people for land and resources. As noted above, key informants also reported that the loss of grazing areas due to invasive alien species has also increased HWC as wildlife and livestock increasingly come into contact in uninvaded areas. This will worsen if invasive species continue to spread at current rates. Overall, the predicted increases in HWC in the future could lead to decreased tolerance of wildlife and more retaliatory killing of problem wildlife species, driving further declines in wildlife populations.

PROJECTED IMPACTS OF CLIMATE CHANGE ON WILDLIFE

The effects of habitat loss, especially the loss of dispersal and migration corridors, could also be exacerbated by climate change. Historically, wildlife has responded to both short-term climate stresses (*e.g.*, droughts) and longer-term climate changes through migration and range shifts (Malcolm *et al.*, 2002). However, this adaptation strategy is impeded today by the extensive loss of natural habitats and landscape connectivity from human activities, particularly for large and/or less mobile species. With substantial range shifts predicted for many species, averting further loss of migration and dispersal routes will become even more crucial for securing the long-term future of wildlife populations (Mawdsley, O'Malley & Ojima, 2009; Ogutu *et al.*, 2014). It has already been suggested that the Amboseli wildebeest population would have collapsed in the most recent severe drought if the connection with Tsavo populations were no longer present (Ogutu *et al.*, 2014). With evidence that rainfall is already becoming more unreliable and drought frequency increasing (Williams & Funk, 2011; Funk, 2012), the ongoing erosion of this migration route under business as usual could mean populations of wildebeest and other species in Amboseli might not recover from future drought and climate change.

Based on analysis of existing SDM outputs for more than 1,000 species, the expected combined species richness of mammals, birds, reptiles, and amphibians is shown in Figure 36. The maps indicate species richness under current conditions, and under the projections of three different climate models for 2070 (models ac, bc, and cc), which show the range in results depending on which future climate model one uses. The term "expected" species richness is used because, in reality, species ranges have been altered by anthropogenic land use and other pressures, meaning that real species richness will be substantially lower in regions where natural habitats have been transformed. For example, the models predicted high potential species richness in the Kenyan highlands to the north of the wildlife landscape, which suggests this region likely had high species richness in the past. However, extensive habitat transformation means that species richness is actually much lower there now.

Areas of highest expected species richness within the Great East African Plains landscape currently are generally associated with wetter and/or higher-lying areas. These include the Serengeti, Masai Mara, and Ngorongoro Conservation Areas in the northwest, as well as areas around Mount Kilimanjaro and the Usambara Mountains in the central and southeastern regions of the landscape, respectively. By 2070, species richness is projected to decline across the landscape, except for around South Kitui and Tsavo East National Parks in Kenya, and the elevated areas in Tanzania (Mount Kilimanjaro, Ngorongoro, and the Usambara Mountains). This pattern is also reflected when species richness is broken down into the broad taxonomic groupings (birds, mammals, etc.) of animals (see Appendix 5).

In addition, species distribution models predict substantial contraction of areas with suitable climatic conditions for most key charismatic wildlife species, including lion, elephant, and wildebeest (Figure 37; also see Appendix 5). In general, areas that already experience high temperatures are predicted to lose at least some charismatic species as conditions become even hotter. Affected regions include the eastern parts around Tsavo, as well as the Rift Valley. On the other hand, the relatively cooler, higher-lying parts of the wildlife landscape, such as Serengeti-Mara, are predicted to remain suitable for most large wildlife species. These cooler regions could become increasingly important climate refugia for wildlife in the future.

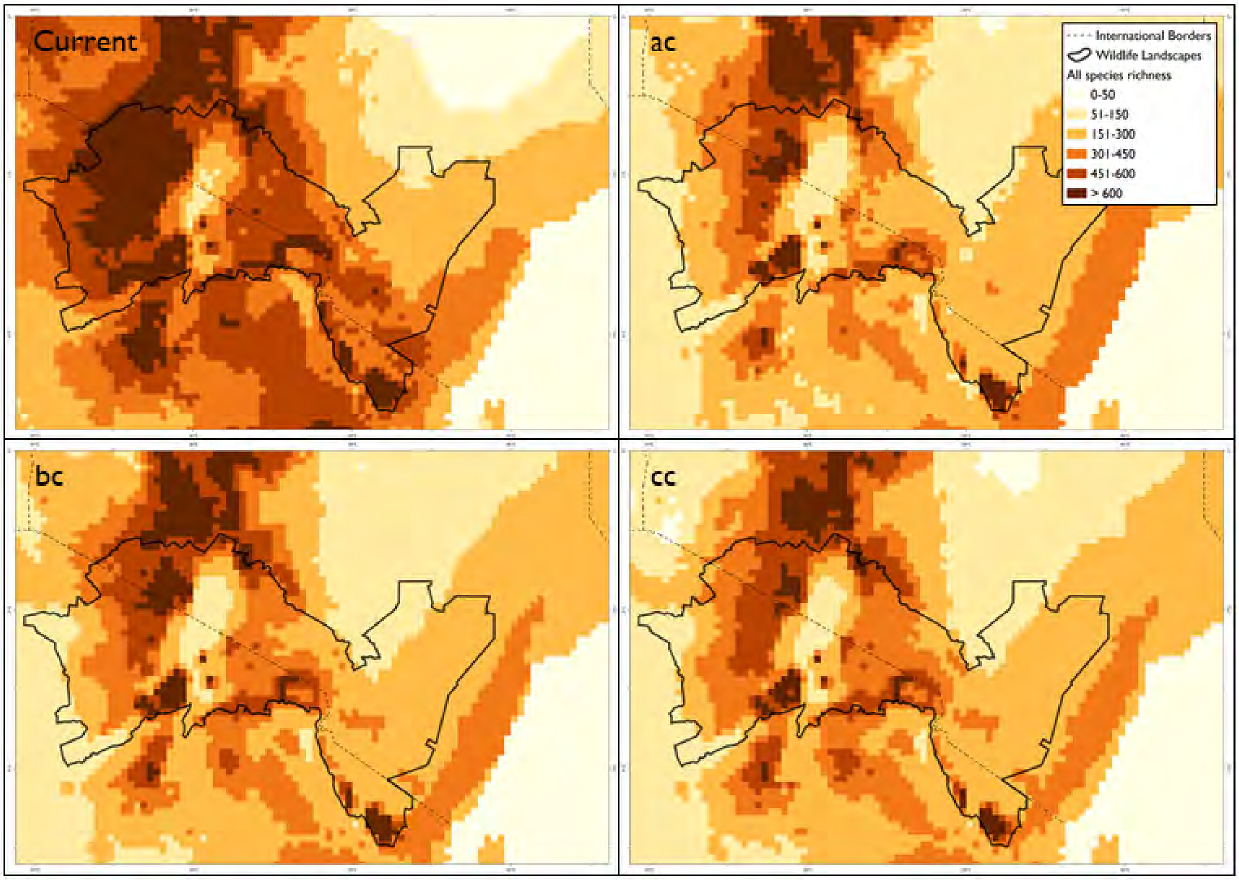


Figure 36. Current geographic variation in species richness (amphibians, birds, mammals, and reptiles) for the Great East African Plains landscape, followed by the projected species richness pattern for each of three climate models used

Source: Based on modelled species distributions from Conservation International

Notwithstanding the inherent inaccuracy in the modelled data, these results suggest that climate change alone could result in significant declines and local extirpation of many species from the landscape. This includes the possibility of climate-induced declines and disappearances of charismatic species from parts of the landscape, which would have serious negative impacts on wildlife tourism in affected regions. The potentially severe risk climate change alone presents to wildlife provides further reason to mitigate other pressures on wildlife in the landscape. In particular, the substantial range shifts predicted under climate change further highlight the importance of maintaining the remaining migration corridors and dispersal routes, as these are crucial to facilitating the movement of wildlife in response to climate stresses. Furthermore, climate change impacts could accentuate other pressures such as rangeland degradation by livestock and HWC. As noted by key informants, increased drought attributed to climate change has resulted in decreased pasture quality and water availability, increasing competition for remaining resources between wildlife on the one hand and livestock and people on the other.

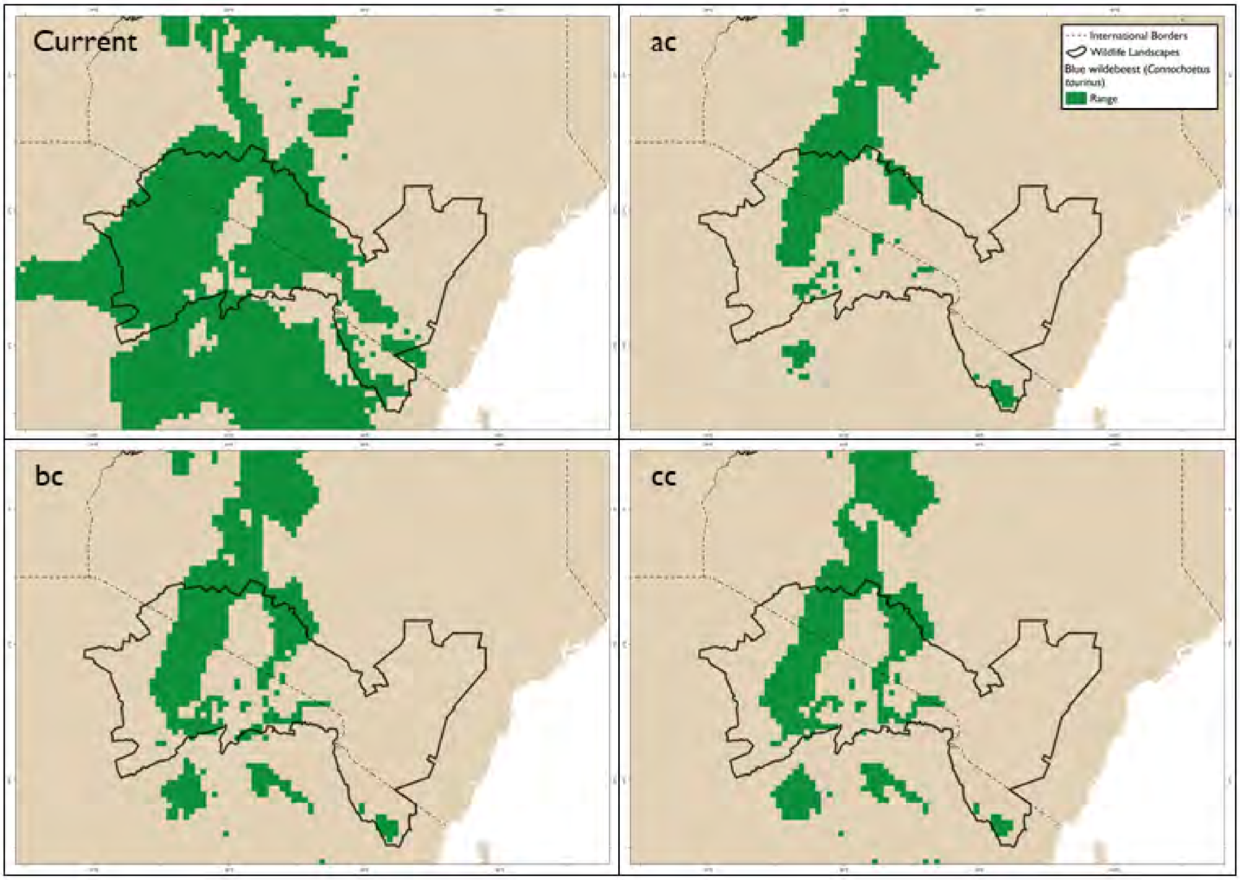


*Figure 37. Current habitat suitability of blue wildebeest (*Connochaetes teurinus*) for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)*

Source: Based on modeled species distributions from Conservation International

OVERALL AFFECT ON WILDLIFE POPULATIONS

Severe declines in wildlife populations have already resulted from the combined effects of these pressures, particularly in the Kenyan portion of the landscape. Populations of most wildlife species are less than half of what they were in the 1970s over the Kenyan parts of the wildlife landscape, where regular aerial counts have been conducted throughout this period (Ogutu *et al.*, 2016). Less comprehensive count data was available for the Tanzanian portion of the wildlife landscape, where broad-scale wildlife monitoring is limited to the Serengeti (Caro & Davenport, 2016). However, there is evidence that populations of most wildlife species have declined here too, particularly in protected area tiers with less enforcement (*e.g.*, game reserves and wildlife management areas) (Stoner *et al.*, 2007). Given the aforementioned threats, wildlife populations may decline further under business as usual. While wildlife populations may remain more stable in protected areas in the short-term, protected areas could increasingly become isolated sanctuaries in a sea of agriculture, with little landscape or genetic connectivity between them (Caro & Davenport, 2016). Hence, the long-term future of wildlife populations in the landscape remains uncertain under business as usual, even in relatively well-managed protected areas.

PROJECTED CLIMATE CHANGE IMPACTS ON SUITABILITY FOR CROPS

Six widely grown crop species were modelled using FAO's EcoCrop analytical tool. In general, suitability of the landscape for cultivation of various crops species was predicted to increase by 2050, relative to current conditions. Given the ongoing shifts toward greater reliance on cultivation over much of the landscape, improved growing conditions in the future will likely encourage further expansion of cultivation under business as usual. Current and predicted suitability for the six crop species is described individually below, while suitability maps can be found in Figure 38. Suitability is described in terms of the suitable area for a given crop (i.e., the region with a suitability score of greater than 0), as well as the relative suitability value, which ranges from 0 (unsuitable) to 1 (optimum conditions).

Table 24. Summary of the expected changes in the suitable area and suitability scores for crops in the Great East African Plains landscape and immediate surrounds

CROP	PRESENT	FUTURE
Beans	Suitability widespread, except for the very high-altitude areas and dry Rift Valley regions. Suitability is highest in the Serengeti-Mara and Kilimanjaro regions.	Suitable area increases to cover the whole landscape. Suitability scores predicted to increase across most of the landscape, with the exception of the eastern regions.
Cassava	Suitable conditions across most of the western and eastern parts of the landscape, though suitability scores are often relatively low. Largest area of high suitability in the Mara and northwest Serengeti region.	Suitable area increases to encompass most of the landscape. Suitability scores predicted to increase across the landscape, with most of the Serengeti-Mara region becoming moderately to highly suitable.
Maize	Suitable area mostly limited to the western portion of the landscape. Other areas of suitability in the central and eastern parts of the landscape associated with higher-lying regions (*e.g.,* Kilimanjaro, Eastern Arc Mountains, Taita Hills).	Suitability scores increase over most of the western part of the landscape, though suitable area does not change much here. Suitable areas in the central and western parts of the landscape expands downslope, accompanied by a general increase in suitability scores.
Millet	Suitable area widespread across the landscape aside from high-lying, wet areas. Suitability highest in the western and eastern regions of the landscape.	Little change in suitable area, but suitability scores generally increase, especially in the central and eastern parts of the landscape. Conversely, some declines in suitability occur in the western portion of the landscape.
Potato	Suitable area widespread across the landscape aside from the highest parts of Kilimanjaro. Suitability highest in the western portion of the landscape, and in higher-lying regions elsewhere (*e.g.,* lower parts of Kilimanjaro, Eastern Arc Mountains).	Little change in suitable area. Suitability scores similar in the western region of the landscape, Increased suitability in the central regions, while suitability generally declines in the eastern part of the landscape.
Sorghum	Most of the landscape suitable, aside from highest slopes of Kilimanjaro and parts of the Rift Valley region. Suitability highest in the eastern part of the landscape.	Suitable area increases to encompass all of the landscape aside from the upper slopes of Kilimanjaro. Suitability scores increase throughout the landscape, with much of the northeast portion becoming highly suitable.

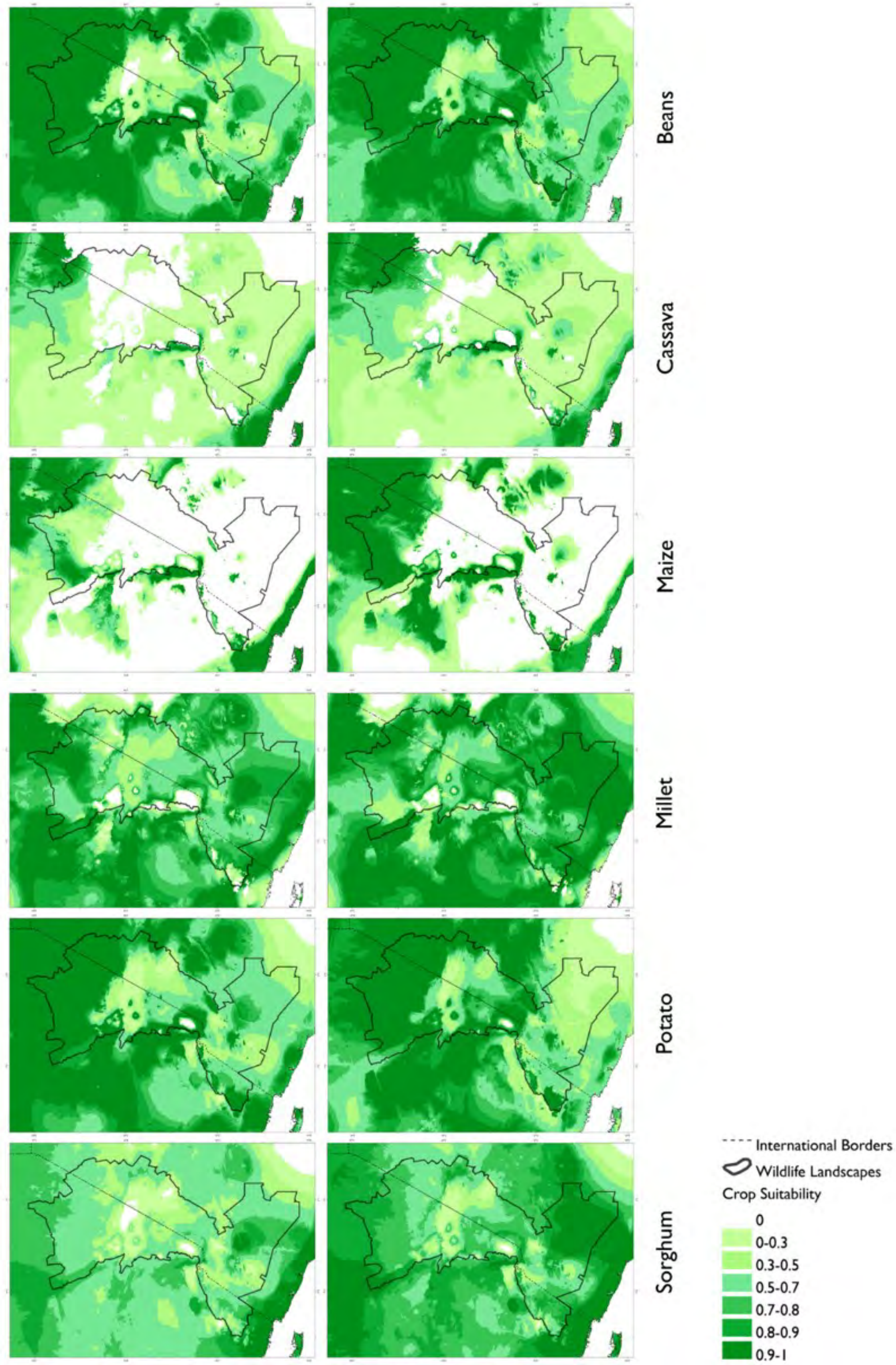


Figure 38. Estimated present and future suitability for some of the key crops grown in and around the Great East African Plains landscape. Model outputs generated using the FAO Ecocrop database and model and climate projections for 2040-60.

POTENTIAL IMPLICATIONS OF A BUSINESS-AS-USUAL SCENARIO

This section provides an integrated, qualitative assessment of the impacts of a business-as-usual scenario on wildlife, ecosystem services and human wellbeing over the period from the baseline (2018) to 2030. The combination of 1) increasing population and demand for land and resources and 2) the impacts of climate change on habitats, species, and agriculture need to be considered. There is a great deal of uncertainty in this. Notwithstanding this caveat, the pressures on wildlife and wildlife habitats are expected to change as follows.

Habitat transformation could continue to fragment wildlife landscapes. Subdivision of land, government policy, and the desire to diversify income sources will likely lead to further expansion of cultivation, settlement, and fencing across the region. The growth of cultivation may be further encouraged under future climate change in areas where conditions become more favorable for agriculture. This will erode remaining migration routes and dispersal areas outside protected areas, and increase conversion pressures within protected areas, especially in areas with dense populations and little land availability. The net result will be increased genetic isolation of wildlife populations in protected areas, and reduced ability for wildlife to migrate in response to threats like drought. Overall, this threatens the long-term viability of wildlife populations across the region. Based on changes in cultivated area from 2015 to 2018 derived from the 100 m Copernicus land cover, it was predicted that the area of **cultivation could increase from 9.4 percent of the landscape in 2018 to 28.5 percent in 2050** under a BAU scenario. If this were to occur, a further 2.4 million ha of habitat would be converted to agriculture by 2050. This would signify a substantial loss of habitat for wildlife, as well as degrading the value of several of the ecosystem services provided by the landscape, as described further below.

The rate of habitat degradation could increase exponentially. Ongoing population growth will worsen land scarcity in the region. Rangeland health could be reduced further as livestock numbers rise, compromising important foraging resources for both wildlife and people. Based on past trends, it was predicted that **livestock could increase by up to 65 percent in the Kenyan portion of the landscape, and 93 percent in the Tanzanian portion by 2050** under a BAU scenario. Worsening rangeland condition outside protected areas will increase livestock encroachment into protected areas. This could be heightened by an increase in rainfall variability and drought under climate change. Woody vegetation across the landscape will likely come under increased pressure, due to both growing rural demand for fuel and fencing, as well as exponentially increasing urban charcoal demand. Based on population growth estimates, it was predicted that **demand for woody resources could increase by around 65 percent by 2050** under a BAU scenario.

Tolerance for wildlife and conservation is likely to decrease. The combination of human encroachment into the wildlife landscape and continuing increases in rainfall variability and drought could lead to more intense human wildlife conflict in the future. Predation of livestock tends to increase during droughts, while herbivorous wildlife may increasingly resort to foraging in fields and around villages as alternative food sources disappear (Ogutu *et al.*, 2014). This is likely to lead to efforts to kill problem animals. HWC will likely continue to worsen as human populations, livestock, and cultivation increase alongside remaining wildlife populations. Where local communities have historically gained some benefits from wildlife tourism, a decline in these could also increase resentment of protected areas and wildlife, especially where HWC and opportunity costs of lost cultivation and grazing land are high. This was corroborated by a community key informant, who noted that carnivores and certain herbivore species like elephant and buffalo were only tolerated by communities due to the tourism benefits they provide.

Poaching could have a significant impact on wildlife populations. Poaching will increase as a result of reduced opportunities for income from crops, livestock, and tourism, and this impact will be severely exacerbated in the short to medium term by the impacts of the COVID-19 pandemic on tourism and the economy in general. In spite of possible increases in crop suitability, crop and livestock income per household will likely be reduced by 1) increased droughts and 2) increased competition for land. Crop failures and livestock deaths increase people's reliance on bushmeat and other natural resources during and after droughts. Wildlife may also become easier to hunt, as animals become weakened and more vulnerable, or move closer to villages in search of food (Ogutu *et al*., 2014; Knapp *et al*., 2015). Together, these factors would increase stress on wildlife already struggling to cope with future climate change and variability. Based on population growth alone, it was estimated that **demand for bushmeat could increase by 58 percent across the landscape by 2050** under a BAU scenario. Given the other pressures mentioned, bushmeat demand in reality could increase beyond this estimate.

The potential overall effects of the above pressures on wildlife and wildlife habitats on ecosystem services under a BAU scenario can be summarized as follows (see Table 25).

Nature-based tourism value has declined significantly due to COVID-19, and recovery will be threatened by wildlife losses and climate change. The pandemic has caused a huge drop in tourism due to the restriction of international travel, reducing the financial viability of protected areas in the short-term. It is predicted that tourism would increase once the COVID-19 pandemic eases. However, in the longer term, **tourism was predicted to plateau and then decline by 2050** relative to its 2018 value, under a BAU scenario. An **annual tourism loss of US$76 million was predicted for the Kenyan portion of the landscape (15 percent of current value) and US$85 million for Tanzania (12 percent of current value)**. Unchecked declines in wildlife populations under a BAU scenario are one cause of this predicted decline. The degradation and loss of habitat and wilderness value as human populations grow, cultivation expands, and livestock encroachment intensifies could also deter tourists. This was corroborated by a private sector key informant who expressed skepticism about the future of nature tourism investments in certain Tanzanian WMAs due to the expansion of agriculture and settlement and increasing livestock populations. Prior to COVID-19, tourism continued to rise despite these pressures. However, it was hypothesized that declining wildlife numbers and ongoing habitat change will eventually reach a threshold value under a BAU scenario, beyond which they will start to have a negative impact on visitor numbers as wildlife tourism starts switching to other areas.

Water availability in the dry season is expected to decrease. Further woody cover loss, rangeland denudation, cultivation, and settlement will result in lower dry season baseflows. Abstraction of water from aquifers, springs, and lakes is also likely to increase due to expansion of cultivation, contributing to overall flow reduction. It was predicted that **baseflow could decline by 21.2 percent by 2050** under a BAU scenario, primarily due to the high water requirements of the expanded areas of cultivation. This represents a loss in baseflow of 3,156 million m^3 relative to the current landscape, with an **annual replacement cost of US$352 million.** Baseflow loss could be even greater than this if there is a significant expansion of irrigation schemes, as the model assumed a substantial portion of the newly cultivated areas would be rainfed. These predicted declines in baseflow would be particularly serious for Kenya, as the country already has a relatively high and increasing water stress index (FAO, 2016). This would have a negative impact on rural livelihoods as well as water-dependent private sector enterprises like the flower industry, who would need to compete with other sectors for increasingly scarce water resources. Further degradation of the landscape's capacity to retain and release rainfall inputs could thus have serious social and economic impacts.

Freshwater systems are expected to become more polluted. Water quality will be compromised by the conversion of natural habitats to cropland, which will substantially increase nutrient loads entering watercourses. This will result in increased nutrient loads and eutrophication of rivers, wetlands, and lakes, reducing the value of those habitats. For the portion of the landscape that drains into Lake Victoria, it was estimated that **phosphorus export would increase by a factor of 2.68 by 2050** in a BAU scenario. The additional 1.1 tons of phosphorus exported each year would have an **annual treatment cost of US$558,000**.

Erosion and sedimentation are expected to increase. The conversion of natural vegetation cover to cropland, and denudation of vegetation cover livestock overgrazing and trampling and the collection of woody biomass will increase the rate of soil loss, causing further loss of value from downstream aquatic ecosystems, declining water quality, and worsening sedimentation of dams. Using our predicted rate of agricultural expansion, we estimated that the **capacity of the landscape to retain sediment and control erosion could decrease by 9.2 percent by 2050** under a BAU scenario, with an additional 166 million tons of sediment entering rivers and waterbodies. This would amount to an **annual US$204 million loss** in value of the soil erosion control service provided by the landscape, based on the greater need for sediment clearance and increased loss of reservoir storage capacity.

The landscape is expected to contribute to further local and global climate change through net carbon emissions. Aboveground storage will decrease due to harvesting of wood for fuel and building materials, and by the conversion of woody habitats to cropland, exacerbated by increasing urban growth in regional centers. Overgrazing and habitat conversion will also compromise belowground carbon storage. It was estimated that **5.1 percent (235.4 MtC) of the carbon stored in the landscape could be released by 2050** under a BAU scenario, representing a **loss of US$40 million** relative to the current value of the carbon storage service.

The landscape's capacity to support agricultural livelihoods could be compromised, affecting the ecological integrity of protected areas. The provision of services such as crop pollination will decrease as cultivation expands further and becomes more intensive, with a reduction or disappearance of natural vegetation patches between fields. In addition, forage for livestock production is likely to remain poor or decline further in already degraded areas due to the persistence of high stocking rates across much of the landscape. Elsewhere, rangeland degradation may increase as expanding human populations, subdivision, and cultivation continue to reduce available land for livestock. This will lead to further incursion of livestock into protected areas, already a significant problem in this landscape. Stocks of harvested resources will decline outside of protected areas due to growing demand from increasing populations. Woody resources, in particular, are under threat from increasing urban charcoal demand. Shortages outside protected areas may also increase harvesting pressures within protected areas, especially in densely populated regions.

Table 25. Estimated changes in the value of ecosystem services and water treatment costs by 2050 caused by land use changes under a BAU scenario for the Great East African Plains. For services with a global value, both total value to the world and value to the East African region only are shown (latter value in parentheses).

ECOSYSTEM SERVICE	CURRENT VALUE (US$)	2050 VALUE (BAU) (US$)	% CHANGE
Nature-based tourism	2,758.8m (1215.0m)	2,391.7m (1053.3m)	-13.3
Biodiversity existence	5,373.5m (1.5m)	4,222.2m (1.2m)	-21.4
Flow regulation	1,002.9m	650.6m	-35.1
Erosion control	2,226.7m	2,023.0m	-9.2
Carbon storage	397.9b (787.9m)	376.8b (747.6m)	-5.1
Water treatment costs	481.5k	640.8k	+33.1

THE NORTHERN SAVANNAS

FEATURES AND LOCAL CONTEXT

WILDLIFE AND WILDLIFE HABITAT

The Northern Savannas study area that encompasses a largely semi-arid region in northeastern Uganda and southern South Sudan (Figure 39) is a rugged savanna landscape where grasses are dotted with iconic tree species such as red thorn acacias and desert dates, and sausage trees and fan palms are found along important perennial waterways. In the northern part of the study region, more than 86 mammal species can be found including leopard, cheetah, wild dog, and elephant, as well as more than 500 bird species. Wildlife moves freely across national borders in this region. A number of wildlife corridors provide cross-border connectivity between Uganda and Kenya. In the north of the study region, a large migration corridor exists between Kidepo Game Reserve in South Sudan and the adjacent Kidepo Valley National Park in Uganda.

In this landscape, the terrain rises gradually from the western parts to the Uganda/Kenya border. There is an escarpment area along much of the Uganda/Kenya border region that also extends a short distance into southeast South Sudan. Beyond this escarpment, elevation drops sharply to the plains of northwest Kenya and southeast South Sudan, where terrain is flat for the most part. Further south, Mount Elgon is the dominant relief feature, rising to 4,321 m. While terrain is generally flat in the western part of the Northern Savannas landscape, the Imatong Mountains are a prominent relief feature along the Uganda/South Sudan border in the northwest, rising to 3,172 m at their highest point.

Rainfall declines from west to east. This is mirrored by a transition from savanna in the west of the study area to *Acacia-Commiphora* bushland in the drier eastern parts. Across this rainfall gradient, these woody vegetation types are interspersed with areas of grassland. Areas of Afromontane forest also occur at wet, higher elevations. Notable forested areas include the Imatong Mountains, mountainous terrain in the far northeast of Uganda and southeast of South Sudan, and Mount Elgon along the Uganda/Kenya border in the far south of the study region. Ericaceous and Afroalpine vegetation are also found on the upper slopes of Mount Elgon, while montane bamboo occurs on the mountain's middle slopes.

Much of the study area has suffered from prolonged violence and instability. This includes civil wars in South Sudan and violent raids and cattle rustling by armed groups in northeast Uganda and northwest Kenya (Jabs, 2007; Gorsevski, Geores & Kasischke, 2013). This has hampered research in the region, which remains limited (Egeru *et al.*, 2014c). This is particularly the case for South Sudan where, since the start of the unrest in 2013, little research has been carried out within the wildlife landscapes of the far south of the country. Available literature suggests pastoralism was traditionally the primary livelihood over much of the region, although the degree of emphasis on livestock versus cultivation varies among ethnic groups (Stites *et al.*, 2007). Over recent decades, there has been an overall shift toward agro-pastoralism and pure cultivation (Burns, Bekele & Akabwai, 2013; Egeru *et al.*, 2014c). However, concerns have been raised around the suitability of crop production given the unpredictability of rainfall and frequency of drought over much of the region. Violence and raiding has also contributed to sedentarization and increased cultivation by traditionally mobile pastoralists in areas of northeast Uganda, while alternative livelihood strategies like gold mining, brick making, and charcoal production have also expanded (Macopiyo, 2011; Egeru *et al.*, 2014c). These activities signify increasing pressure on natural resources in the region, where notable declines in woodland have occurred (Egeru *et al.*, 2014c). Egeru *et al.* (2014) also report an expansion of bushland into grassland, indicating that poor grazing practices have resulted in bush encroachment in some areas.

Protected areas themselves are not immune to this pressure (Government of Uganda, 2013). For example, mining and high livestock pressure has led to degradation of Uganda's Matheniko Game Reserve (Bintoora, 2016). Much of this is driven by transboundary Turkana pastoralists from western Kenya in search of water and forage during dry periods. Overall, instability and poor accessibility have hampered tourist visitation of the region's protected areas. South Sudan's wildlife areas have been decimated by decades of civil war. Political instability coupled with the devaluation of the South Sudanese Pound has resulted in a complete breakdown of law enforcement and management. There are currently no park rangers on the ground in Kidepo Game Reserve in South Sudan, and most wildlife that was there has likely been hunted or has moved into the neighboring Kidepo Valley National Park in Uganda, safer from poaching. This was corroborated by community key informants in South Sudan who indicated they no longer saw species like elephant. As the country gradually returns to peace, there is an opportunity to reestablish effective management in these wildlife areas.

PROTECTED AREAS AND NATURAL RESOURCE MANAGEMENT

Uganda's protected areas cover national parks, wildlife reserves, wildlife sanctuaries, community wildlife management areas (CWMAs) and forest reserves (Figure 39). The central forest reserves (CFRs) are managed by the National Forestry Authority (NFA) and the local forest reserves by local governments and communities. The wildlife protected area estate is managed by Uganda Wildlife Authority (UWA), including national parks and wildlife reserves. CFRs found in this study region include Kadam, Moroto, and Napak. The NFA has increasingly pursued collaborative management of these reserves with communities, which has reportedly improved community perceptions of forest conservation.

Kidepo Valley National Park is the main tourist attraction in this study region. However, visitor numbers are well below those in Uganda's more popular savanna national parks like Queen Elizabeth and Murchison Falls (Government of Uganda, 2013). The forested Mount Elgon National Park, in the south of this wildlife landscape and on the border with Kenya, is also a popular tourist attraction. Wildlife reserves include Pian Upe, Matheniko, and Bokoro Corridor, and community wildlife areas include Amudat, Karenga, and Iriri.

In the Kenyan portion of the landscape, Mount Elgon National Park is managed by the Kenya Wildlife Service (KWS), while the Mount Elgon Forest Reserve is managed by the Kenya Forest Service (KFS). Wildlife can move freely between the Kenyan and Ugandan portions of Mount Elgon. Nasolot National Reserve, located to the northeast of Mount Elgon, is also managed by KWS. Livestock grazing is permitted in these forest reserves and national reserves.

In South Sudan, development of legislation, policy and strategies for protected areas is undertaken by the Ministry of Wildlife Conservation and Tourism. However, management presence is lacking in the South Sudanian portion of the landscape as a result of the war and ongoing instability. The dominant protected areas, their management, and defining features are described in Table 26.

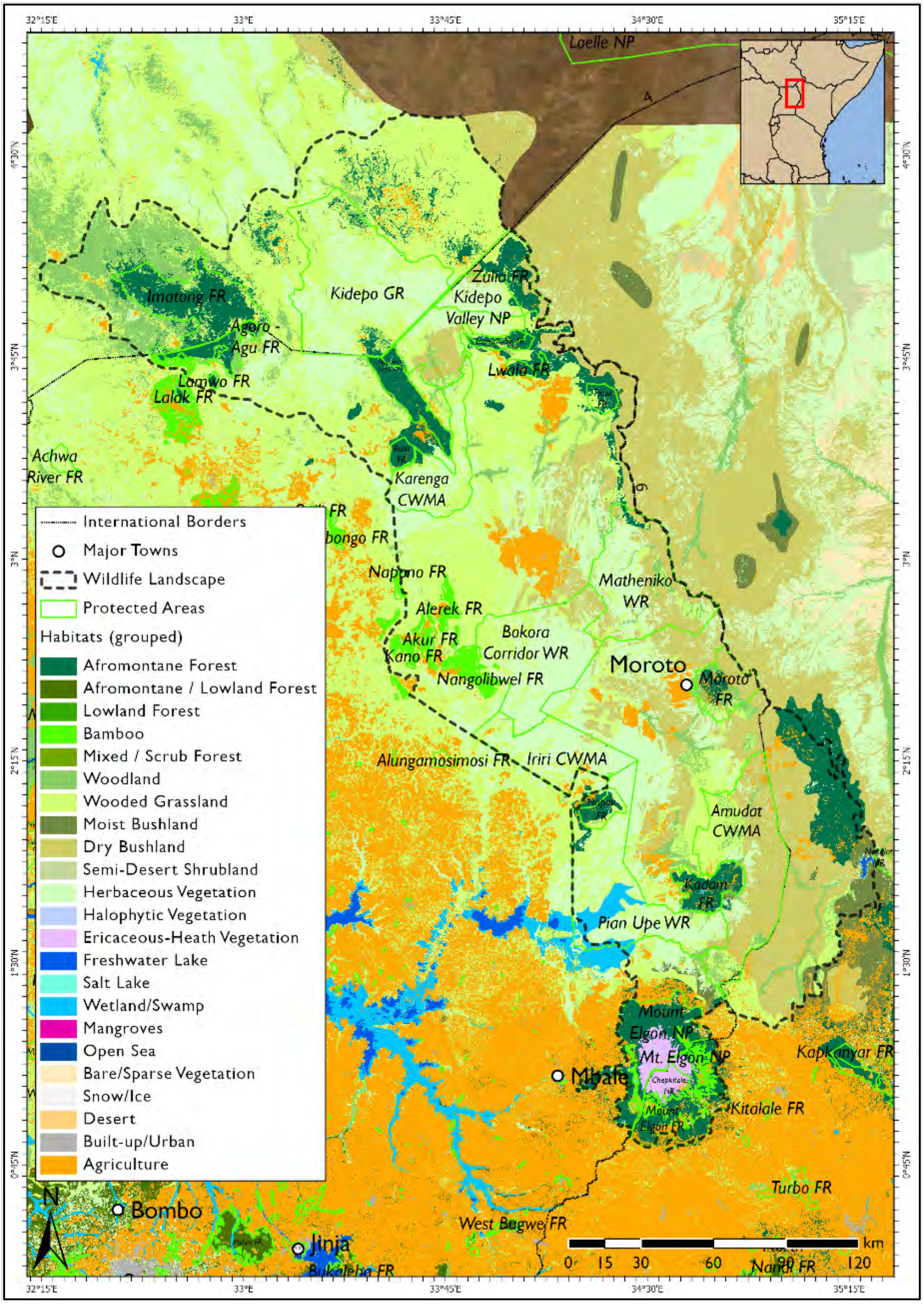


Figure 39. Land use and habitat types and protected areas of the Northern Savannas wildlife landscape (Kenya – Uganda border coincides with protected area boundary between Mount Elgon and Moroto Forest Reserve, and the wildlife landscape boundary from Moroto town up to the South Sudan border)

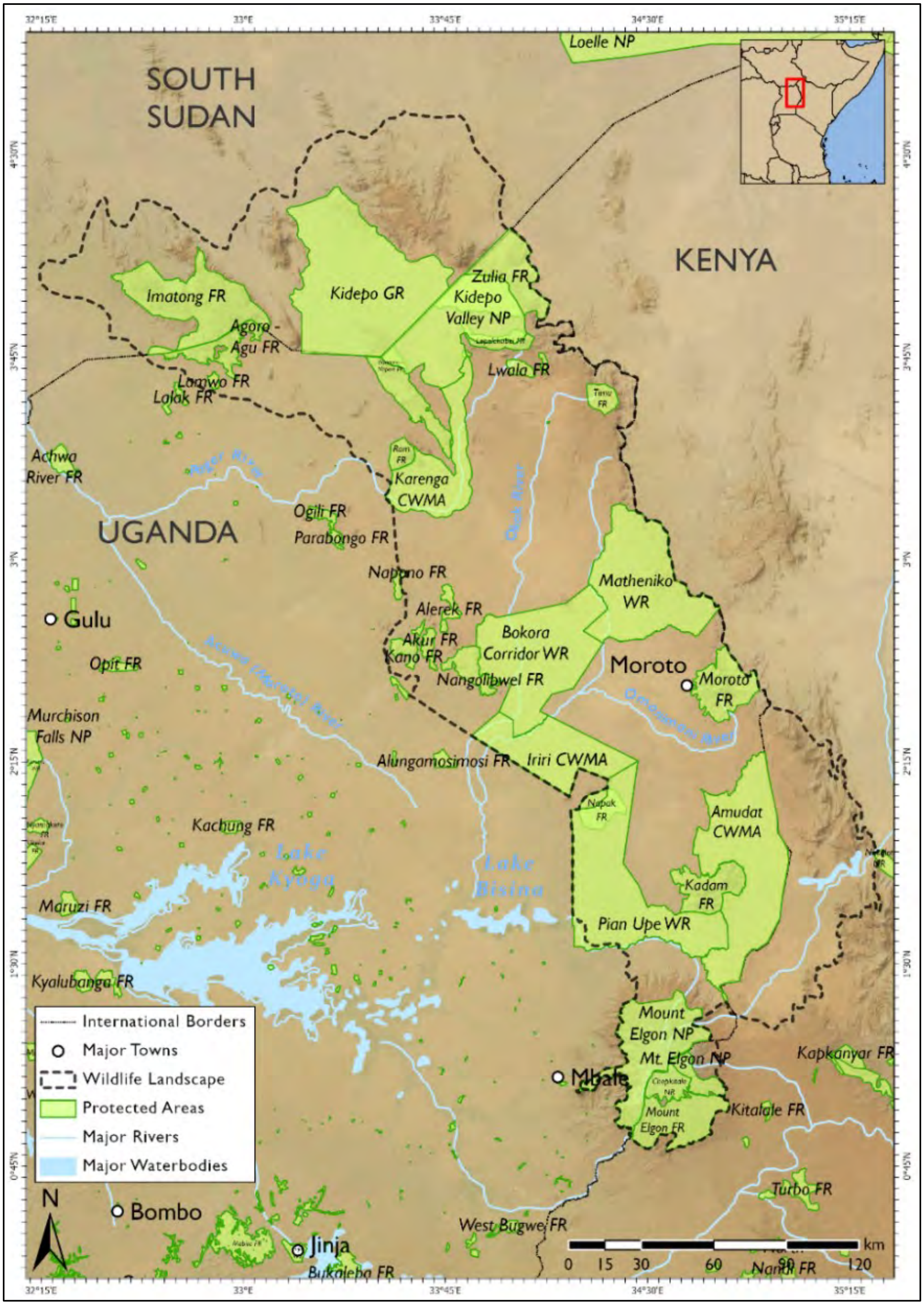


Figure 40. Protected areas of the Northern Savannas wildlife landscape

Table 26. The dominant protected areas within the Northern Savannas landscape and their defining features

PROTECTED AREA	SIZE (HA)	MANAGEMENT	DEFINING FEATURES
Imatong Central Forest Reserve	116,037	Established 1950, Ministry of Wildlife Conservation & Tourism	Lowland tropical forest. The highest peak in South Sudan is located within the reserve, Mount Kinyeti at 3,187 m. Twelve rivers originate in the reserve, which feed into the Bandingilo National Park. A few small antelope and several species of primates are found in the reserve as well as an estimated 500 species of birds.
Kidepo Game Reserve	285,783	Established 1975, Ministry of Wildlife Conservation & Tourism	A rugged savanna grassland and woodland landscape that borders onto Kidepo Valley National Park in Uganda. Mount Morungole at 2,750 m is a dominant feature and the reserve is transected by the Kidepo and Narus Rivers where groves of palm trees grow. Prior to the current conflict in South Sudan, the reserve was home to elephant, buffalo, lion, cheetah, and a variety of plains game. Today, only baboon and some small antelope species remain.
Kidepo Valley National Park	143,089	Established 1962, managed by UWA	Uganda's most isolated national park lies in the rugged, semi-arid valleys between the borders with Sudan and Kenya. Rated as one of Africa's finest wildernesses, it boasts of spectacular landscapes and large populations of mammals. The park is home to more than 77 mammal species including big game (elephant, lion, buffalo, leopard) as well as around 475 bird species. Rare species include wild dog, pangolin, aardwolf, caracal, and bat-eared fox, as well as several rare antelope and bird species.
Pian Upe Wildlife Reserve	214,996	Established 1964, managed by UWA	This game reserve comprises untouched grassland and wooded grassland. There are numerous charismatic mammal species such as lion, elephant, black rhino, and giraffe, and rare bird species. This reserve is part of the Mount Elgon Conservation Area.
Matheniko Wildlife Reserve	175,714	Established 1964, managed by UWA	Semi-arid savanna and shrubland dominate this reserve, which is connected to Pian Upe through the Bokoro Corridor. The reserve is remote and has a diverse array of mammal and bird species. Rare species include the Bright's gazelle, roan antelope, mountain reedbuck, Ugandan kob, Bohor reedbuck, topi, olive baboons, patas monkey, and cheetah.
Bokoro Corridor Wildlife Reserve	181,584	Established 1964, managed by UWA	Connected to Matheniko and Pian Upe Game Reserves, the Bokoro Corridor is comprised of savanna and shrubland with a variety of wildlife and bird species. This reserve is poorly accessible for tourists.
Nasolot National Reserve	92,000	Established in 1979, managed by KWS	Remote reserve consisting of plains and rugged hills, with spectacular views from Nasolot Hill. Holds a diverse wildlife population including large herds of elephant and significant populations of cheetah and pangolin. However, visitor numbers are low as the reserve is not well known and has no accommodation facilities.
Mount Elgon National Park (Uganda and Kenya)	92,915 (Uganda) 15,800 (Kenya)	Established 1968 in Kenya managed by KWS, and in 1992 in Uganda managed by UWA	An extinct volcano and the oldest and largest solitary mountain in East Africa, the park is home to more than 300 species of birds, including the endangered Lammergeyer. The higher slopes are protected by national parks in Uganda and Kenya, creating an extensive transboundary conservation area, which has been declared a UNESCO Man & Biosphere Reserve. Diverse terrain and range of altitudes and rainfall create four distinct vegetation zones: lush montane forest, mixed bamboo, dense scrub, and open moorland grassland.

Source: UWA, KWS

Despite cross-border movement of wildlife among the three countries that constitute the wildlife landscape, key informants note that there are no cross-border conservation agreements between Kenya, South Sudan, and Uganda that would allow for protected area staff from these three countries to work together. This was noted as a major gap for controlling illegal activities like poaching. It can also result in different and potentially conflicting protected area management policies. As a result, one key informant emphasized the need for a shared management plan to harmonize conservation of Mount Elgon. Nevertheless, there have reportedly been discussions and staff exchange tours between the KFS and UWA, suggesting that transboundary collaboration in conservation between Kenya and Uganda is improving.

To complement state protected areas in the landscape, there have been efforts to pursue community-based natural resource management in Kenya and Uganda in the form of community conservancies. Key informants from both Kenya and Uganda also noted that community scouts make a valuable contribution to reducing poaching in the landscape. Community scouts work directly with the KWS in the Mount Elgon area. A number of conservancies have been established in northwest Kenya. This includes Pellow Conservancy, which forms a large buffer around the Nasolot National Reserve. Conservancies in this part of Kenya have reportedly improved the interaction between communities and wildlife in the region and provide an alternative source of income to livestock. According to key informants, conservancies in Kenya have also contributed to reducing inter-ethnic conflicts through acting as a geographic buffer between clashing ethnic groups in the region. Through providing an income source, they have also reportedly decreased criminal activities such as cattle rustling and highway robberies as well as reducing poaching. However, the conservancies in the area have recently become a source of controversy. Because they were promulgated by Tullow Oil, suspicions have arisen that they will be used as a tool to gain access to oil reserves. Community conservancies are less well established in Uganda, where the government is currently in the process of developing community conservancy guidelines. According to key informants, the Northern Savannas landscape is one of the areas where the concept is being developed in Uganda, with plans to develop a conservancy that will exceed Kidepo Valley National Park in size. So far, 17 sub-counties in the region have created wildlife associations to support formation of the conservancy. Private sector informants see great potential for investment in nature-based businesses through the creation of community conservancies. However, they also note that insecurity remains a barrier to greater private sector activity in the region.

Partnerships with NGOs and donors help support the work of government conservation agencies in the landscape, though insecurity remains a barrier to collaboration, particularly in South Sudan. Donors and NGOs play an important role in facilitating the development of conservancies in Kenya, including the Northern Rangelands Trust. Donors such as USAID and Research Triangle International have been instrumental in promoting the concept in Uganda. Donors and NGOs may also support conservation indirectly through promoting alternative livelihoods and increasing stocks of resources outside of protected areas. For example, key informants report that NGOs and donors have funded beekeeping projects as an alternative livelihood and large-scale tree planting projects to restore degraded habitats or create woodlots for fuelwood and building materials. Around the Ugandan portion of Mount Elgon, afforestation has also been conducted as a carbon sequestration initiative under global carbon payment standards as part of the Trees for Global Benefits Program run by the Environmental Conservation Trust of Uganda (ECOTRUST).

PEOPLE AND LIVELIHOODS

Several different ethnic groups live in and around the wildlife landscapes of this study region, undertaking a variety of livelihood activities. In South Sudan, while the livelihood patterns are focused on agriculture, most households supplement this with fishing, hunting, and gathering a range of wild foods and bush products (FEWS NET, 2018). Their dependence on wild resources has only increased during the ongoing conflict. The main crops grown are maize, cassava, sorghum, and ground nuts. Some households may also keep goats and chickens.

The local communities surrounding Kidepo Valley National Park in the Karamoja region of Uganda include the pastoral Karamojong people and the Ik people who keep few livestock and are experts at hunting and gathering wild species, but also farm some crops for subsistence purposes. The main food crops grown here are maize, sorghum, potatoes, and vegetables. Honey and the collection of other wild foods such as fruits and wild game are important for poorer households. The Karamojong people are nomadic agropastoralists who keep cattle. Further south and southwest, the main source of livelihood is rainfed subsistence agriculture with mixed cropping, alongside animal husbandry (FEWS NET, 2010b). Most households keep cattle and goats. Central Karamoja is more suitable for rearing livestock, with wealthier households keeping cattle and poorer households keeping sheep and goats. Rearing of livestock is characterized by seasonal movement in search of water and pasture.

Further south around Mount Elgon where rainfall is plentiful and less erratic, households farm bananas, coffee, and vegetables. This area is less susceptible to prolonged dry spells and is considered a food surplus area. Households also supplement with livestock, mainly cattle and goats.

The total population within this study region is 6.3 million, with 97 percent considered rural (Table 27). The average household size in the region is 5.3 and is highest in South Sudan.

Table 27. Population statistics for the Northern Savanna study region

COUNTRY	TOTAL POPULATION	NUMBER OF RURAL HOUSEHOLDS	AVERAGE HOUSEHOLD SIZE	% RURAL
South Sudan	1,978,568	335,351	5.9	100
Uganda	3,226,161	641,610	5.0	95
Kenya	1,158,263	220,887	5.1	97
Total for study region	6,362,993	1,197,847	5.3	97

Due to the history of armed conflict in the region, the area hosts large refugee populations. This is particularly the case in northern Uganda, which hosts several hundred thousand refugees from the civil war in South Sudan (UNHCR, 2019). While no major refugee settlements are located within the wildlife landscape, the Palabek refugee settlement in Lamwo District is situated relatively close to the landscape's northwestern border. Around 50,000 refugees currently reside here, exerting significant pressure on surrounding natural resources. Additionally, key informants report that large numbers of refugees from South Sudan use the wildlife landscape as an entry point into Uganda.

ECOSYSTEM SERVICES

NATURE-BASED TOURISM

Until recently, tourism in the northeastern Karamoja region of Uganda was underdeveloped and off the radar for even the most adventurous of tourists. The region was largely inaccessible by road and isolated from the rest of Uganda, and tribal conflicts raised security concerns for potential travelers. However, newly paved roads, a chartered air service, the construction of safari lodges, and a relative return to peace across the region has resulted in a significant increase in the number of tourists to this remote wilderness. Despite these recent improvements, a further improvement to the region's security status and road network will still be needed for tourism in the area to reach its full potential. Tourism in Karamoja is centered on nature and nature-based activities. In Kidepo Valley National Park, vast grasslands extend in all directions toward distant mountain ranges; it is home to big game including elephant, giraffe, lion, cheetah, zebra, eland, hartebeest, hyena, and one of Africa's largest herds of buffalo. Wildlife viewing safaris are the most popular activity, but other activities are also attracting visitors to this part of Uganda, including safari hunting opportunities. Karatunga is a notable nature-based tourism initiative that has achieved success in promoting tourism across less visited parts of Karamoja, such as wildlife viewing in Matheniko and Pian Upe Game Reserves and hiking on Mounts Napak, Moroto, or Kadam and other peaks in the landscape. The Tour of Karamoja bike ride, which started in 2018, is a bike tour that takes place over several days, exploring the wildlife landscapes of the region across gravel roads. The 2020 edition of this race started in Pian Upe Wildlife Reserve and went to Kidepo Valley National Park. Cultural group safaris are also offered, as are daily hiking trips to visit Uganda's most remote community, the Ik people, who live on the ridges of the Lomej Mountain. These activities are attracting adventurous travelers to this remote area of Uganda in search of a unique wildlife and cultural experience that is devoid of crowds typically seen in the most popular national parks of the region.

In the war-torn nation of South Sudan, the tourism industry (as well as much of the wildlife) is essentially non-existent, with very few, if any, tourists visiting the country to explore its six national parks and 13 game reserves. However, the country is now emerging from its conflict and with a peace deal in place, is focusing on diversifying revenues with the hope of growing tourism. It is reported that less than 6 percent of the national budget is put toward the wildlife ministry, and in 2018 no money was allocated toward tourism development.[4] Without the necessary investment, the industry will likely take decades to develop. In this assessment, we assume that there is no functioning tourism industry in South Sudan. However, we recognize that there is great potential for tourism to grow and flourish, and if wildlife landscapes are properly managed, they could provide income, jobs, and numerous valuable ecosystem services to the people of South Sudan. Indeed, Gowdy & Lang (2015) estimate that the potential economic contributions of the Sudd Wetland, just north of this study region, to be almost US$1 billion per year. The study states that a key component to the future economic health of South Sudan lies in the preservation of its unique wildlife heritage (Gowdy & Lang, 2015).

Mount Elgon is situated in the far south of this wildlife landscape. The higher slopes are protected by national parks in both Uganda and Kenya, creating an extensive transboundary conservation area, which has been declared a UNESCO Man & Biosphere Reserve. Nature-lovers are attracted to this park for its

[4] See https://www.theguardian.com/global-development/2019/sep/06/south-sudan-turns-to-tourism-in-bid-to-draw-line-under-past-unrest

wide variety of activities, including hiking, rock climbing, mountain biking, birdwatching, nature walks, and camping. Waterfalls and caves are scattered throughout the park, the most famous cave being Kitum, which is regularly visited by elephants, bushbuck, duiker, and buffalo in search of the salts that are found in its mineral-rich soils.

Park visitor numbers were available for the two national parks in Uganda: Kidepo Valley and Mount Elgon (Figure 41). While the total number of visitors to Kidepo National Park represents only 3 percent of the total park visitors to Uganda, the annual growth rate of 13 percent is one of the highest visitor growth rates of all the parks. In 2016, a total of 7,824 people visited the park, an increase of 524 percent since 2002 when only 1,443 people visited. Visitor numbers to Mount Elgon National Park have remained relatively stable over time, peaking in 2008 at 3,708 visitors. In 2016, a total of 3,335 people visited the park, an annual growth rate of just 0.2 percent.

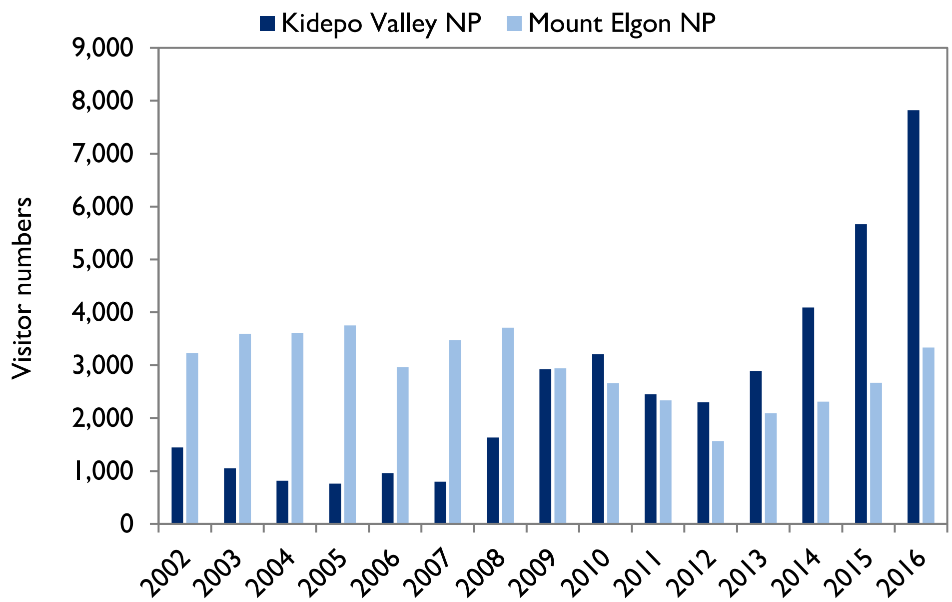


Figure 41. Total number of visits to Kidepo Valley and Mount Elgon National Parks in Uganda from 2002-2016

Source: Uganda MTWA

There are mechanisms in place to ensure local communities living around protected areas benefit from nature-based tourism. In Uganda, 20 percent of park gate fees are shared with local communities. These funds are channeled through the district and sub-county local governments to fund development projects that benefit the community as a whole (e.g., health facilities and schools). A key informant from the Kidepo Valley region noted that communities also earn revenues from sport hunting and through gate collection fees charged for visiting community conservation areas adjacent to the national park. Sport hunting in community areas can be particularly lucrative, as 90 percent of these hunting revenues are meant to go to communities. In Kenya, revenues raised from protected areas are shared with local communities through the county governments. Such mechanisms do not currently exist in the South Sudan portion of the landscape, given the absence of tourists. This was confirmed by community key informants, who said that tourists do not visit the area and thus communities do not accrue any benefits from nature-based tourism.

Community benefits from nature-based tourism are not limited to revenue sharing. Community informants noted that they also benefit from selling items to tourists and from money generated by cultural shows. Communities also benefit from wildlife-based tourism through employment opportunities, particularly around Kidepo Valley National Park where a key informant reported there are nine tourist lodges each employing around 50 people. In Uganda, some private sector tour operators benefit communities through providing employment, supporting cultural groups and paying community-based organizations for conservation important natural and cultural tourism sites. Community informants from the Kenyan side of Mount Elgon similarly noted they benefit from employment as guides and porters for tourists.

This study uses a combination of tourism data and patterns of geotagged photographs uploaded to the Internet to estimate ecosystem contribution to nature-based tourism value in 2018 in the wildlife landscapes of the Northern Savannas. This study follows the methods of Turpie et al. (2020) and Turpie et al. (2017) who used this approach to estimate the value of nature-based tourism at a national and sub-national scale. Tourism's direct contribution to GDP was extracted from WTTC reports and multiplied by the proportion of leisure spending to generate a value of total leisure spending in each country. The proportion of tourism expenditure attributed to tourist attractions, as opposed to activities such as visiting family and friends, attending conferences or religious events, or receiving medical treatment, was then estimated for each tourist group (holiday, visiting friends and relatives, business, and other) based on information collated from individual country tourism statistics reports and information related to tourist spending patterns (Table 28). The spatial distribution of tourism value was mapped using the InVEST Recreation Model 3.5.0 (http://www.naturalcapitalproject.org). This model uses geotagged photographs uploaded on the website flickr.com to estimate the relative value of tourism across an area.

Table 28. Typology of tourists to Uganda and Kenya in 2018

PURPOSE OF VISIT	UGANDA (%)	KENYA (%)
Holiday	22	74
VFR	39	7
Business	32	13
Other	7	6

Leisure tourists (*i.e.*, tourists visiting on a holiday) represent most of the tourists in Kenya but less than a quarter of the tourists to Uganda (Table 28). These tourists have the highest daily spending rate and generate the most revenue. Their spending is largely on visiting attractions. The total attraction-based tourism value in 2018 for Kenya was estimated to be US$1.69 million and for Uganda US$220 million. These values were spatially allocated in proportion to photo density (from the InVEST Recreation Model) to generate an estimate of the value of the wildlife landscape, *i.e.*, the proportion of the total attraction-based tourism value associated with the natural areas within the Northern Savannas study region. This represents the nature-based tourism value of these landscapes (Figure 42). The total nature-based tourism value of the Northern Savannas landscape was estimated to be US$8.9 million in 2018:

US$6.6 million in Uganda and US$2.3 million in Kenya. In Uganda, this represents 3 percent of the total attraction-based spending in the country, and in Kenya it represents less than 1 percent of the total national attraction-based spending (Table 29).

Table 29. The estimated total attraction-based tourism value for Uganda and Kenya in 2018 and estimated nature-based tourism value of the Northern Savannas landscape

COUNTRY	TOURISM DIRECT CONTRIBUTION TO GDP	LEISURE SPENDING AS A PROPORTION OF TOTAL SPENDING (%)	TOTAL ATTRACTION-BASED TOURISM VALUE PER COUNTRY	TOURISM VALUE OF WILDLIFE LANDSCAPE	% OF NATIONAL VALUE
Uganda	$715 m	87	$220 m	$6.59 m	3
Kenya	$2,983 m	64	$1,693 m	$2.29 m	0.15

All values in 2018 US$ millions.

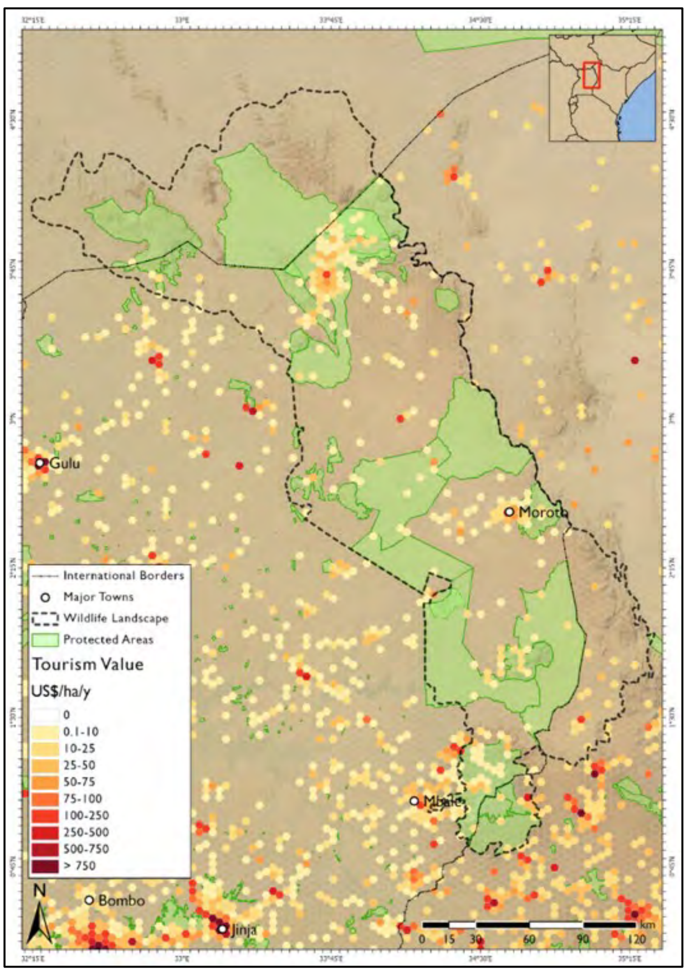


Figure 42. Tourism value (US$/ha/y) for 2018 across the Northern Savannas wildlife landscape, based on the distribution of geo-referenced photographs uploaded to Flickr

The Kidepo Valley National Park in Uganda has the highest tourism value at US$1.9 million, followed by Mount Elgon National Park in Uganda and Kenya (Table 30, Figure 42). Mount Elgon National Park in Uganda has the highest per hectare value at US$18. The forest reserves, community wildlife management areas, and wildlife reserves in Uganda have average per hectare values ranging from US$0.1 to US$6.3. Kidepo and Mount Elgon National Parks in Uganda contribute 34 percent of the total tourism value of the Northern Savannas wildlife landscape.

Table 30. The estimated tourism value of the main protected areas within Uganda and Kenya that are situated within the Northern Savannas landscape

PROTECTED AREA	COUNTRY	TOURISM VALUE (US$/Y)	TOURISM VALUE (US$/HA/Y)
Kidepo Valley National Park	Uganda	1,890,000	13.2
Mount Elgon National Park	Uganda	1,109,000	18.3
Mount Elgon National Park	Kenya	182,000	11.5
Moroto Forest Reserve	Uganda	306,000	6.3
Karenga Community Wildlife Management Area	Uganda	306,000	3.2
Bokora Corridor Wildlife Reserve	Uganda	192,000	1.1
Pian Upe Wildlife Reserve	Uganda	57,000	0.3
Chepkitale National Reserve	Kenya	52,000	2.8
Matheniko Wildlife Reserve	Uganda	11,000	0.1

All values 2018 US$.

FLOW REGULATION

Natural ecosystems regulate seasonal surface flows through infiltration of rainfall into groundwater flows, and in so doing reduce the seasonal variation in flows by slowing down water through the landscape and contributing to river base flows during the dry season. This reduces the size of reservoirs that are needed to meet water demands, as well as affecting the availability of water to people who draw water directly from streams and rivers. In this study, the flow regulation service was evaluated as the difference in the contribution to baseflow (*i.e.*, water that reaches a stream) between current land cover and a scenario in which all land cover is converted to bare ground.

The Northern Savannas wildlife landscape was estimated to have an average baseflow contribution of 1,252 m^3 per hectare per year (Figure 43). Recharge, and thus contribution to baseflow, is generally higher in areas under natural vegetation and higher rainfall, although soil characteristics are another moderating factor. The highest local recharge values in the modelled Northern Savannas region were often associated with high rainfall forested areas such as the Imatong Mountains in South Sudan in the far north of the landscape and Mount Elgon along the border of Uganda and Kenya in the far south of the landscape, where recharge values were higher than 2,900 m^3 per hectare per year. The drier valleys of Kidepo in the north and the central areas to the west of Moroto make little to no contribution to baseflows. The heavily cultivated areas in the high rainfall areas surrounding Mount Elgon also make little to no contribution to recharge, suggesting that the clearance of natural habitats has resulted in declined dry season flows in these parts of the region.

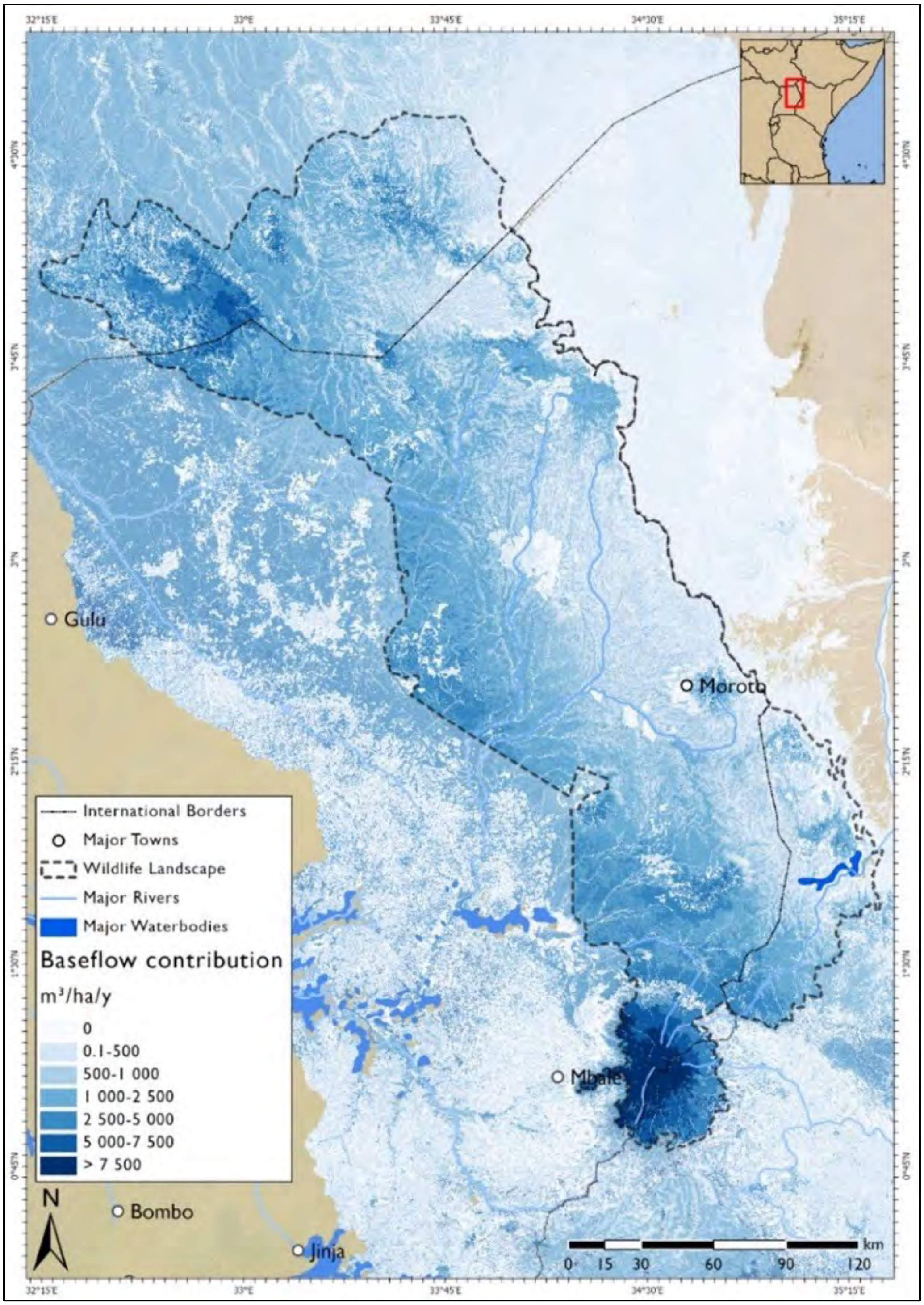


Figure 43. Baseflow contribution (m^3 per ha per year) by ecosystems of the Northern Savannas wildlife landscape relative to a barren landscape

We estimated the total baseflow contribution across the Northern Savannas landscape to be 7,044 million m^3 per year, retaining 4,618 million m^3 per year when compared to a barren landscape. The average water retention was estimated to be 855 m^3 per hectare per year in Uganda, 782 m^3 per hectare per year in South Sudan, and 693 m^3 per hectare per year in Kenya. If these flows were not being infiltrated and retained by the landscape, the cost of having to construct storage infrastructure that would be needed to capture these additional flows was estimated to be approximately US$515 million per year: US$357 million per year in Uganda, US$113 million in South Sudan, and US$46 million in Kenya.

WATER QUALITY AMELIORATION

Levels of phosphorus export are generally low across the Northern Savannas wildlife landscape (Figure 44). However, some notable areas with higher nutrient export values are visible on the map. These are associated with cultivated areas, the most sizeable of which are dotted across the Ugandan portion of the wildlife landscape. High nutrient export values are associated with more intensively cultivated land adjacent to the southern part of the wildlife landscape, particularly on the Kenyan side where farmers apply more fertilizer on average. The situation is different further north where cultivation is less prevalent. As a result, no clear contrast exists in levels of nutrient export inside and outside the more northerly parts of the wildlife landscape.

For valuation of the nutrient retention service, we focused on the portion of the wildlife landscape situated in the catchment of Lake Kyoga. This encompassed much of the central and southern parts of the Ugandan portion of the wildlife landscape. From our InVEST model, we estimated current phosphorus export from this portion of the wildlife landscape to be on the order of 664 tons. We estimated both the active and passive nutrient retention service provided by this part of the wildlife landscape. The active service refers to the current retention of nutrients by vegetation. If vegetation in the Lake Kyoga catchment portion of the wildlife landscape stopped retaining phosphorus, we estimated the replacement cost of the retention service to be on the order of US$502,591. This value of the active service is fairly small, as most of the cultivated areas exporting large loads of phosphorus to Lake Kyoga are located downstream of the wildlife landscape (Figure 46). Since nutrient export is generally low across the wildlife landscape, active retention of nutrients by vegetation in the wildlife landscape is in turn low. However, export of phosphorus to Lake Kyoga would be much higher if this portion of the wildlife landscape were converted to agriculture, as demonstrated by the high nutrient export values associated with cultivated areas. The nutrient export avoided by maintaining natural vegetation, at the expense of cultivation, is the passive service provided by this portion of the wildlife landscape. We estimated the replacement cost of this passive service to be on the order of US$573,618.

EROSION CONTROL

Natural habitats reduce soil erosion and transport of sediment to downstream habitats. This can occur through both *in situ* retention of soil due to vegetation cover, as well as through the trapping of sediments that have been eroded from elsewhere in the landscape. By doing so, natural vegetation can reduce the negative impacts of excess sediment loads in watercourses, such as reduced water quality and loss of reservoir storage capacity. In this study, the sediment retention service was evaluated by the difference in sediment export between current land cover and a scenario on which all land cover is converted to bare ground. This difference provides a measure of the amount of sediment currently being retained by the landscape.

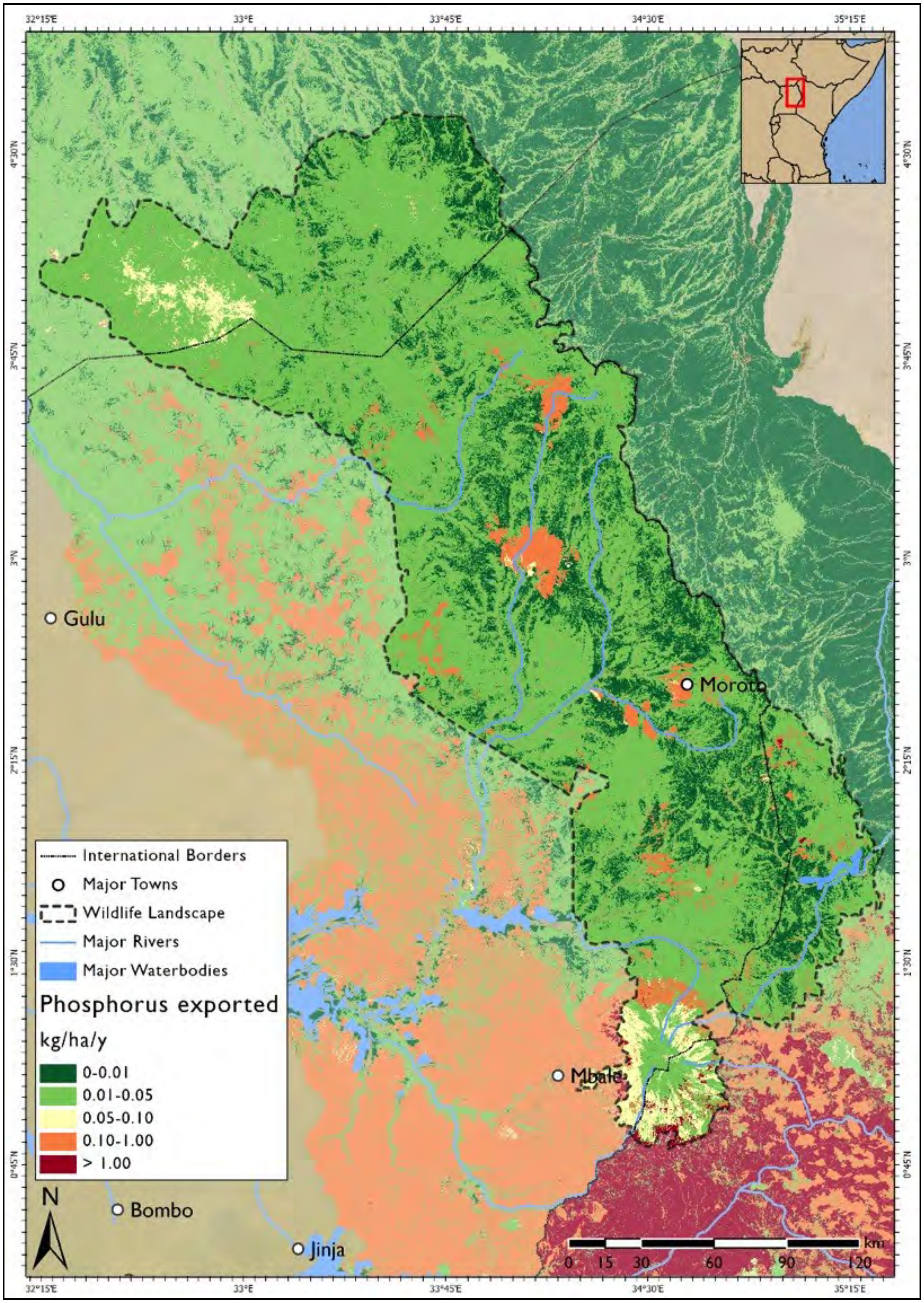


Figure 44. Average phosphorus exported (kg per ha per year) by ecosystems of the Northern Savannas wildlife landscape

Our model produced a relatively high average sediment retention value for the Northern Savannas wildlife landscape of 226 tons per hectare per year (Figure 45, Table 31). However, significant variation in sediment retention values across the region is evident. Sediment retention values were estimated to be relatively low across much of the central and western parts of the wildlife landscape (< 15t/ha), where gradients are gentle. That said, much higher sediment retention values were estimated for high relief parts of the landscape, such as the Imatong Mountains in the far northwest and Mount Elgon and the escarpment areas of the Uganda/Kenya border region. Steep slopes, often with higher rainfall, mean potential soil erosion is high in these areas in the absence of vegetation cover. Sediment retention values in excess of 600 t/ha are seen in these regions, indicating that the presence of natural vegetation substantially lowers soil erosion relative to a bare ground scenario. Several rivers have their headwaters around Mount Elgon, indicating the importance of forest habitats for retaining sediment in this high-risk area for soil erosion. Notably, the headwaters of Lake Victoria and Lake Kyoga originate from Mount Elgon, indicating that maintaining natural vegetation is of value for reducing sediment loads into these important lake systems. Part of Mount Elgon and the Loima Hills to the northwest also fall within the catchment of the Turkwel River, a sub-catchment of Lake Turkana. Natural vegetation in these regions thus plays an important role in reducing sediment export and loss of storage capacity in the Turkwel Gorge Dam, an important hydroelectric facility and irrigation source in dry northwest Kenya.

In total, we estimated that current land cover retains 1.27 billion tons of sediment per year (226t/ha/y) relative to a scenario in which all land cover is converted to bare ground (Table 31). Most of this retention falls within Uganda and South Sudan. Mean sediment retention was highest in South Sudan where the natural vegetation retains 398 tons of sediment per hectare per year. It was lowest in Uganda, where only 149 tons of sediment per hectare, on average, was retained by the natural vegetation. If this sediment were not being retained by the landscape, the replacement cost of this service, in terms of the construction of sediment check-dams, was estimated to be US$1.56 billion per year.

Table 31. Total sediment retained, mean sediment retained per hectare per year, and the total annual cost of sediment retention (US$ million/y) for the Northern Savannas wildlife landscape

COUNTRY	TOTAL SEDIMENT RETAINED (MT/Y)	MEAN SEDIMENT RETAINED (T/HA/Y)	TOTAL ANNUAL VALUE (US$ MILLION/Y)
Kenya	201	338	246.8
South Sudan	514	398	631.0
Uganda	557	149	809.9
Total	**1,271**	**226**	**1,561.9**

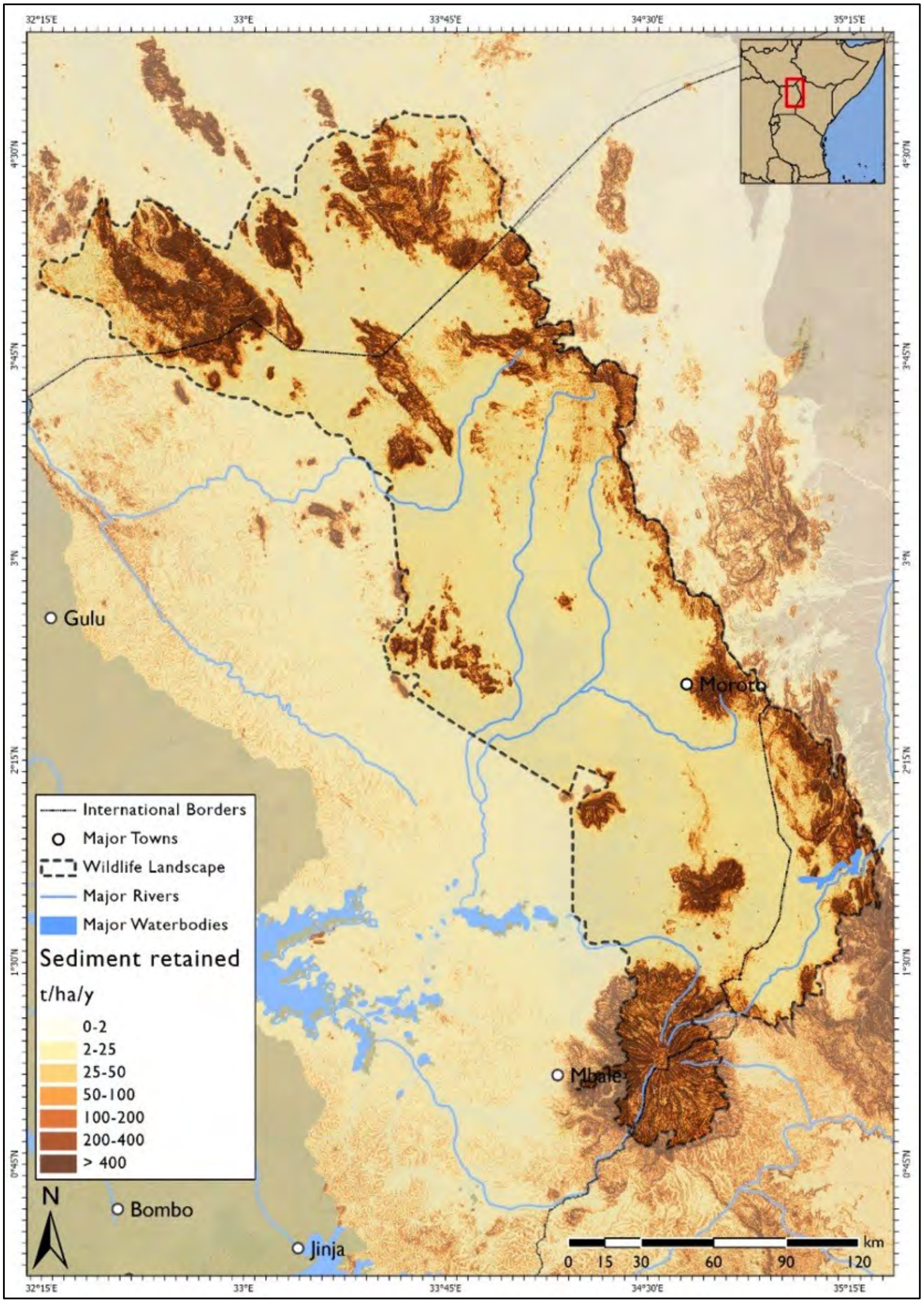


Figure 45. Average sediment retained by ecosystems in the Northern Savannas landscape (tons per ha per year) relative to a barren landscape

CARBON STORAGE

Natural ecosystems make a significant contribution to global climate regulation through the sequestration and storage of carbon. About half of all vegetative biomass comprises carbon. In addition to accumulation in woody biomass, carbon accumulates in soils and peat as a result of the accumulation of leaf litter and partially decayed biomass. Degradation of vegetated habitats releases carbon and contributes to global climate change with impacts on biodiversity, water supply, droughts and floods, agriculture, energy production, and human health, whereas restoration or protection of these habitats mitigates or avoids these damages, respectively. The conservation and restoration of natural systems thus helps to reduce the rate at which greenhouse gases accumulate in the atmosphere and the consequent impacts of climate change.

Much like the Great East African Plains, the Northern Savannas wildlife landscape is dominated by grassland and wooded grassland. However, in the far north and south of this landscape, there are larger areas of Afromontane forest (*e.g.*, Imatong Mountains in southern Sudan, the mountainous terrain of northeastern Uganda, and around Mount Elgon). Therefore, this wildlife landscape makes a significant contribution to mitigating climate change. Although grasslands are less carbon-dense than bushland and forest, where above-ground vegetation makes up only a small proportion of the total carbon pool, they play an important role in mitigating climate change through the sequestration of soil carbon (Dlamini *et al.*, 2016). Indeed, grasslands are estimated to contain up to one-third of above- and below-ground carbon stocks globally (Tessema *et al.*, 2020). Furthermore, the soil organic carbon pool in grasslands is critically important for soil fertility and plant productivity, contributing to flow regulation. Soil organic carbon is an important indicator of grassland productivity, and deep-rooting African grasses have been shown to be highly productive in sequestering carbon (Tessema *et al.*, 2020). The perennial nature of grasslands allows for the continuous input of carbon from above-ground vegetation to the subsoil via extensive root systems, to depths of several meters. As a result, soil carbon contributes more than two-thirds of the ecosystem carbon that is found in grasslands (Dlamini *et al.*, 2016). However, grasslands are threatened by overgrazing (amongst other factors, but overgrazing is the main contributor to grassland degradation), and evidence suggests that the impacts of overgrazing on soil organic carbon can be significant, with global losses in grassland soil organic carbon stocks of between 1.2 percent and 4.2 percent, as a result (Dlamini *et al.*, 2016; Tessema *et al.*, 2020).

Based on global datasets derived from satellite data (see FAO & ITPS, 2018; Spawn & Gibbs, 2020), it was estimated that approximately 2.2 billion tons of carbon are stored within the vegetation and soils of the Northern Savannas wildlife landscape (Table 32, Figure 46). Given that most of this landscape is within Uganda, it is unsurprising that the majority of the total carbon stored (53 percent) is found in that country. However, the average carbon stored per hectare is highest in Kenya, at 750 tons, and lowest in Uganda at 350 tons per hectare. These mean values fall between what was recorded in the Albertine Rift Forest landscape and the Great East African Plains landscape. Densities are highest in the areas where Afromontane forest is found, such as the Imatong Mountains in South Sudan and around Mount Elgon along the Uganda-Kenya border. The potential for carbon sequestration initiatives in the latter region has already been capitalized on through run by the Environmental Conservation Trust of Uganda's (ECOTRUST) Trees for Global Benefits Program. This is a cooperative carbon offsetting scheme that links smallholder farmer agroforestry initiatives in Uganda to the voluntary carbon market using the Plan Vivo Standard (ECOTRUST, 2019). It has thus been described as a payment for ecosystem services (PES) scheme where farmers are paid for increasing rather than decreasing tree cover in a region that has experienced heavy deforestation. This can help provide a financially attractive alternative to harvesting woody biomass for charcoal production.

The total global damage costs avoided by retaining the total stock of biomass carbon is significant, at almost US$150 billion per year (Table 33). The avoided damage costs to the region amount to about $260 million per year.

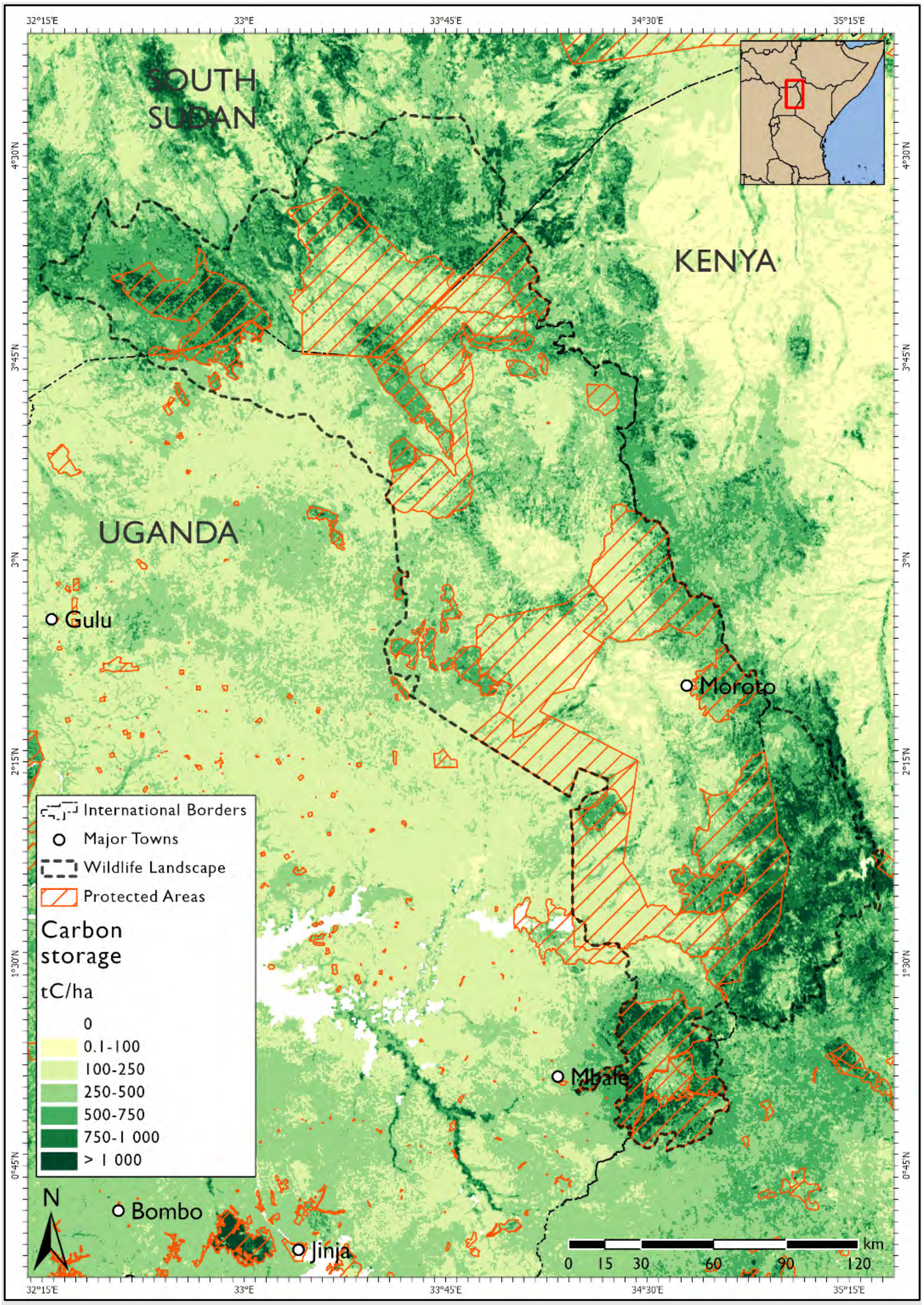


Figure 46. Total carbon storage (tons/ha) across the Northern Savannas wildlife landscape

Table 32.The total amount of carbon stored within the Northern Savannas landscape and summary statistics (tons carbon per hectare) per country

COUNTRY	TOTAL STOCK OF CARBON (METRIC TONS)	MEAN T/HA	MIN T/HA	MAX T/HA
Uganda	1,134,397,486	349.45	64.26	1,917.07
South Sudan	625,143,362	558.17	77.53	1,623.71
Kenya	388,060,992	749.81	6.36	1,558.25

Table 33. The total global damage costs avoided by retaining the total stock of biomass carbon and the avoided damage cost to each country (US$ million/y)

	KENYA	SOUTH SUDAN	UGANDA	REST OF THE WORLD
Carbon storage value (damage costs avoided, US$ million/y)	57.0	55.6	147.5	149,720

POLLINATION OF CROPS

Pollination services are widely recognized as critical for human wellbeing and survival given their vital role in ensuring food security. However, the value of wild pollinators remains unclear. This is concerning for sub-Saharan Africa, a region highly dependent on subsistence agriculture as a main source of livelihood (Tibesigwa *et al.*, 2019). The presence of wild pollinators is directly linked to natural vegetation (Kremen *et al.*, 2004), which plays an essential role in certain life cycle stages of pollinator species, such as through the provision of nesting sites or forage at certain times of year. Insects are responsible for 80-85 percent of all pollinated commercial crops, which represents about one-third of global food production (Allen-Wardell *et al.*, 1998; Klein *et al.*, 2007).

In the northern parts of this wildlife landscape, smallholder cultivation is not as dominant as it is in the southern areas around Mount Elgon where rainfall is plentiful. The dominant food crops grown in the north are maize, cassava, sorghum, ground nuts, and, in some areas, vegetables. In the south, households farm a larger variety of fruits and vegetables, as well as coffee. While not all of these crops require insect pollination (*e.g.*, maize), the majority of them do (*e.g.,* vegetables, fruits, coffee, beans, groundnuts), either having reduced yields (of up to 90 percent) or showing a reduction in seed/breeding yield without wild pollination (*e.g.*, cassava).

The distribution and value of the pollination service to cultivated crops is shown in Figure 47; the value of wild pollination service (contribution to production from crops in smallholder cultivated land) is summarized in Table 34. Based on the percentage share of natural vegetation within a 1,000 m buffer distance of all cultivated land surrounding the wildlife landscape, we estimate the value of wild pollination services to nature-dependent smallholder cultivated land in this study region to be US$144.3 million per year. Just over two-thirds of this value is within Uganda, 22 percent in Kenya, and 11 percent in South Sudan. The overall mean pollination value across the landscape was estimated to be US$125 per hectare, and was highest in Uganda and lowest in South Sudan. The vast extent of natural vegetation, both inside and outside of protected areas, provides valuable pollination services to smallholder farmers across this study region. Values are highest (>US$450/ha) in the central areas around Moroto, just north of the Bokoro Wildlife Corridor, around Kaabong just south of Kidepo National Park, and in the south around Mount Elgon (Table 34). It is also valuable in the far north, in the areas northeast of Kidepo Game Reserve in South Sudan.

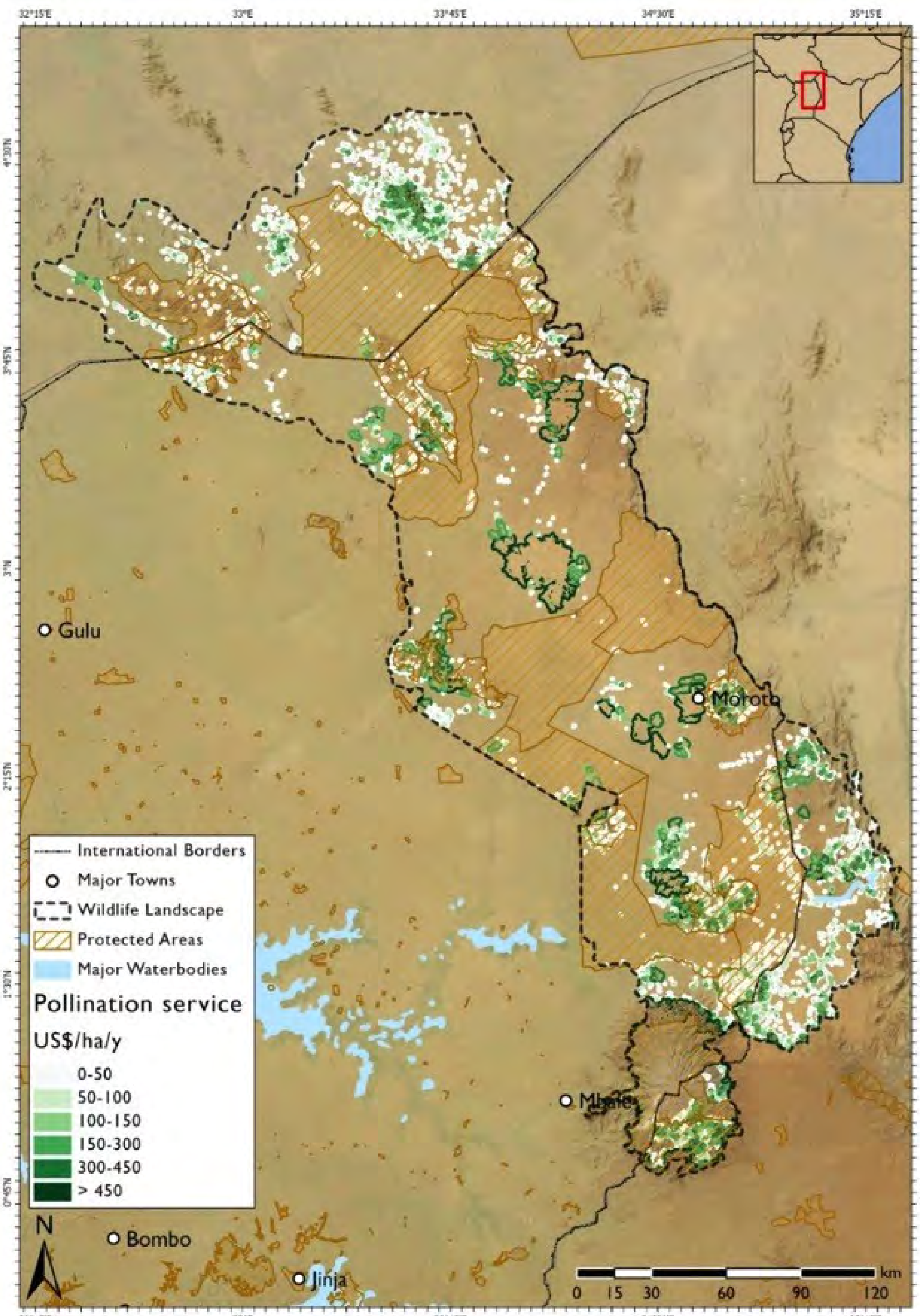


Figure 47. Contribution of pollination services from natural habitats to smallholder farmer revenues in the Northern Savannas wildlife landscape (US$/ha/y)

Table 34. The value of the wild pollination services within Northern Savannas wildlife landscape (2018 US$)

	TOTAL HA OF NATURAL VEG IN BUFFERS	TOTAL POLLINATION SERVICE VALUE (US$)	MAX VALUE (US$/HA)	MEAN VALUE (US$/HA)
Uganda	611,448	96,701,184	946	158
South Sudan	298,671	15,325,153	874	51
Kenya	246,904	32,295,652	874	131
Total	**1,157,023**	**144,321,989**	**946**	**125**

Note that the minimum value in all cases was zero.

FORAGE FOR LIVESTOCK PRODUCTION

The natural rangelands of the Northern Savannas wildlife landscape support the pastoralist and agro-pastoralist communities whose main livelihood is livestock production. In northeastern Uganda, pastoral production systems dominate, with farmers keeping indigenous breeds in herds ranging from a few to 100 head (FAO, 2019). The pastoral Karamojong people live in the arid Karamoja Region of Uganda, moving their livestock seasonally to neighboring areas in search of water and pasture. Similarly, agro-pastoralism is the main livelihood system in the rural areas of South Sudan. While most households cultivate cereal crops, they rarely produce enough to meet their needs and rely heavily on livestock sales each year to purchase food crops from market (Catley, 2018). In the far south of this study region around Mount Elgon, pastoralism is less dominant, with households usually keeping fewer livestock, where they are fed forage and mainly kept for dairy.

It is estimated that there are 2.6 million LSUs across the Northern Savannas wildlife landscape, with just over 2 million LSUs in Uganda, 209,000 LSUs in South Sudan and 353,364 LSUs in Kenya. The density of livestock was estimated to be 0.63 LSUs per hectare in Uganda, 0.19 in South Sudan, and 0.68 in Kenya.

Livestock production contributes some US$1.2 billion to Uganda's GDP, US$3.6 billion to South Sudan's GDP, and US$3.4 billion to Kenya's GDP (in 2018 prices; Uganda Bureau of Statistics, Kenya National Bureau of Statistics, IGAD, 2015). The value of livestock production was mapped based on 2010 density estimates for cattle, sheep, and goats at 10 km resolution (refer to Figure 6). Within the Northern Savannas wildlife landscape, it is estimated that the natural rangelands contribute US$372.3 million per year to livestock production – US$237 million in Uganda, US$64 million in South Sudan, and US$71 million in Kenya, representing 20.4 percent, 1.7 percent, and 2.1 percent of total country production, respectively. The value of this service is highest in the Karamoja region of Uganda.

HARVESTED RESOURCES

In addition to their conservation value, natural resources play an important role in supporting the livelihoods of people. Wild plant and animal resources are harvested for food, medicine, energy, and raw materials, particularly where there are limited economic opportunities. The capacity of the landscape to supply different types of wild resources is related to vegetation type and condition, availability of water, and other factors. However, a number of other factors determine their use and value, and these vary in space and time. The accessibility of wild resources is determined by regulations such as land tenure and harvesting rights, social norms and informal agreements, geographic features such as topography and rivers, and human-made features such as roads. The demand for wild resources is influenced by the socio-economic circumstances of households and the prices of alternatives.

Wildlife habitats usually require full, no-take protection, not only because of the risk associated with overharvesting that changes the nature and functioning of wildlife habitats, but also because of the disturbance that it can cause, especially affecting the shier and more vulnerable wildlife species. The people that live around these wildlife habitats are largely dependent on wild resources, particularly during times of economic stress. Examples of such stressors include crop disease, drought, or floods, which are likely to only worsen with climate change, and international pandemics such as COVID-19. During these times, people fall back on nature to fill livelihood needs. However, this is a potentially vicious cycle of unsustainability as more people rely on nature for food and raw materials and stocks become depleted. The stocks of resources protected within parks and reserves help to maintain the stocks utilized outside of these wildlife habitats. The more resources harvested unsustainably, the fewer there will be available in the future and the less people can rely on nature to fill this need. As resource stocks outside of wildlife habitats become degraded, there will be a higher demand for the resources on the edge of these landscapes as well as on the inside.

Entrance to and harvesting of resources in most protected areas in the landscape is prohibited by law in theory. Nevertheless, significant harvesting likely still takes place, especially where law enforcement effectiveness is low. Due to the absence of management in the South Sudan portion of the wildlife landscape, it is likely that local communities can harvest freely in protected areas there. Restrictions on access and harvesting were a cause of resentment among key community informants living around protected areas in the Ugandan portion of the landscape, particularly where communities had been historically displaced by the gazetting of these areas. However, Uganda does have some provisions for limited legal access to state protected areas for natural resource harvesting purposes. Under these arrangements, the Uganda Wildlife Authority allows communities limited access to parks to collect resources such as thatching grass, medicinal plants, and honey (Tolbert *et al.*, 2019). Harvesters are only granted permission following registration and authorization, and the quantities and types of resources permitted for harvesting are limited to avoid negative environmental impacts. Additionally, key informants report that limited harvesting of specific resources, including firewood and honey, is permitted in forest reserves on the Kenyan side of Mount Elgon.

THE DEMAND FOR NATURAL RESOURCES

In the northern parts of this study region, where rainfall is limited and erratic, people are particularly reliant on the collection of wild plant foods to supplement their diets. While subsistence agriculture is the main livelihood activity there, it is prone to hazards such as drought and crop disease. Political instability in this region has also contributed to people's reliance on wild resources as a source of income. Households collect, consume, and sell wild fruits and vegetables, thatching grass, shea butter, and fuelwood. In the central parts of the region, livestock production is the main livelihood activity, with farmers moving livestock vast distances in search of grazing and water during the dry season. These farmers are also known to collect wild resources. In the southern parts of the study region around Mount Elgon, rainfall is more reliable, and agriculture is the dominant activity. People living here grow a wide variety of fruits and vegetables for home consumption and for sale at market. The main cash crops are coffee and bananas. Poorer households rely on hunting and gathering of fruits, berries, local vegetables, and game to meet total food needs (FEWS NET, 2010a). Conditions are similar in higher rainfall regions in the north of the study region, such as the Imatong Mountains in South Sudan, where a variety of crops are grown and harvesting of wild resources is prevalent (AWF, 2014). Across most of this study region, thatching grass is an important resource that is harvested for use in construction of roofs and for other household items.

The vast majority of households in the region depend on woodfuels as their main energy source, primarily in the form of firewood (Barnes, Ensminger & O'Keefe, 1984; Ellis *et al.*, 1984; AWF, 2014; UBOS, 2018). Charcoal is used to a lesser extent, by only 2 percent of households in the rural areas of South Sudan (Drew, 2012), while it is the main energy source for 15 percent of households in Uganda's Karamoja subregion (UBOS, 2018). Estimates of amounts of firewood consumed were lacking for the region. In the Soroti District, Uganda, slightly west of the study region, Egeru (2014) estimated annual firewood consumption to be 542 kg per capita, or 3,688 kg per household, and charcoal consumption to be 232 kg per capita.

Wood appears to be widely used as a construction material in the region, both for building houses and uses like fencing, but quantitative estimates were again scarce. Across Uganda's Karamoja subregion, 78 percent of huts were reported to use wood as a wall material, mostly in combination with mud (UBOS, 2018). In another Karamoja study, all respondents reported using wood for fencing (Egadu, Mucunguzi & Obua, 2007). Around Mount Elgon, 86.5 percent of households used timber for construction, while 66 percent of households adjacent to the forest used wood from the forest for poles (Scott, 1998). Most of the timber used in construction is bought, with only 3 percent of households in the Mount Elgon region involved in harvesting the timber (Scott, 1998).

Palm leaves are harvested in northern Kenya and South Sudan for use in thatching and for making of various crafts and household items (Table 36). The doum palm (*Hyphaene thebaica*), native to the northern half of Africa, grows in areas where groundwater is present and is found along the Nile River in South Sudan and in localized riverine areas in Kenya. The leaves and trunks of the doum palm are also used for thatching and the construction of walls. In Turkana County in Kenya, which borders South Sudan, the use of palm leaves for thatching ranged from 1-44 percent of households (KNBS, 2019).

Table 35. Proportion of rural households harvesting woody resources for wood fuel and raw materials within each country in the Northern Savannas study region, and the estimated demand per average household per year

COUNTRY	FIREWOOD		CHARCOAL		POLES & WITHIES		TIMBER	
	% RURAL HH	M^3/HH/Y	% RURAL HH	M^3/HH/Y	% RURAL HH	M^3/HH/Y	% RURAL HH	M^3/HH/Y
South Sudan	94	4.1	2	0.1	88	0.8	20	0.3
Uganda	80	4.0	17	0.8	69	0.7	10	0.2
Kenya	82	3.9	12	0.6	76	0.7	13	0.2

Figure 48. Livestock production is the main source of income for many households in Uganda's Karamoja region, and across most of the study region houses are constructed using wooden poles, withies, and thatching grass
Credit: Rod Waddington /Flickr, top; Robin Yamaguchi /Flickr, bottom-left; Rod Waddington /Flickr, bottom-right.

Thatching grass is an important resource across most of this region. In South Sudan, 93 percent of rural households live in grass-thatched mud houses (Table 36). In Kenya, grass thatch is used by 40 percent of households in Turkana County and by 5 percent of households in the Mount Elgon area (KNBS, 2019). Rural households in Uganda also use grass thatch for their roofs (see Figure 48). The use of grass thatch is higher in the northern parts of this study region, with 86 percent of households in Acholi sub-region and 79 percent of households in the Karamoja sub-region using grass thatch (UBOS, 2018). In the sub-region of Mount Elgon, the use of thatch is lower, at around 7 percent (UBOS, 2018).

Table 36. Proportion of rural households harvesting non-woody raw materials within each country in the Northern Savannas study region, and the estimated demand per average household per year

COUNTRY	BAMBOO		PALM LEAVES		THATCHING GRASS	
	% RURAL HH	CULMS/HH/Y	% RURAL HH	KG/HH/Y	% RURAL HH	KG/HH/Y
South Sudan	-	-	37	13.0	93	28.3

COUNTRY	BAMBOO		PALM LEAVES		THATCHING GRASS	
	% RURAL HH	CULMS/HH/Y	% RURAL HH	KG/HH/Y	% RURAL HH	KG/HH/Y
Uganda	45	7.9	7	2.4	45	14.6
Kenya	38	5.5	5	1.7	28	8.5

Bamboo is of localized but high importance in the Mount Elgon region (Table 36). Stems are harvested for a range of purposes, including construction, stakes for growing crops like beans and bananas, and for weaving into granaries and baskets (Scott, 1998). The shoots are also eaten, but the area's Bagisu are the only known ethnic group in Africa that regularly consumes bamboo. Bamboo on Mount Elgon is used to meet a large demand for shoots across Mbale District in Uganda (Scott, 1998). The importance of bamboo is reflected by the fact that 95 percent of households near the forest edge harvest shoots for eating and stems for construction. Due to the large demand for bamboo in the region, selling of shoots and stems also provides an important income source to many households living close to the Mount Elgon forests (Scott, 1998).

Estimates for use of wild plant medicines were scarce for the study area. At the national level, it is estimated that about 80 percent of Uganda's population uses traditional plant-based medicines (Kanabahita, 2001). As northeast Uganda is one of the poorest and least developed parts of the country, dependence on traditional medicines is likely to be particularly high there. A more localized estimate comes from Uganda's Nkasongola District, where 54 percent of households indicated they obtain medicines from savanna woodlands (Kalema, 2010). While Nkasongola lies outside the study area, the savanna woodland is broadly comparable. As some households may buy or receive plant medicines as a gift, this could explain why Kalema's estimate of households harvesting plant medicines is lower than the 80 percent estimate for plant medicine usage in Uganda (Table 37). Low medicinal plant harvesting has been reported among South Sudanese communities around the Imatong Mountains, where only 14 percent household participation was estimated (AWF, 2014). However, it was reported that most households were within 10 km of a healthcare facility, which may have reduced reliance on self-harvested medicines. It is thus unclear whether this is representative of harvesting in other areas of South Sudan, which may have poorer access to healthcare facilities. In contrast, high use of medicinal plants around the Ugandan side of Mount Elgon was reported by Scott (1998). Here, 88 percent of households adjacent to the forest edge used plant medicines, while 78 percent of households further away from the forest were users. In the absence of further information, we assume harvesting on the Kenyan side of Mount Elgon is comparable.

Wild plant foods provide an important nutritional supplement in the area (Table 37). They become particularly important during periods of famine and drought, when they may provide an emergency food source, particularly in the drier parts of the study region (Stites *et al*., 2007; Arensen, 2015). Some of the most important food species in the savanna areas include the widely distributed desert date *Balanites aegyptiaca* and tamarind *Tamarindus indica.* As it bears both edible leaves and fruits and tolerates annual rainfall as low as 400 mm, *B. aegyptiaca* is a particularly important species to communities in the more arid parts of the study region (Hall, 1992; Egeru, Okia & de Leeuw, 2014b; Arensen, 2015). The African shea tree *Vitellaria paradoxica* is another important species, although it is limited to the wetter western savanna parts of the study region, where rainfall exceeds 750 mm (Naughton, Lovett & Mihelcic, 2015). Shea fruits can be eaten directly, while the kernels are dried to make shea butter, which is prized as an edible oil, cooking fat, soap, cosmetics, and medicine (Booth & Wickens, 1988; Naughton *et al.*, 2015). A substantial global demand for shea products in the cosmetic, pharmaceutical, and confectionary industries has also emerged, indicating the commercial potential of the species (Elias & Carney, 2007; Lovett, 2013). The value of wild foods during times of hardship is reflected by the fact that famines in South Sudan in the 1980s have been named after the wild foods people had to use to survive them (Arensen, 2015). However, despite the vital role played by wild plant foods in the region, quantitative estimates of usage are generally lacking. An exception was around the forests of Mount Elgon, where Scott (1998) reported that 63 percent of households adjacent to the forest edge consumed wild vegetables on the Ugandan side. This declined to 38 percent of households further away from the forest. Mushrooms were also widely consumed here, by 81 percent of households adjacent to the forest and 67 percent of households further away (Scott, 1998).

Table 37. Proportion of rural households harvesting wild plants foods and medicines within each country in the Northern Savannas study region, and the estimated demand per average household per year

COUNTRY	WILD PLANT FOODS		MEDICINES	
	% RURAL HH	KG/HH/Y	% RURAL HH	KG/HH/Y
South Sudan	88	124.1	54	6.7
Uganda	82	115.8	41	5.1
Kenya	76	107.1	43	5.3

Harvesting of wild honey in the savanna parts of the study region is a noted activity (Ayoo, Opio & Kakisa, 2013; Burns *et al.*, 2013; Egeru *et al.*, 2014b; Visser *et al.*, 2017), but quantitative data was not found. One estimate was obtained for the forests of Mount Elgon on the Ugandan side, where honey was widely consumed; 88 percent of households adjacent to the forest consumed honey, as did 78 percent of households further away (Scott, 1998). In the absence of further information, we assume honey consumption was similar on the Kenyan side.

Data on bushmeat consumption for the region is scarce. Kayanja & Byarugaba (2001) noted that bushmeat harvesting in Uganda is "significant, but largely unrecorded," which remains generally true today. Quantitative estimates of bushmeat consumption in northeast Uganda could not be found, but declines in availability due to excessive hunting have been reported by local communities (Stites *et al.*, 2007). Olupot, McNeilage & Plumptre (2009) carried out surveys of bushmeat in various regions of western Uganda. While outside of the study area, the Murchison Falls National Park and Kafu Basin areas have broadly comparable savanna vegetation. Reported bushmeat use in these areas was fairly low, at 32 percent and 12 percent respectively, and it was generally not consumed often by these households (Olupot *et al.*, 2009). About two-thirds of meat caught in these areas was sold by hunters, indicating economic incentives are an important contributor to bushmeat hunting in Uganda. Around Mount Elgon in Uganda, similarly moderate consumption of bushmeat was reported by Scott (1998), where 33 percent of households adjacent to the forest and 22 percent of households further away from the forest, consumed bushmeat (Table 38). Detailed information for these resources for the Kenyan side of Mount Elgon could not be found, so it was again assumed harvesting would be comparable here.

Table 38. Proportion of rural households harvesting wild animal resources within each country in the Northern Savannas study region, and the estimated demand per average household per year

COUNTRY	MAMMALS, PRIMATES, BIRDS		WILD HONEY		FISH	
	% RURAL HH	KG/HH/Y	% RURAL HH	LITERS/HH/Y	% RURAL HH	KG/HH/Y
South Sudan	34	16.6	76	2.8	0.3	0.01
Uganda	34	13.2	76	2.8	0.3	0.01
Kenya	28	12.9	76	2.8	0.3	0.01

Somewhat north of the study area boundary, bushmeat consumption appeared to be widespread among communities around Juba and Bandingilo National Park, with an average household consumption of 49 kg per year (Langoya, 2017). In contrast to the findings of the Ugandan studies, consumption of bushmeat exceeded that of domestic meat in this part of South Sudan. Jubara (2019) also found widespread use of bushmeat around Bandingilo National Park, with 95 percent of respondents indicating they consume bushmeat and 55 percent consuming it weekly. Estimated household consumption in this study was 141 kg of bushmeat a year (Jubara, 2019).

THE SUPPLY, USE, AND VALUE OF HARVESTED WILD RESOURCES

To briefly recap, the resource use results are the combined product of natural resource stocks, the availability of these resources for harvesting (protected area status), and the local demand for the various resources. Stocks of natural resources per unit area varied according to habitat type and condition. However, the supply of natural resources was also moderated by protected area status, as we reduced the proportional availability of natural resources where they occurred within protected areas. The magnitude of this reduction varied according to the level of protection. Finally, the data for available stocks per hectare was combined with estimated household demand per hectare. Demand is a function of both the average quantity of resources used per household and the number of households in the area (population density).

Despite moderate to low population densities across much of the Northern Savannas region, fairly high use was estimated for several natural resources (Table 39). This can be partly explained by the greater reliance on certain natural resources in this region, for instance wild foods. Another contributing factor is the relatively low coverage of national parks relative to other wildlife landscapes. While the wildlife landscape does incorporate several sizeable protected areas, Kidepo Valley in the northeast corner of Uganda and the two Mount Elgon National Parks in Kenya and Uganda are the only national parks. Our model assumed a lower level of protection, and thus greater harvesting, in protected areas other than national parks. Hence, the limited national park coverage also contributed to relatively high use values for the wildlife landscape.

A similar general pattern can be identified in the spatial patterns of use for most of the more widely distributed natural resources. Relative to the northern parts of the Ugandan wildlife landscape, use of natural resources was mostly higher in South Sudan (Figure 49-Figure 52; Table 39). Natural habitats across this portion of the landscape are also relatively intact, even outside of protected areas, resulting in a largely continuous distribution of natural resources. Natural resources are also distributed continuously over much of the northern Ugandan portion of the wildlife landscape, often with relatively low estimated use of natural resources. This includes large areas outside of protected areas, reflecting the moderate population densities and dominance of rangeland uses in this dry portion of the country. However, there are still sizeable areas with zero natural resource use in this region, associated with more densely cultivated areas. Resource use increases moving further south across the Ugandan wildlife landscape and adjacent areas of western Kenya (Figure 49-Figure 52). However, natural land cover remains relatively intact until the Mount Elgon region, which contrasts strongly with the rangeland-dominated areas further north. Due to a wetter climate, population densities are much higher here, and substantial conversion of natural habitats to cultivation has occurred. As a result, little natural habitat remains outside of the two Mount Elgon National Parks and associated protected areas, with high estimated resource use for remnant patches of natural habitat, indicating substantial harvesting pressures. As would be expected given high demand and limited availability of natural resources, high use was also estimated for the outer portions of the protected areas around Mount Elgon. This suggests adequate protection is particularly important for maintaining the remaining natural habitats in this portion of the Northern Savannas wildlife landscape.

The total value of wild harvested resources was estimated to be US$313.5 million across the landscape, US$135.3 million in Uganda, US$117.9 million in South Sudan, and US$60.3 million in Kenya (Table 39). Of the natural resources analyzed, wild plant foods and medicines had the highest monetary value per hectare (Table 39; Figure 51). This can be explained by the particularly high importance of wild foods in the drier parts of this wildlife landscape. Several productive fruit tree species are associated with the wooded grassland habitats of these regions, which provide a vital safety net function in drought and famine periods. Notably, the Northern Savannas are the only wildlife landscape where wild plant foods and medicines have a substantially higher monetary value per hectare than all other natural resources. Also, of note are the high quantities of thatching grass used per hectare, which far exceed those of the other three wildlife landscapes (Table 39; Figure 51). This reflects the prevalence of thatched roofs across much of the region, and the high stocks of thatching grass associated with the extensive wooded grassland and grassland habitats. Bamboo and fish were estimated to have a much more localized distribution than the other harvested natural resources. Bamboo is limited to the Mount Elgon region (Figure 50), where it is prized as both a construction material and a food source. Our model estimated higher use on the Ugandan side, with virtually all bamboo stocks in the region falling inside protected areas. Only small, patchily distributed fish use was predicted for the region. Palm frond use was estimated to be highly localized, as our supply layer confined the occurrence of palm wooded grassland to a very small area of northeast Uganda in the Kidepo Valley region.

Table 39. Average quantities, monetary values per hectare, and total value (US$ millions) for subsistence harvesting of wild resources in the Northern Savannas study region

RESOURCE		SOUTH SUDAN			UGANDA			KENYA		
	UNIT	USE (UNIT/HA)	US$/HA	TOTAL US$ MN	USE (UNIT/HA)	US$/HA	TOTAL US$ MN	USE (UNIT/HA)	US$/HA	TOTAL US$ MN
Fuelwood	m^3	0.54	10.25	22.8	0.31	7.44	38.4	0.60	17.33	56.9
Poles & withies	m^3	0.11	2.56	3.7	0.06	1.39	6.4	0.10	2.45	9.2
Timber	m^3	0.04	4.97	4.8	0.02	2.22	12.2	0.03	3.47	9.0
Thatching grass	kg	2.95	1.09	0.5	1.12	0.42	2.8	1.10	0.41	2.0
Bamboo	culms	-	-	0.4	0.17	0.11	-	0.95	0.62	0.4
Palm leaves	kg	< 0.01	0.01	-	< 0.01	0.01	-	-	-	0.1
Wild plant foods & medicines	kg	22.62	23.52	27.3	12.46	12.29	55.6	21.54	22.17	54.5
Bushmeat	kg	1.25	1.63	1.8	0.63	0.82	2.7	0.63	0.83	0.4
Honey	l	0.33	0.33	0.3	0.14	0.13	0.8	0.27	0.26	0.6
Fish	kg	-	-	>0.01	< 0.01	< 0.01	>0.01	< 0.01	< 0.01	>0.01

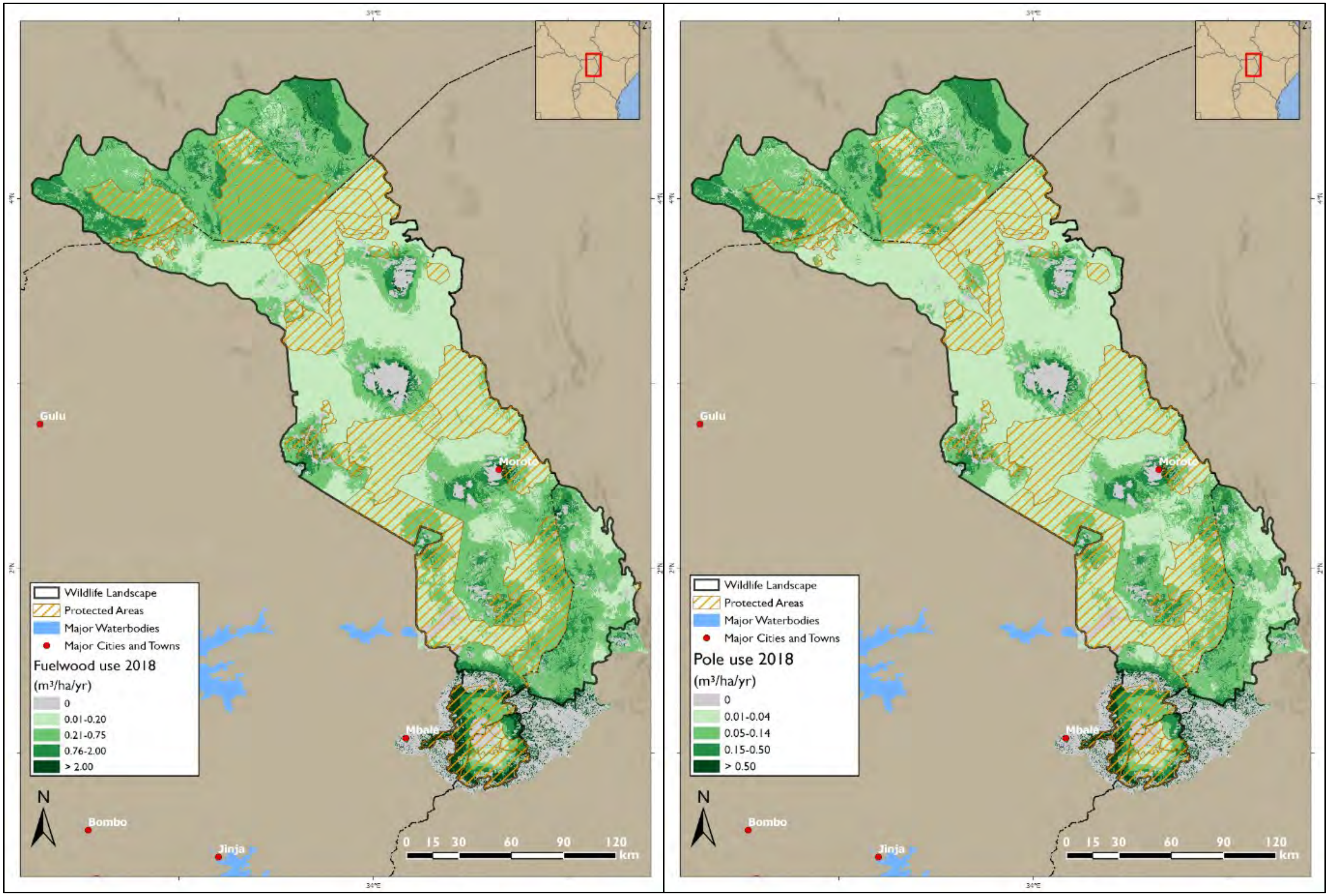


Figure 49. Estimated variation in the subsistence harvesting of fuelwood (left) and poles (right) across the Northern Savannas region

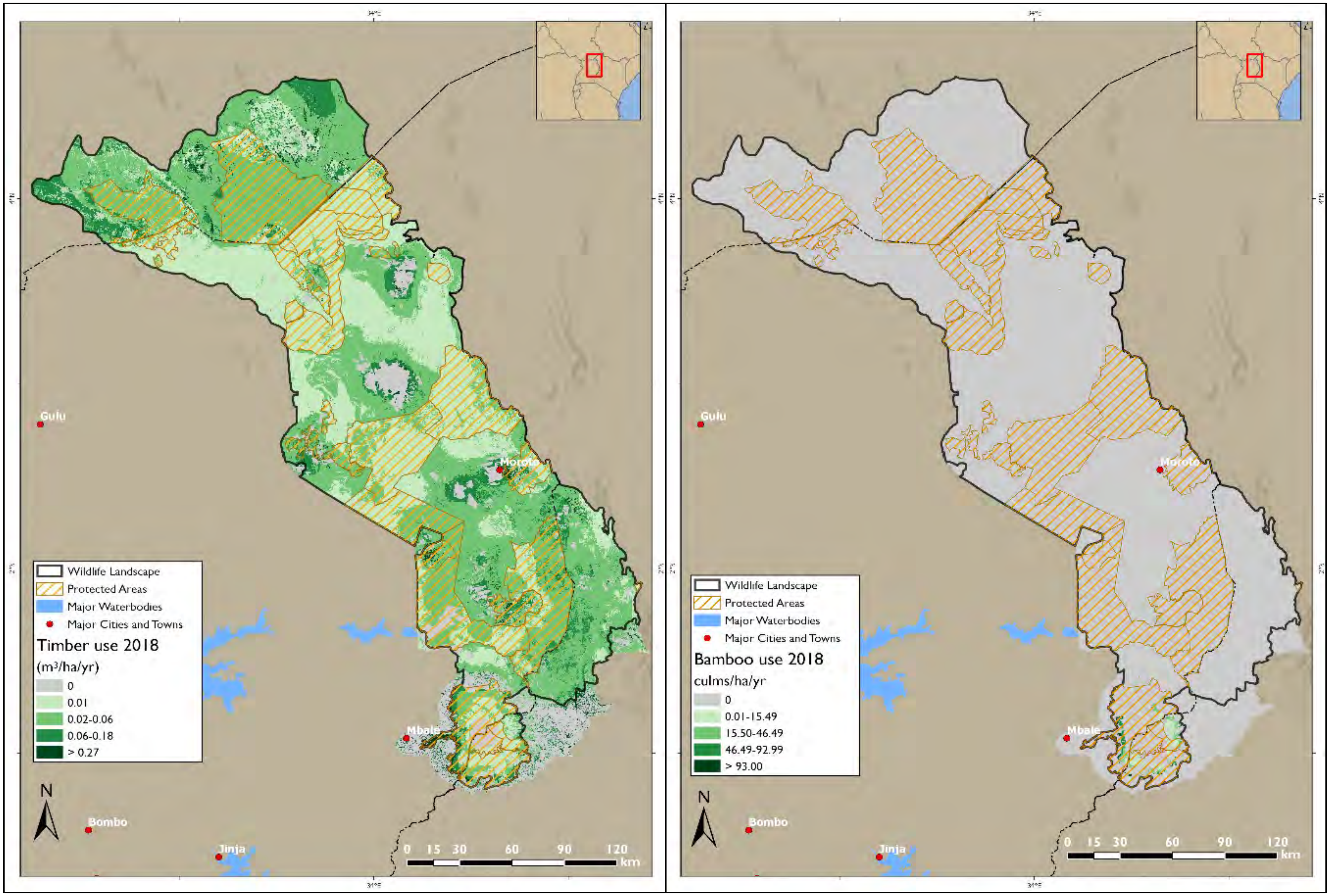


Figure 50. Estimated variation in the subsistence harvesting of timber (left) and bamboo (right) across the Northern Savannas region

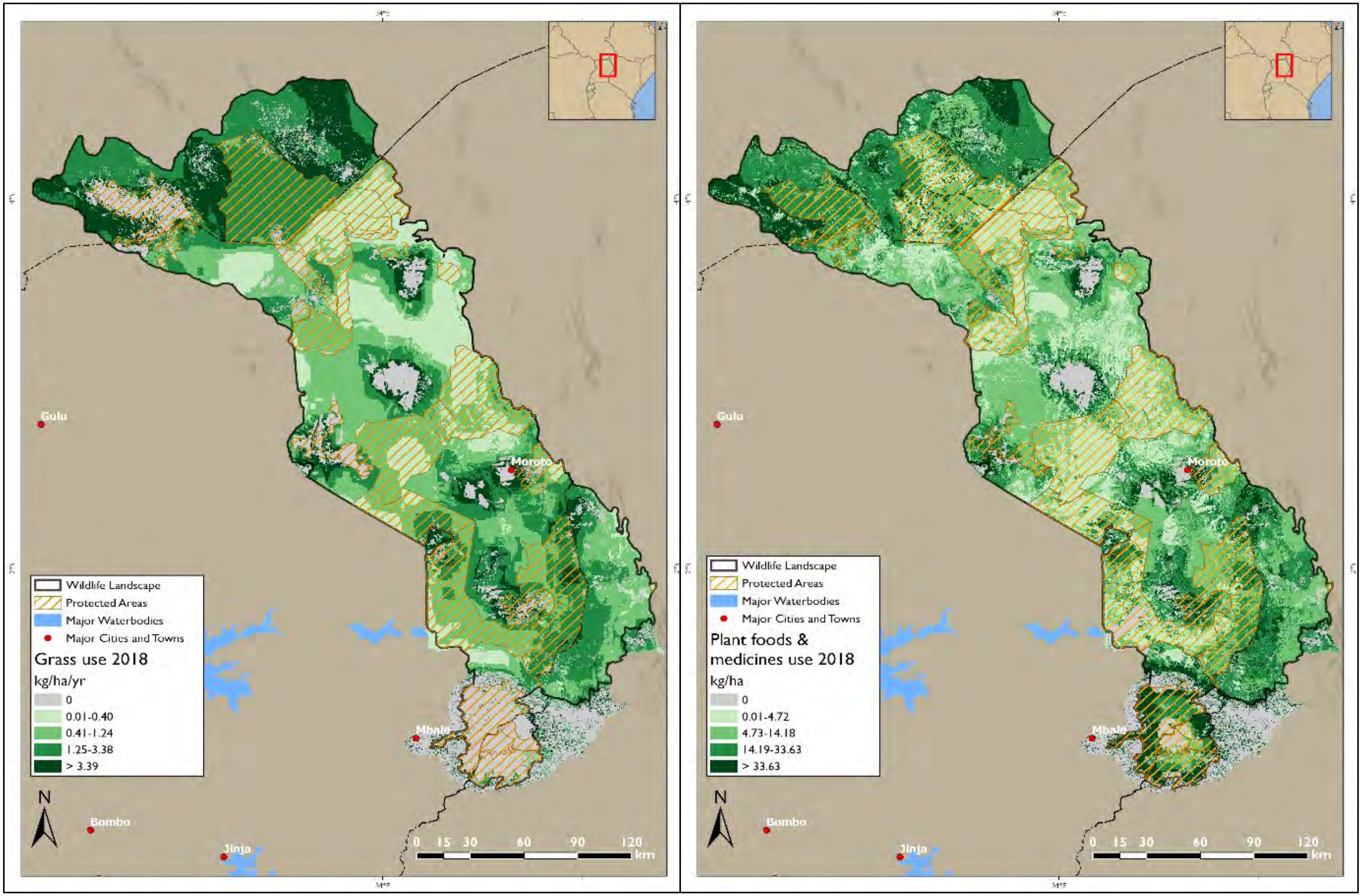


Figure 51. Estimated variation in the subsistence harvesting of thatching grass (left) and wild plant foods and medicines (right) across the Northern Savannas region

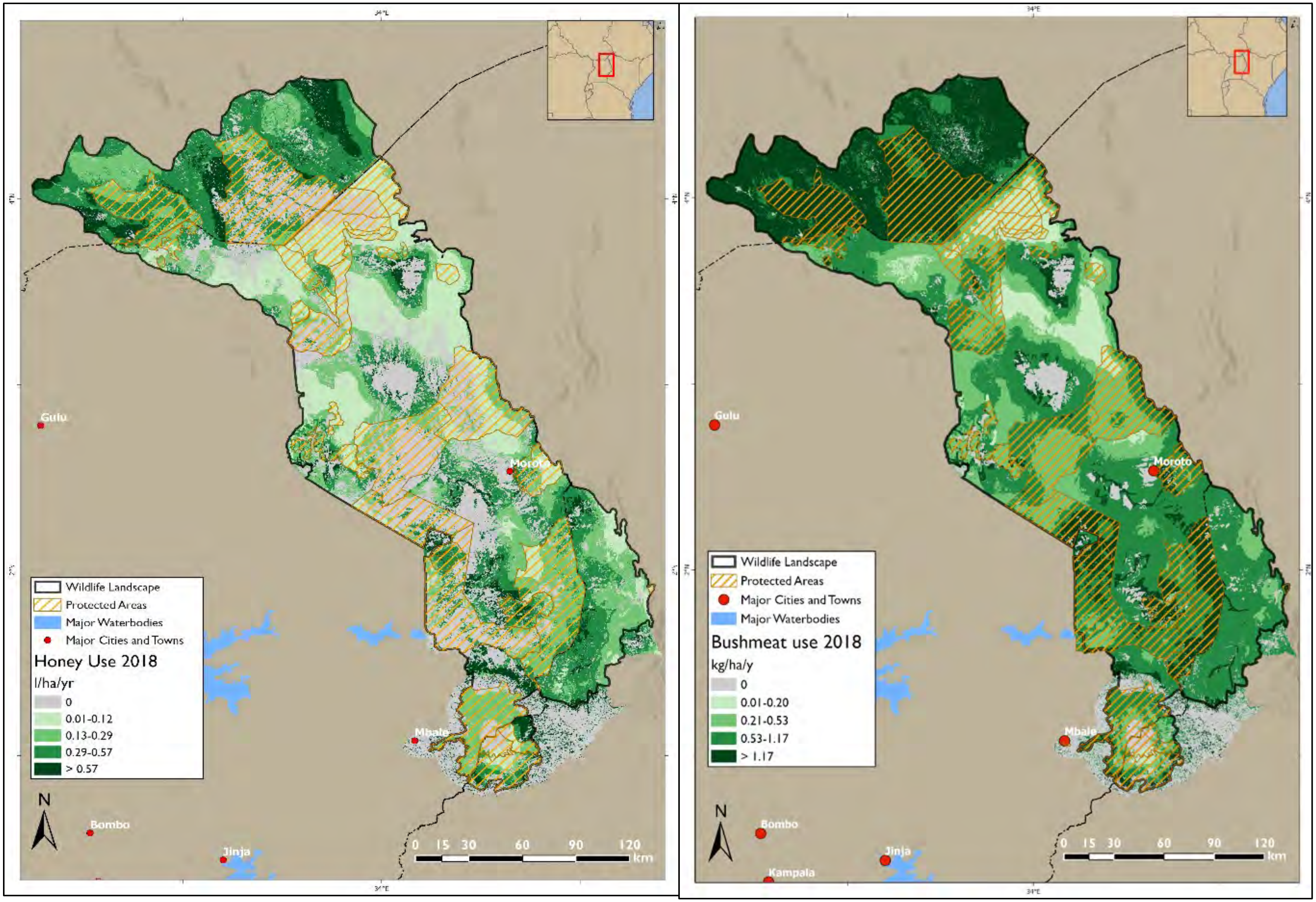


Figure 52. Estimated variation in the subsistence harvesting of honey (left) and bushmeat (right) across the Northern Savannas region

SUMMARY

The Northern Savannas wildlife landscape is a remote wilderness characterized by unique landscapes and a diverse assemblage of mammal and bird species. It is a rugged landscape where grasslands are dotted with iconic tree species such as red thorn acacias and desert dates, and sausage trees and doum palms are found along important perennial waterways. In the northern part of the study region, more than 86 mammal species can be found including leopard, cheetah, wild dog, and elephant, as well as more than 500 bird species.

The contribution of the landscape to tourism value as of 2018 was estimated to be US$8.9 million per year: US$6.6 million in Uganda and US$2.3 million in Kenya. This area of Uganda, in particular Kidepo National Park, has experienced a significant rise in the number of tourists over the last five years. While the tourism industry in South Sudan is currently non-existent, the wildlife landscapes in this country have enormous potential to generate significant nature-based tourism value in the future. This tourism also generates an estimated $3.8 million in net benefits (consumer surplus) to international visitors.

Table 40. Summary of the benefits derived from ecosystem services of the Northern Savannas wildlife landscape. All values in US$ millions per year

	KENYA	SOUTH SUDAN	UGANDA	REGION	REST OF WORLD	TOTAL
Nature-based tourism	2.3	?	6.6	8.9	11	20
Biodiversity existence	0.3	0.1	0.1	0.6	2,024	2,024
Flow regulation	46.1	112.6	356.7	515.4	-	515
Erosion control	246.8	631.0	684.1	1 561.9	-	1 562
Water quality amelioration	0.2	0.2	0.2	0.5	-	0.6
Carbon storage	57.0	55.6	147.5	260.1	149,720	149,980
Crop pollination	32.3	15.3	96.7	144.3	-	144
Livestock production	70.9	64.2	237.1	372.2	-	372
Harvested resources	60.3	117.9	135.3	313.5	-	314
Total value $ millions per year	**516.2**	**996.9**	**1,664.3**	**3,177.4**	**151,755**	**154,933**
Total value $ per ha per year	**995.6**	**890.1**	**512.7**	**650.5**	**31,067**	**31,717**

The hydrologically linked ecosystem services have significant value in this wildlife landscape. The natural vegetation was estimated to contribute some 4.6 million m^3 to annual recharge of base flows, with a replacement cost of some US$515.4 million per year. An estimated 1.3 billion tons of sediment is retained by the natural ecosystems per year, with a replacement cost value of some US$1.6 billion. Natural vegetation in the landscape reduces phosphorus loadings of some 795-1,258 tons from reaching Lake Kyoga (depending on the alternative land use), which has a replacement value of about US$503,000 to US$573,600 per year. These estimates should be refined in the future, with more detailed modeling at finer scales and with the provision of reliable monitoring data on environmental processes in Uganda, Kenya, and South Sudan. Based on satellite data, the vegetation and soils of this wildlife landscape also store an estimated 2.2 billion tons of carbon, which is estimated to avoid local climate change damages of some US$260 million per year. In addition, these carbon stocks avoid damages of some US$150 billion per year at a global scale.

The wildlife habitats of the Northern Savannas landscape also contribute to agricultural production within the landscape and around its margins. Wild pollinators in the wildlife landscape were estimated to increase crop production by some US$144.3 million per year. This estimate is based on findings from Tanzania and could be improved with panel data from the region. In addition, the natural rangelands were estimated to support livestock production worth some US$372.3 million per year.

The wildlife landscape of the Northern Savannas provides a wide array of wild resources that play an important role in supporting the livelihoods of people, especially in the northern-most parts of the study region. People living in or close to the Northern Savannas wildlife landscape were estimated to harvest some 4 million m^3 of firewood, 137,000 tons of wild fruits, vegetables, and medicinal plants, and 1.7 million liters of honey, with an estimated total value of US$313.5 million per year. Including a conservative estimate of the existence value of biodiversity, the wildlife landscape is estimated to be worth at least $650/ha/year on average to East Africa, and almost $32,000 per ha globally.

IMPLICATIONS OF THE CURRENT, POST-COVID-19 TRAJECTORY

The Northern Savannas landscape retains extensive areas of natural to semi-natural habitat. However, wildlife has been on a declining trajectory across most of the landscape for the last several decades. The landscape has a history of weak law enforcement and protected area management, which persists today and underlies many of the threats described below. A major reason is the decades of insecurity that have plagued large parts of the region. Northern Uganda was heavily affected by the conflict between government forces and the Lord's Resistance Army, which resulted in a total withdrawal of staff and management activities in certain protected areas (Nampindo, Phillipps & Plumptre, 2005). Meanwhile, Sudan experienced a protracted civil war until 2005, culminating in the formation of the independent nation of South Sudan in 2011 (Gorsevski *et al.*, 2012). A legacy of these conflicts has been a proliferation of firearms among local communities, which persists today and continues to undermine peace in the region. On the ground management and enforcement remains virtually non-existent in the South Sudan portion of the wildlife landscape (Institute of Natural Resources, 2014; A. Schenk, pers. comm.), while it remains limited across most of the protected areas in the central Ugandan portion of the landscape (Rugadya & Kamusiime, 2013). This has resulted in significant negative impacts on habitats and wildlife across much of the region. Furthermore, existing pressures are set to be exacerbated by climate change in the future. Thus, in the following sections, we start by describing some of the expected impacts of a range of existing pressures based on past trends. We then describe some of the expected impacts of future climate change derived from modelling studies. Finally, we draw together the discussion on existing pressures and future climate change impacts to predict the future of wildlife, habitats, and ecosystem services provided by the transboundary landscape under a business-as-usual scenario.

HABITAT CONVERSION TO CULTIVATION AND SETTLEMENT

Cultivation is on the rise in parts of the landscape that have been historically dominated by pastoralism (Bintoora, 2016). This has been encouraged by government and developmental organizations as a way of promoting food security (Egeru *et al.*, 2014c). A history of cattle raiding in the region has also led to some people dropping out of pastoralism in favor of cultivation. According to Egeru *et al.*, (2014c), cropland in the Karamoja sub-region of Uganda expanded from 7,500 ha in 1986 to 16,000 ha in 2013, though this is still only 0.6 percent of the total area they assessed. Nevertheless, conversion to cultivation has reduced the connectivity of the central portion of the landscape for wildlife (Bintoora, 2016). Land conversion due to the settlement of refugees in the landscape was also identified as a major cause of habitat loss by key informants.

Cultivation has caused more substantial habitat loss in wetter parts of the landscape. For example, notable deforestation has occurred in the Dongotana Hills in South Sudan (Gorsevski *et al.*, 2013). Conversion of forest may intensify in this region with relative stability, as people have been able to return to areas vacated during the civil war. The lack of park management also means unchecked encroachment of cultivation and settlement into the Imatong Forest Reserve and Kidepo Game Reserve can be expected to continue. Substantial loss of natural habitats has occurred at the southern end of the landscape, where high rainfall and fertile volcanic soils have led to high population densities in the Mount Elgon region (Sassen, 2014; Nakakaawa *et al.*, 2015). Extensive conversion of natural habitats to cultivation has already effectively eliminated landscape connectivity between Mount Elgon and the savanna regions to the north. While forest cover has remained largely intact within protected areas around Mount Elgon over recent decades, intense encroachment pressure and conflict between local people and park management are serious threats (Myhren, 2007; Petursson, Vedeld & Vatn, 2013; Nakakaawa *et al.*, 2015). Land outside protected areas has become increasingly scarce due to the dense and growing populations living adjacent to the forests, forcing people to cultivate increasingly small plot sizes and increasing pressure to convert the remaining forests. As land availability worsens with further population growth under business as usual, these threats are likely to intensify, threatening the long-term sustainability of the protected areas around Mount Elgon.

The ESA CCI land cover data at 300 m resolution suggest that while the area under crops grew rapidly in the 1990s, it has been shrinking since the early 2000s (*i.e.*, a move away from cropping; Table 41). However, again the Copernicus 100 m landcover data series contradicts this result. These data, which go back to 2015, suggest that there has been an increase in cropland in the study area from 2015 to 2018 of 5,391 hectares per year. This highlights the potential inaccuracy of land cover data products and the need for ground-truthing. Given the information in the literature, the latter trend is more likely.

Table 41. Change of land cover classes in the Northern Savannas region for 1992-2004 and 2004-2018

LAND COVER CHANGE	1992 TO 1998	1998 TO 2004	2004 TO 2010	2010 TO 2015	2015 TO 2018
Average annual change in area under crops (ha/year)	3,177	-509	-444	-208	-2,207
Average annual change in built-up area (ha/year)	16	39	0	0	-131

Source: Based on ESA CCI Land Cover 300m resolution (European Space Agency, 2018)

FUELWOOD HARVESTING

Cutting of trees for fuelwood is a driver of forest and woodland degradation in the region and appears to be on the rise. This is particularly the case in the Ugandan portion of the landscape, where increasing demand from growing urban areas to the west of the landscape has expanded the charcoal harvesting belt northwards and eastwards (Egeru *et al.*, 2014c). Poverty encourages local people to take up wood and charcoal production as an income source in response to this growing demand, particularly where they have been affected by declining livestock productivity or stock theft. Furthermore, key informants reported that climate change has pushed many people in the region to abandon pastoralism in favor of selling firewood or producing charcoal, a pressure that could intensify further if climatic conditions become more unfavorable. Additionally, key informants from local communities noted that the breakdown of cultural norms and beliefs that regulate the management of natural resources has also contributed to the problem. For example, several tree species are regarded as sacred by communities in the landscape. However, key community informants note that these same tree species now get cut down for charcoal or fuel for brick burning.

Overall, these factors have led to considerable losses of woodland across the pastoral regions of the landscape (Nampindo *et al.*, 2005; Egeru *et al.*, 2014c). Nampindo *et al.*, (2005) reported a consistent decline in woody cover between 1985 and 2002 across Uganda's Karamoja region, with Moroto and Nakapiripirit Districts in particular recording high woody cover losses of 15 percent and 36 percent, respectively. Protected areas were not immune to these declines in woody cover, with significant reductions noted in Matheniko and Bokora Wildlife Reserves, while woody cover in Kidepo Valley National Park conversely showed little change (Nampindo *et al.*, 2005). Fuelwood harvesting is also a substantial threat in the Mount Elgon region, where dense populations and extensive conversion to cultivation mean remaining forests are heavily relied upon as a source of fuelwood and other natural resources (Scott, 1998; Myhren, 2007; Sassen, 2014). Substantial degradation of the remaining forests has already occurred inside protected areas, particularly during the 1970s and 1980s on the Ugandan side, coinciding with the Amin and post-Amin regimes, and associated periods of insurgency (Petursson *et al.*, 2013). Restricted access to forest resources is a severe cause of resentment among local people, especially since these resources are scarce outside protected areas in the Mount Elgon region (Petursson *et al.*, 2013; Sassen, 2014; Nakakaawa *et al.*, 2015). These pressures on remaining forest and woodland habitats are likely to intensify with continuing increases in population and urbanization under business as usual.

OVERGRAZING

Overgrazing is another important cause of habitat degradation, chiefly in the central pastoral regions of the landscape. Traditionally, pastoralists in the area were often mobile, migrating to take advantage of varying forage and water resources, including much cross-border movement between Kenya, South Sudan, and Uganda (Mugerwa, Kayiwa & Egeru, 2014; Bintoora, 2016). This helped herders to cope with the relative aridity, high rainfall variability, and severe droughts that characterize the central portion of the landscape. However, population growth, insecurity, stock theft, and cultivation have resulted in increased sedenterization and reduced land availability for livestock (Bintoora, 2016). Together with growing livestock populations, these factors have resulted in declining productivity of many grazing regions. Furthermore, bush encroachment has emerged as a rapidly increasing threat to grassland habitats in the region, and has been related to changes in traditional nomadic pastoral practices (Egeru *et al.*, 2014c). Additionally, Egeru *et al.*, (2014) note that bush encroachment is predicted to worsen with the recent spread of *Prosopis juliflora*, an invasive alien plant species, into the landscape. According to key informants, *Prosopis* and other invasive species have reduced pasture availability and thus the livestock carrying capacity of the landscape.

Livestock grazing has also resulted in serious degradation of habitats inside protected areas, particularly those where law enforcement and management capacity are weak. For example, Matheniko Wildlife Reserve has been extensively degraded by heavy livestock grazing, which includes large herds of livestock crossing into Uganda in search of forage from the arid Turkana region of Kenya (Bintoora, 2016). Matheniko and other wildlife reserves in Uganda's Karamoja region were established with the understanding that they would serve as critical dry-season grazing areas for livestock (Government of Uganda, 2013). However, poor management and excessive livestock numbers have resulted in serious loss of forage and displacement of wildlife from these protected areas (Bintoora, 2016). Degradation of Matheniko and other wildlife reserves (Pian Upe and Bokora Corridor), and the rangelands surrounding them, has also disrupted the historical southward migration of wildlife from Kidepo Valley National Park (Government of Uganda, 2013).

HUNTING PRESSURE

Hunting has caused substantial declines in wildlife throughout the landscape. To a large degree, this has been facilitated by insecurity and poor law enforcement. For example, wildlife populations in Uganda declined drastically during the breakdown in law and order under Idi Amin in the 1970s, which compromised protected area management throughout the country (Lamprey & Michelmore, 1996; Nampindo *et al.*, 2005). The government actively encouraged people to hunt in protected areas during this period. Furthermore, civil war against Idi Amin's forces and later conflict between Yoweri Museveni's government and the Lord's Resistance Army led to a high prevalence of automatic weapons in northeast Uganda, enabling more severe poaching (Wanyama *et al.*, 2014; Ferguson *et al.*, 2019). By the 1990s, oryx and black and white rhino had all been removed from the landscape (Nampindo *et al.*, 2005), while species like giraffe were hunted to the brink of extinction (Marais *et al.*, 2016). Extirpation of most large wildlife also occurred on the Ugandan side of Mount Elgon. While populations of many species recovered or remained more stable in Kidepo Valley National Park, information on wildlife populations in the other Ugandan savanna protected areas is scarce and their status uncertain.

Due to long-term instability, recent data on wildlife populations in the South Sudan portion of the landscape is even more scarce. As a result of the years of conflict there, automatic weapons are prevalent throughout that country. Combined with weak to no protected area management, depending on the region in question, this has facilitated large-scale poaching (Institute of Natural Resources, 2014; Perry, 2020). In the Imatong Mountains, it is thought that little wildlife remains due to uncontrolled hunting (Institute of Natural Resources, 2014). In South Sudan's Kidepo Game Reserve, no protected area management has been carried out for years, and it is thought that wildlife populations have been drastically reduced there too (A. Schenk, pers. comm.). While greater management presence exists in Uganda's Kidepo Valley National Park, hunting by South Sudanese poachers and herders with automatic weapons is a threat to this park as well (Wanyama *et al.*, 2014).

On a positive note, key informants from the Kenyan portion of the landscape noted that poaching had declined with the establishment of community conservancies around Nasolot National Reserve. The reserve itself reportedly retains good populations of various wildlife species, while species such as greater kudu and zebra have become more common in the surrounding conservancy areas. The Kenyan portion of Mount Elgon also still retains sizeable populations of various large wildlife species (Petursson *et al*., 2013), with populations reported to be growing according to some key informants. Nevertheless, key informants reported that there are issues of cross-border poaching from Uganda.

Even with a relative return to stability in the region, key informants note that bushmeat hunting remains a serious threat due to poverty, the potentially lucrative nature of the bushmeat trade, and cultural practices. The presence of automatic weapons in the region also remains high, increasing the ease of hunting. Hunting pressure was noted to be particularly serious in South Sudan, where enforcement of laws against poaching remains low. Worryingly, community informants here reported that soldiers who are ostensibly meant to prevent hunting are themselves poaching wildlife in the area. As most of the landscape is arid to semi-arid, livelihood options are limited. Key informants note that in many areas, there is no alternative livelihood option to hunting, apart from livestock rearing. Increased drought and rainfall variability, combined with ongoing rangeland degradation, could further increase reliance on bushmeat as a livelihood option. Key informants have recommended participatory conservation programs which involve local communities as a potential response, with community members in some areas noting that community scouts have played a significant role in reducing poaching. Given the cross-border nature of bushmeat poaching in many areas, key informants also call for improved cross-border conservation efforts. For example, key informants from the Kidepo region lamented the absence of a cross-border program to coordinate transboundary conservation between South Sudan and Uganda to help mitigate this threat. More equitable sharing of wildlife conservation benefits with local communities was also recommended as a way of reducing poaching.

HUMAN WILDLIFE CONFLICT

Human-wildlife conflict (HWC) is another serious issue in the region, negatively affecting local communities due to loss of crops and even personal injury or death. At the same time, it has negative impacts on wildlife through promoting negative perceptions of wildlife, potentially prompting local communities to try solve the issue through retaliatory killings and elimination of wildlife populations. Community informants identified elephants as major raiders of crops, while leopards, hyenas, and cheetahs pose a threat to livestock. An informant from the Kidepo region reported that scarcity of water during droughts is another factor that causes wildlife to move out of protected areas into adjacent community lands. Ugandan informants felt the government has not put in place adequate measures to address HWC, lamenting the fact that numerous people had been killed by wildlife in Karenga and Kabong Districts, yet no compensation had been provided. Reportedly, around three people are killed by wildlife each year in the communities surrounding Kidepo Valley National Park.

Similar resentment was expressed by community key informants from South Sudan, who tended to view wildlife as a danger and nuisance. They also complained that they receive no assistance from the government for dealing with problem animals, nor any compensation for damage done. This does not bode well for the future of wildlife populations in the area, especially as communities in South Sudan are not receiving any benefits from nature-based tourism that might help them to view wildlife in a more positive light.

Throughout the landscape, HWC and resulting resentment toward wildlife could intensify in the future as populations living adjacent to wildlife areas grow and encroachment pressures into remaining wildlife habitat worsen. Although it has negative impacts on migratory routes and landscape connectivity, fencing has helped to reduce HWC in parts of the landscape according to key informants, most notably around

Mount Elgon on the Kenyan side. Community informants here noted that prior to construction of the fence, farmers had to secure their fields from wildlife in shifts, which exerted a large time burden on households.

PROJECTED CHANGES IN TEMPERATURE AND RAINFALL

Total annual precipitation in the Northern Savannas landscape for the period 2040–2060 is expected to increase by just over 13 percent relative to historical (1960–1990) precipitation, with decreased rainfall in June predicted and significantly increased winter rainfall (December to March, Figure 53). Mean annual temperature across the Northern Savannas landscape is expected to increase by 2.7°C on average across the year (Figure 55) with the summer months of June to August expected to increase the most, by almost 3°C (Figure 54).

The Northern Savannas landscape is predicted to experience a large shift in mean annual temperature and the greatest changes precipitation in the next few decades of all four wildlife landscapes. Changes in mean annual temperature are relatively consistent across the landscape, with key protected areas predicted to have mean annual temperature increases of 2.6-2.7°C (Figure 55, Table 42). Changes in precipitation are more variable across the landscape, with predicted changes ranging from a 4.0 percent increase in Kidepo Game Reserve in South Sudan to as much as an 8.6 percent increase in mean annual precipitation in Chepkitale National Reserve in Kenya. This suggests that rainfall will increase more in the southern part of the region and decrease with an increase in latitude toward the northern part of the landscape (Figure 55).

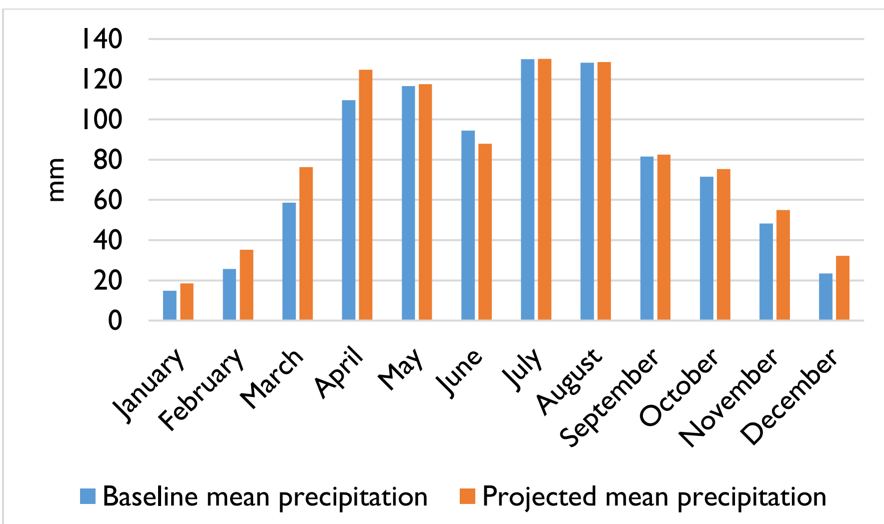


Figure 53. A comparison between historic and projected mean monthly precipitation (mm) for the Northern Savannas landscape

Source: Based on data from WorldClim Version2 and CMIP5

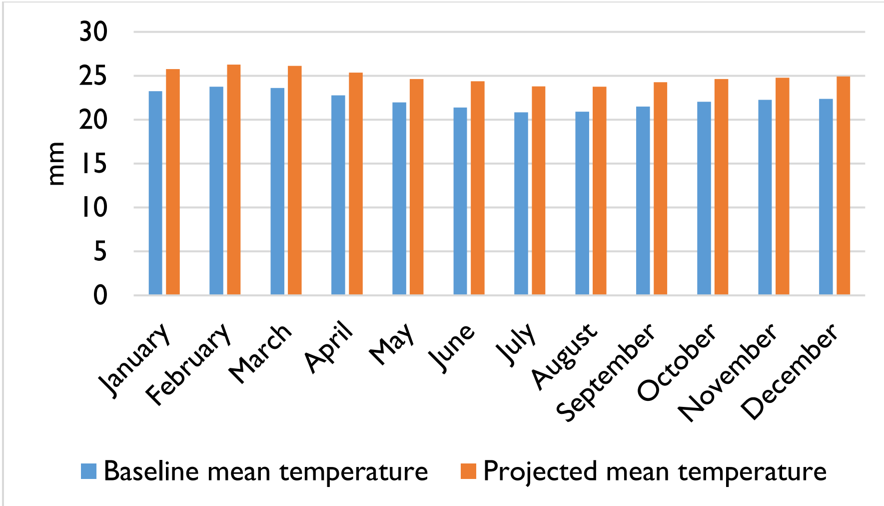


Figure 54. A comparison between historic and projected mean monthly temperature (°C) for the Northern Savannas landscape

Source: Based on data from WorldClim Version2 and CMIP5

Table 42. Historic, projected, and percentage changes for mean annual temperature (°C) and total annual precipitation (mm) for key protected areas in the Northern Savannas study area

	MEAN TEMPERATURE (°C)			MEAN PRECIPITATION (MM)		
PROTECTED AREA	HISTORIC ANNUAL AVG.	PROJECTED ANNUAL AVG.	CHANGE	HISTORIC ANNUAL TOTAL	PROJECTED ANNUAL TOTAL	% CHANGE
Kidepo	24.2	26.9	2.7	737	767	4.0
Pian Upe	22.9	25.5	2.6	1 090	1 183	8.5
Bokora Corridor	23.5	26.2	2.7	837	901	7.6
Matheniko	22.9	25.6	2.7	660	718	8.8
Kidepo Valley	22.6	25.3	2.7	671	704	4.9
Mount Elgon (Uganda)	13.6	16.2	2.6	1 680	1 822	8.4
Chepkitale	9.6	12.2	2.6	1 721	1 869	8.6
Mount Elgon (Kenya)	12.4	15.0	2.6	1 585	1 717	8.3

Source: Based on data from WorldClim Version2 and CMIP5. Protected areas are listed in descending order of area.

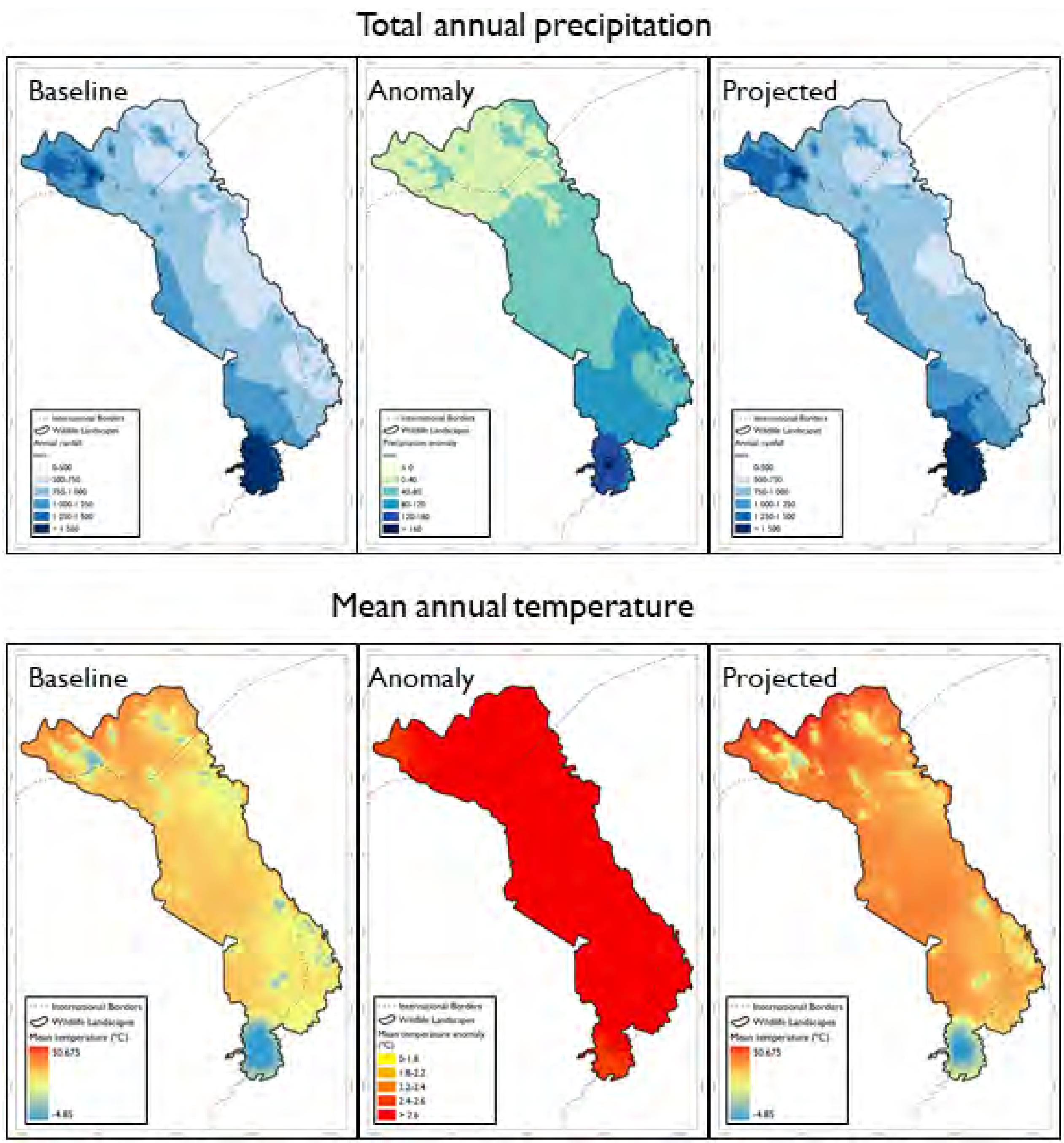


Figure 55. Baseline/historic (1960 – 1990) and projected (2040 – 2060) total annual precipitation (mm) and mean annual temperature (°C) across the Northern Savannas landscape

Source: Based on data from WorldClim Version2 and CMIP5

PROJECTED CLIMATE CHANGE IMPACTS ON WILDLIFE

Using the available species distribution models (SDMs), the expected combined species richness of mammals, birds, reptiles, and amphibians are shown in Figure 56. The maps indicate species richness under current conditions, and under the projections of three different climate models for 2070 (models ac, bc, and cc), which shows the range in results depending on which future climate model one uses. The term "expected" species richness is used because species ranges have been altered by anthropogenic land use and pressures, meaning that real species richness will be substantially lower in regions where natural habitats have been transformed.

According to the SDMs used, areas with highest expected species richness are associated with higher-lying, wet areas, particularly in the northern part of the landscape. These include the Imatong Mountains and mountainous areas in the far northeast of Uganda. Large declines in species richness are predicted across the majority of the landscape under all three future climate models. This includes the biodiverse mountainous regions in the north of the landscape, which had high expected species richness under current conditions. Conversely, increases in species richness are predicted for Mount Elgon and the lower-lying extreme northern parts of the study region under some (but not all) of the future climate scenarios. This divergence in the predictions highlights the inherent uncertainty in both SDMs and future climate models. Nevertheless, under all future climate scenarios, the SDMs predict most of the landscape will experience declining species richness.

To get a more detailed understanding of the impacts of climate change, predictions for individual key charismatic species, such as lion and elephant, were also investigated. Ranges of a number of these species were predicted to contract substantially under future climates, particularly in the central and northern parts of the landscape. Worryingly, this includes Kidepo Valley National Park, the main remaining stronghold for savanna wildlife in the landscape. The predicted distributions of charismatic wildlife species under climate change are presented and discussed in more detail in Appendix 5.

Notwithstanding the inherent uncertainties in the models, these results suggest that future climate change presents a severe threat to wildlife in the region, given that substantial declines in species richness are predicted across most of the landscape from climate change alone. This includes significant reduction in the ranges of key charismatic species, which would have a negative impact on the potential for wildlife tourism in affected regions. The severity of the threat posed by climate change also provides further reason to mitigate other pressures currently threatening wildlife in the landscape. In particular, it highlights the need to maintain migration corridors and dispersal routes, as these are crucial for allowing wildlife to move in response to climate stresses. The models also suggest securing the long-term future of certain charismatic species like elephant and cheetah may require improving the status of protected areas in the central and southern parts of the landscape, as conditions in Kidepo Valley National Park may become increasingly unsuitable for these species under future climates.

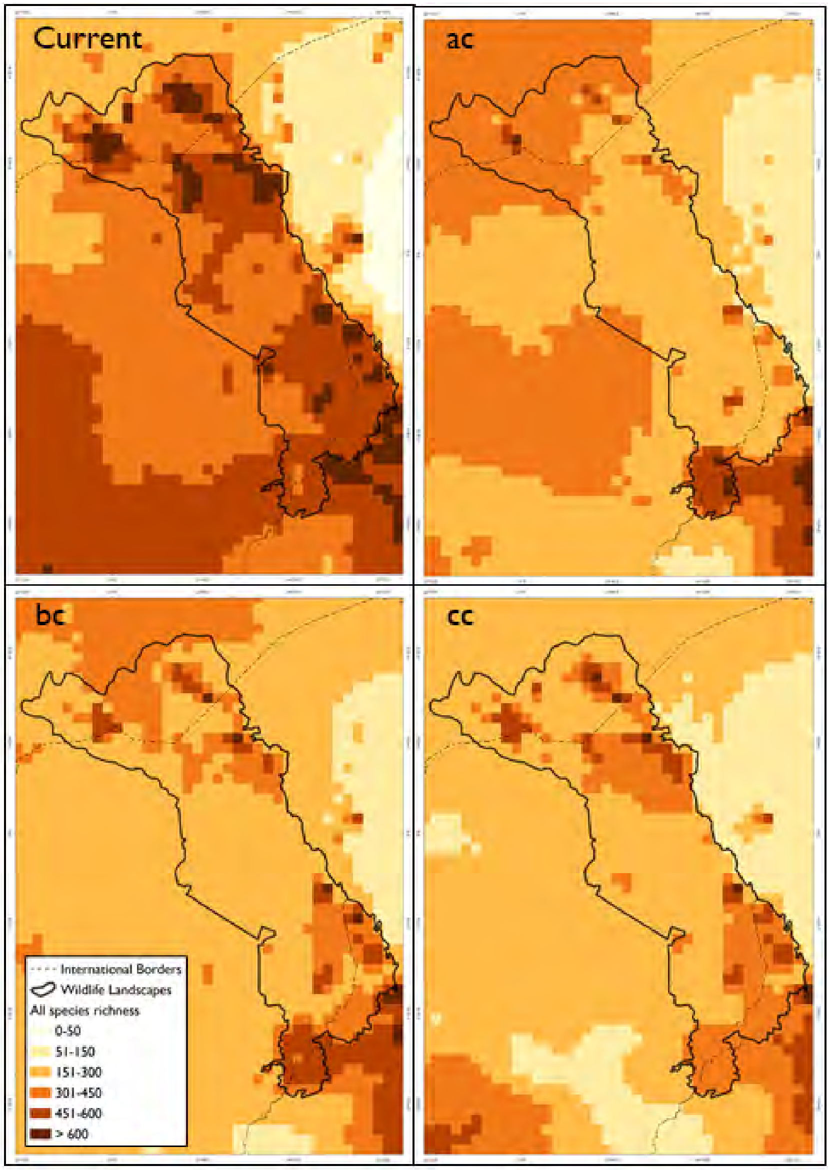


Figure 56. Current geographic variation in species richness (amphibians, birds, mammals, and reptiles) for the Northern Savannas landscape, followed by the projected species richness pattern for each of three climate models used

Source: Based on modelled species distributions from Conservation International

PROJECTED CLIMATE CHANGE IMPACTS ON SUITABILITY FOR CROPS

Six widely grown crop species were modelled using FAO's EcoCrop analytical tool (Table 43, Figure 57). This tool produces estimates of crop suitability under current and future conditions. Suitability is described in terms of the suitable area for a given crop (*i.e.*, the region with a suitability score of greater than 0), as well as the relative suitability value, which ranges from 0 (unsuitable) to 1 (optimum conditions). In general, suitability of the landscape for cultivation of various crops species was predicted to increase by 2050, relative to current conditions. This is particularly the case for cereal crops, where both suitable area and relative suitability scores consistently increased under future conditions. More favorable conditions for crop growth in the future may encourage further expansion of cultivation into the pastoral regions of the landscape, causing further habitat loss and fragmentation.

Table 43. Summary of the expected changes in the suitable area and suitability scores for crops in the Northern Savannas landscape and immediate surrounds, based on the maps shown in Figure 57

CROP	PRESENT	FUTURE
Beans	High suitability across virtually all of the landscape, with the upper slopes of Mount Elgon the only area not suitable for growth.	Virtually all of the landscape remains suitable, while unsuitable area on Mount Elgon expands upslope. However, suitability scores do decline somewhat over the central and southern regions of the landscape.
Cassava	Most of the landscape suitable, aside from higher-lying areas. However, areas with high suitability generally limited to the western regions of the landscape.	Suitability scores increase, with an eastward expansion of moderate to high suitability conditions. Unsuitable area expands further upslope, thus increasing suitable area in mountainous regions.
Maize	Suitability mostly limited to the western part of the landscape, with isolated suitable areas associated with higher-lying land elsewhere.	Substantial eastward expansion of suitable area, along with a general increase in suitable area and suitability scores in higher-lying areas.
Millet	High suitability across most of the landscape, with only the upper slopes of mountainous regions unsuitable.	Suitable area expands upslope, accompanied by a general increase in suitability throughout the landscape.
Potato	Virtually all of the landscape suitable, except for the upper slopes of Mount Elgon.	Virtually all of the landscape remains suitable, while unsuitable area on Mount Elgon expands upslope. However, a slight decline in suitability scores occurs across much of the landscape.
Sorghum	Virtually all of the landscape suitable, except for the upper slopes of Mount Elgon.	Suitability score increases throughout the landscape, while suitable area expands upslope on Mount Elgon.

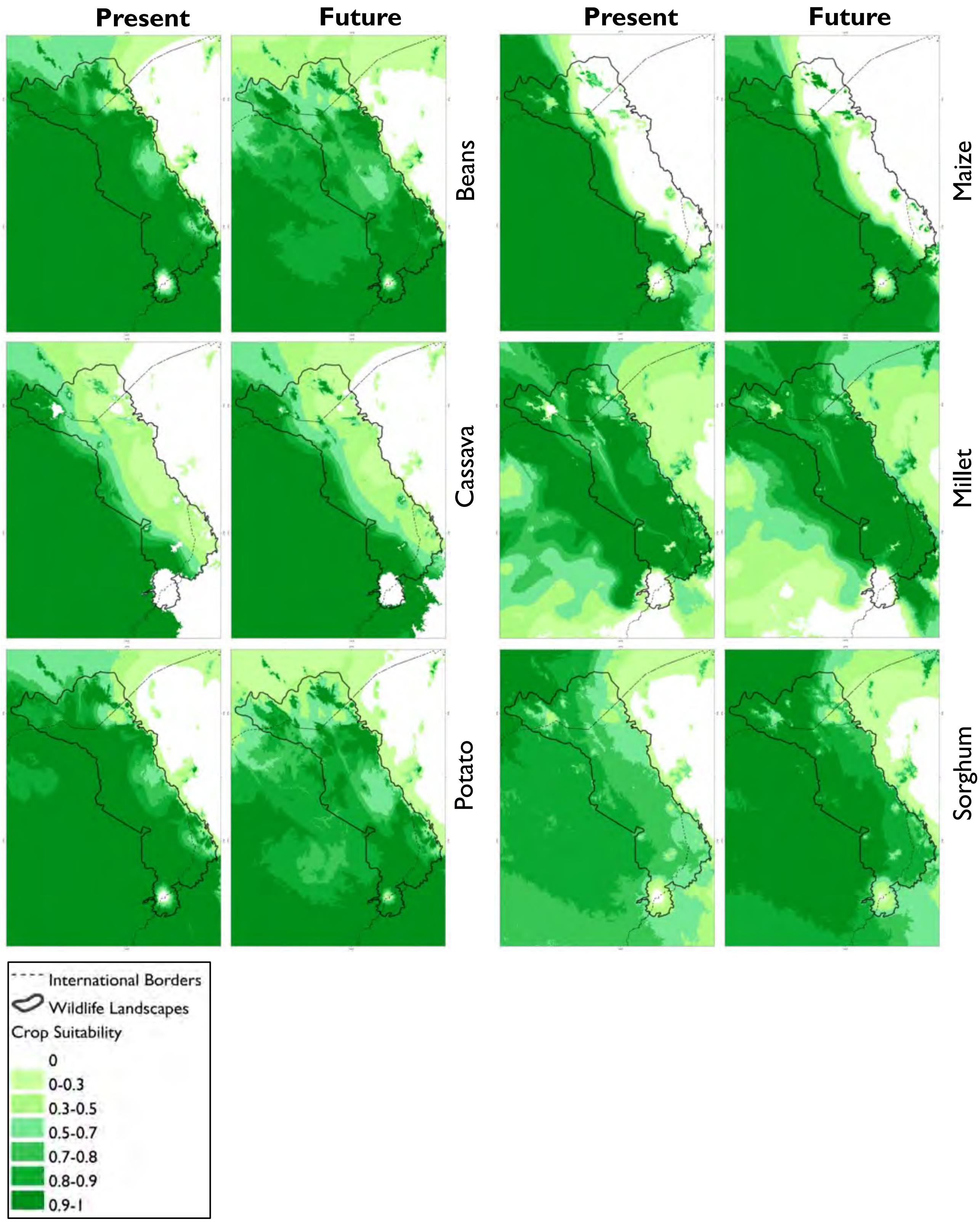


Figure 57. Estimated present and future suitability for some of the key crops grown in and around the Northern Savannas landscape. Model outputs generated using the FAO EcoCrop database and model and climate projections for 2040-60

POTENTIAL IMPLICATIONS OF A BUSINESS-AS-USUAL SCENARIO

This section provides an integrated, qualitative assessment of the impacts of a BAU scenario on wildlife, ecosystem services, and human wellbeing over the period from the baseline (2018) to 2030. The combination of 1) increasing population and demand for land and resources and 2) the impacts of climate change on habitats, species, and agriculture need to be considered. There is a great deal of uncertainty in this. Notwithstanding these caveats, the following impacts could be expected.

Agricultural expansion could continue to reduce and fragment wildlife habitats. Cultivation will likely continue to spread through the landscape due to population growth and the encouragement of governments and developmental agencies, as well as from the desire of local people to diversify and increase their income. Climate change could be a further driver of agricultural expansion in areas where conditions become more favorable for cultivation. This will erode remaining migration routes and dispersal areas outside protected areas, and increase conversion pressures within protected areas, especially in those where law enforcement is weak. The net result will be increased genetic isolation of wildlife populations in protected areas, and reduced ability for wildlife to migrate in response to threats like drought and climate change. Overall, this threatens the long-term viability of wildlife populations across the region. In particular, the increased isolation of Kidepo Valley National Park as the only remaining stronghold for savanna wildlife could significantly reduce the capacity of species to respond to climate change. The prediction of future expansion of cultivation may prove to be particularly challenging for this landscape, with conflicting information on trends coming from different land cover products and literature case studies. Based on what was judged to be the most reliable estimates of land cover change (Osaliya *et al.*, 2019), it is predicted that the area of cultivation would increase from 5.1 percent of the landscape in 2018 to 7.4 percent by 2050 under a BAU scenario. This would result in the conversion of an additional 109,000 ha of habitat, with implications for baseflow, water quality, soil erosion, and tourism as described below.

Habitat degradation is likely to worsen due to increased overharvesting and overgrazing. Ongoing population growth will increase demand for fuelwood, including growing demand for charcoal as urbanization increases. Based on population trends, it was predicted that the **demand for woody resources could increase by about 35 percent by 2050** under a BAU scenario. This will lead to further degradation of woody habitats and may increase harvesting in protected areas as natural resources become scarce elsewhere. Increased competition with cultivation and settlement could worsen rangeland health, compromising important foraging resources for both wildlife and people. The emergence of alien species (*Prosopis*) in the landscape could also lead to increasingly rapid rangeland degradation, as it worsens bush encroachment into grazing areas.

Livestock numbers could increase by 65 percent and 224 percent in the Kenyan and Ugandan portions of the landscape, respectively, while the **trend is more uncertain for South Sudan due to the country's security situation**. This might be seen as an opportunity for economic growth, particularly for the Ugandan portion of the landscape, which holds about 20 percent of the country's total livestock value and accounts for 0.9 percent of national GDP (the total value of all livestock in Uganda is 4.2 percent of GDP). Livestock grazing in the landscape accounts for a much smaller portion of national GDP in Kenya (0.1 percent) and South Sudan (0.5 percent), which is partly due to the relatively small areas of these countries captured in the wildlife landscape. Against the potential for economic benefits, the predicted increases in livestock numbers could also lead to further degradation of areas already subject to overgrazing, whilst spreading the extent of overgrazing to new parts of the landscape. Furthermore, this assessment found that ecosystem services like soil erosion control and flow regulation have a higher value per hectare than livestock grazing. While the intensification of livestock would not lead to a total loss of these hydrological services, their value could still be decreased significantly by degradation of rangeland areas due to overgrazing, as the loss of vegetation cover reduces the ability of the landscape to retain sediment and to capture and release runoff. Investment in maintaining landscape condition through conservation of remaining natural areas could thus be worthwhile for keeping these high value ecosystem services in optimum condition. At the same time, poor rangeland condition outside protected areas would increase livestock encroachment into protected areas, particularly into those where management capacity is weak, reducing forage quality, displacing wildlife, and potentially degrading the value of the ecosystem services provided by untransformed habitats. These encroachment pressures could be heightened by an increase in rainfall variability and drought under climate change.

Tolerance for wildlife and conservation could decrease. Continuing increases in rainfall variability and drought could also lead to more intense human wildlife conflict in the future. Predation of livestock tends to increase during droughts, while herbivorous wildlife may increasingly resort to foraging in fields and around villages as alternative food sources disappear (Ogutu *et al.*, 2014). This is likely to lead to efforts to kill problem animals. Human-wildlife conflict will also likely worsen as human populations, livestock, and cultivation increase.

Poaching could increase, significantly affecting wildlife populations. Poaching could increase with reduced opportunities for income from crops and livestock and tourism. Crop and livestock income will likely be reduced by 1) increased droughts and 2) increased competition for land. Crop failures and livestock deaths increase people's reliance on bushmeat and other natural resources during and after droughts. Wildlife may also become easier to hunt, as animals become weakened and more vulnerable, or move closer to villages in search of food (Ogutu *et al.*, 2014; Knapp *et al.*, 2015). Together, these factors would increase stress on wildlife already struggling to cope with future climate change and variability. Based on population growth alone, it was estimated that **demand for bushmeat could increase by 30 percent across the landscape by 2050** under a BAU scenario. However, given the other pressures mentioned, bushmeat demand in reality could increase beyond this estimate by 2050. While not necessarily as attention-grabbing as high-profile poaching of rhino horn and ivory, increasing bushmeat demand presents an equally severe threat as it has the potential to undermine the broader general wildlife population, more so than targeted poaching of elephant and rhino. The threat of increased bushmeat demand could be particularly serious in the Northern Savannas landscape, as much of it is already characterized by dwindling or absent wildlife populations. An increase in bushmeat hunting could also increase the risk of novel zoonotic disease transmission, due to the greater numbers of people coming into contact with meat from wild species (Wolfe *et al.*, 2005). This risk would be accentuated by likely increases in human population densities from population growth and the expansion of settlements (Rohr *et al.*, 2019).

Severe declines in wildlife populations have already resulted from the combined effects of these pressures across most of the landscape. Kidepo Valley National Park in Uganda and Kenya's Mount Elgon National Park are the only parts of the landscape where significant populations of large wildlife are known to remain. Even here, wildlife populations have been severely affected. For example, oryx and rhino went extinct in Kidepo Valley National Park, while the giraffe population remains very small, even after translocations from Murchison Falls National Park (Nampindo *et al.*, 2005; Ferguson *et al.*, 2019).

While no recent data exists, it is generally thought that little large wildlife remains in the South Sudan portion of the wildlife landscape (Institute of Natural Resources, 2014; A. Schenk, pers. comm.). Little information on the current status of the various Ugandan wildlife management areas was found, but indications are that many species remain scarce or were eliminated completely starting in the Idi Amin era. Recovery of remaining wildlife populations in these protected areas remains uncertain, given limited management capacity as well as conflict and uncertainty about the tenure status and extent of these protected areas among both management and local people (Rugadya & Kamusiime, 2013). Large wildlife also remains absent from the Ugandan side of Mount Elgon (Petursson *et al.*, 2013).

The potential overall effects of the above pressures on wildlife and wildlife habitats on ecosystem services under a BAU scenario can be summarized as follows (see Table 44).

Nature-based tourism is expected to decrease, and future recovery is uncertain due to insecurity and climate change. Tourism revenue is already modest in the region. As elsewhere, the COVID-19 pandemic has had a substantial impact on the tourism that was present due to restrictions on international travel. Weak management capacity in many of the landscape's protected areas, combined with encroachment pressures from livestock, settlement, and cultivation, mean that the attractiveness of a number of protected areas for wildlife tourism could decline under a BAU scenario. The volatile security situation, particularly in the northern parts of the landscape, also means there is a high risk of the fragile tourism industry declining or disappearing with any increase in insecurity. Climate change could also be an increasing threat to wildlife in Kidepo Valley National Park, the current focal point for wildlife tourism in the landscape. In light of these factors, it was predicted that there would be an initial rise in tourism once the effects of the COVID-19 pandemic abate, but that tourist numbers would quickly plateau and eventually start declining by 2050 under a BAU scenario. This is due to the aforementioned pressures on protected areas and weak management capacity to respond to these across much of the region. It was predicted that **annual tourism value would decline by US$1.45 million in the Ugandan portion of the landscape (22 percent decline) and US$280,000 in the Kenyan portion (12 percent decline)**. The smaller relative decline in the Kenyan portion reflects the relatively effective management and protection of Mount Elgon National Park, the primary tourist attraction in the Kenyan portion of the landscape. Since current tourist value in the South Sudan portion is virtually non-existent and the future growth of the industry uncertain, there was no basis from which to make growth predictions here.

Erosion and sedimentation are expected to increase. Denudation of rangeland from overgrazing and conversion of natural habitats to agriculture will increase erosion due to a loss of vegetation cover. This will result in further loss of value of downstream aquatic ecosystems affected by increased sedimentation, including lost storage capacity and lowered water quality in critical water sources for people and livestock in the drier parts of the region. Using the predicted rate of agricultural expansion, it was estimated that the **capacity of the landscape to retain sediment and control erosion could decrease by 0.4 percent by 2050** under a BAU scenario, with an additional 4.8 million tons of sediment entering rivers and waterbodies. If the capacity of the landscape to control soil erosion was reduced in this way, the cost in terms of lost reservoir storage capacity and the greater need for sediment clearance is estimated to be around **US$6 million per year by 2050**.

Water availability in the dry season is expected to decrease. Further woody cover loss, rangeland denudation, cultivation, and settlement will result in lower dry season baseflows, as the reduction or loss of vegetation cover means rainfall runs off quickly instead of being stored and released more gradually. Again, this will be a particularly serious issue in the drier parts of the region where water is already scarce. It was predicted that **baseflow could decline by 2.5 percent by 2050** under a BAU scenario, primarily due to the high water requirements of the expanded areas of cultivation. This represents a loss in baseflow of 205 million m^3 relative to the current landscape. If the capacity of the landscape for infiltration and release of flows were reduced in this way, **the cost of reservoir storage to retain this amount of water was estimated to be US$23 million by 2050**.

Freshwater ecosystems are expected to become more polluted. Degradation and clearance of vegetated areas downstream of agricultural fields will reduce the capacity of the landscape to retain nutrients in agricultural runoff. Conversion of natural habitats to agriculture will also increase nutrient inputs into the landscape due to increased fertilizer use. This will increase nutrient loads and eutrophication of rivers, wetlands, and lakes, reducing the value of these habitats. For the portion of the landscape that drains into Lake Kyoga, it was estimated that **phosphorus export would increase by 4.7 percent by 2050** in a BAU scenario, representing an additional 31,000 t of phosphorus export. If nutrient export from the landscape increased in this way, **annual water treatment costs would rise by US$223,000 by 2050.**

The landscape is expected to contribute to further local and global climate change. Increased harvesting of woody habitats to meet growing demands for fuelwood and building materials will release carbon stored in vegetation, as will habitat conversion. Rangeland degradation and habitat conversion will also release carbon stored in soils. It was estimated that **carbon storage in the landscape could decline by 0.5 percent (10.7 MtC) by 2050** under a BAU scenario. This would amount to an annual cost to the region of US$560,000 in climate-change-related damages.

The landscape's capacity to support agricultural livelihoods is expected to be compromised, affecting the ecological integrity of protected areas. The provision of services such as crop pollination will decrease as cultivation expands and becomes more intensive, with a reduction or disappearance of natural vegetation patches between fields. In addition, forage for livestock production is likely to remain poor or decline further in already degraded areas. Elsewhere, rangeland degradation may expand as human population growth and cultivation reduce available land for livestock. This will drive increased incursion of livestock into protected areas, already a significant problem in this landscape. Stocks of harvested resources will decline outside of protected areas due to habitat conversion and growing demand from increasing populations. Woody resources in particular are under threat from increasing urban charcoal demand. Shortages of natural resources outside protected areas may increase harvesting pressures within protected areas, especially in densely populated regions and/or where enforcement capacity is weak or non-existent.

Table 44. Estimated changes in the value of ecosystem services and water treatment costs by 2050 caused by land use changes under a BAU scenario for the Northern Savannas landscape. For services with a global value, both total value to the world and value to the East African region only are shown (latter value in parentheses).

ECOSYSTEM SERVICE	CURRENT VALUE (US$)	2050 VALUE (BAU) (US$)	% CHANGE
Nature-based tourism	20.2m (8.9m)	16.2m (7.2m)	-19.5
Biodiversity existence	2,024.8m (0.6m)	1,973.6m (0.5m)	-2.5
Flow regulation	515.4m	492.5m	-4.4
Erosion control	1,561.9/m	1,556.0m	-0.4
Carbon storage	150.0b (260.1m)	149.5b (258.8m)	-0.5
Water treatment costs	481.5k	557.8k	+1.3

THE ALBERTINE RIFT FORESTS

FEATURES AND LOCAL CONTEXT

WILDLIFE AND WILDLIFE HABITAT

This study region encompasses the Albertine Rift, one of the most biodiverse areas on the planet and part of the Eastern Afromontane Hotspot of Biodiversity (Figure 58). Species richness is exceptionally high, with over 50 percent of Africa's bird species and 40 percent of the continent's mammals found there. This diversity is all the more remarkable considering the region accounts for just 1 percent of Africa's surface area (Plumptre *et al.*, 2016). Importantly for conservation purposes, the Albertine Rift also holds more endemic and globally threatened vertebrates than any other region in mainland Africa (Plumptre *et al.*, 2007).

The Albertine Rift Forests wildlife landscape is found within the western arm of the Great Rift Valley, also known as the Albertine Rift. Rugged, mountainous terrain dominates this region, which includes the Rwenzori Massif, the third highest peak in Africa. Several lakes occupy the deep valley floor, the largest of which is Lake Tanganyika, Africa's deepest lake. The Virunga volcanoes of northern Rwanda and southwestern Uganda mark the divide between the Nile and Congo watersheds, with the direction of drainage along the Albertine Rift reversing at this point (Marchant *et al.*, 2018). To the north of the Virunga volcanoes, rivers drain northwards, while from Lake Kivu southwards drainage occurs in a southerly direction.

A diversity of natural habitats can be found across the study region (Figure 58). Montane forest is the dominant natural vegetation type in the region, though human activities have substantially reduced forest cover in the area. While only pockets of montane forest remain today, this vegetation type would once have covered most of the study region in Rwanda, Burundi, and southwest Uganda. In the far north of the study area, forest is interspersed with areas of savanna, such as along the shores of Lake Albert and in Uganda's Queen Elizabeth National Park (Figure 59). At high elevations, heather and alpine moorland can be found, such as on the Rwenzori Massif, which tops out at 5,100 m.

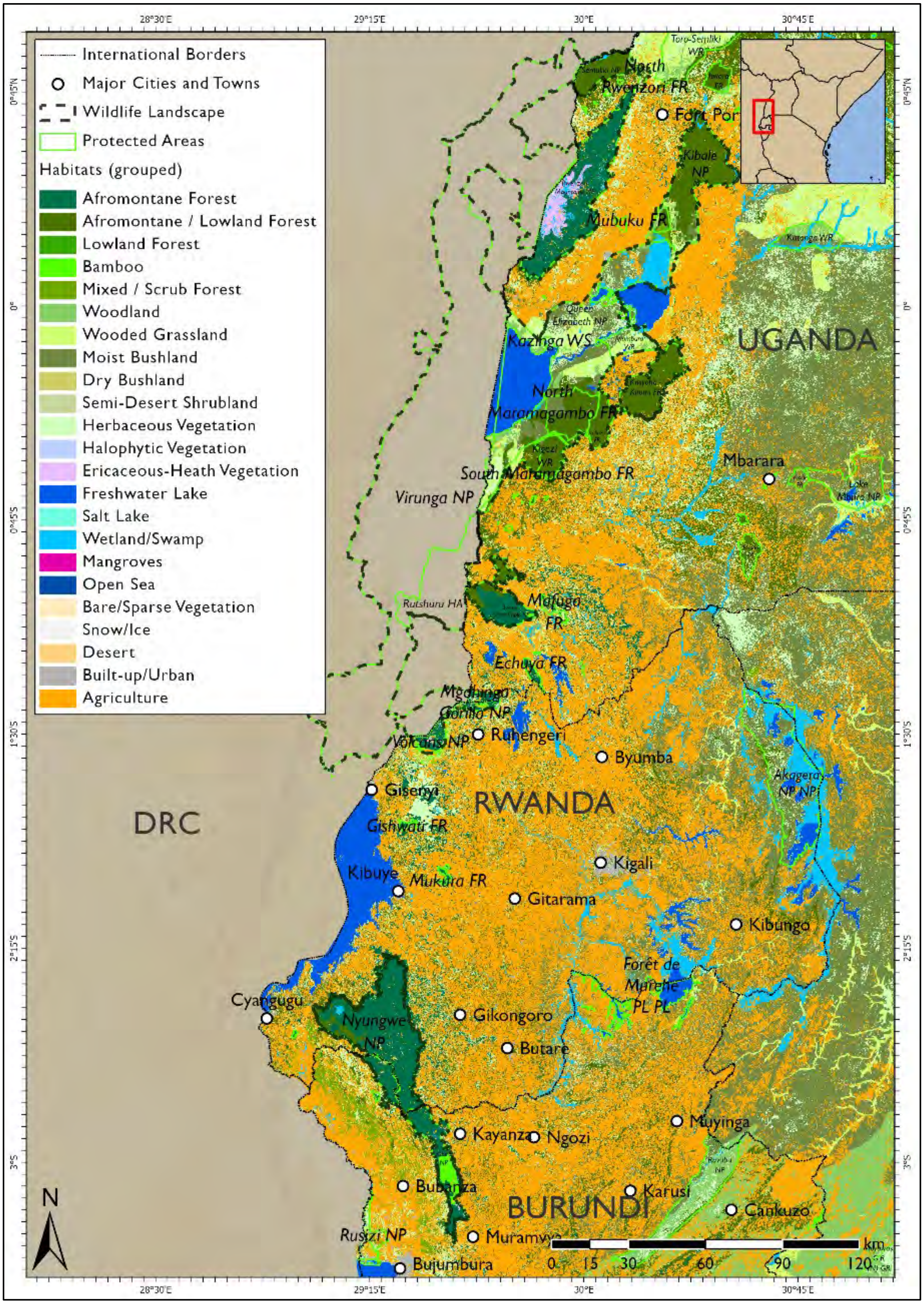


Figure 58. The Albertine Rift Forests wildlife landscape showing land cover

Figure 59. The Albertine Rift Forests wildlife landscape showing its protected areas

PROTECTED AREAS AND NATURAL RESOURCE MANAGEMENT

The dominant protected areas within the Albertine Rift Forests study region are Kibale, Queen Elizabeth, Rwenzori Mountains, and Bwindi Impenetrable National Parks in Uganda, Volcanoes and Nyungwe Forest National Parks in Rwanda, and Kibira National Park in Burundi (Figure 59). These six national parks cover close to 600,000 hectares (Table 45). However, they are becoming increasingly isolated in a matrix of agricultural fields and settlements. The only remaining connection between the national parks in northern Rwanda and southern Uganda is through corridors of forest that connect these parks to neighboring Virunga National Park in the Democratic Republic of Congo. Without this connection, these parks would become isolated forest patches. Nyungwe and Kibira National Parks in Rwanda and Burundi are contiguous and form part of the proposed Nyungwe-Kibira Transboundary Conservation Area (TFCA, IUCN ESARO, 2020), but there are no corridors that link these parks to other forested landscapes. Queen Elizabeth adjoins Kibale National Park and a network of forest reserves to form a 180 km long corridor for movement of wildlife between these two parks. The parks, their management, and their defining characteristics are described in Table 45.

The government agency responsible for the management of forests and protected areas in Burundi is the Burundian Office for the Protection of the Environment (OBPE), which works under the supervision of the Ministry of the Environment, Agriculture and Livestock. In Rwanda, a number of government agencies are responsible for conservation. The two main implementing agencies for protected area and forest management are the Rwanda Development Board (RDB), which is overseen by the Ministry of Trade and Industry, and the Rwanda Forestry Authority (RFA), which is overseen by the Ministry of Environment. Management of protected areas is the mandate of the RDA while the RFA is responsible for conservation and forest management. According to key informants, the overlapping mandates of these agencies and ministries can lead to a lack of clarity around the roles and responsibilities of these different government bodies in implementing conservation programs.

In Uganda, the Uganda Wildlife Authority (UWA) is responsible for management of the national parks and wildlife reserves situated within the landscape, while the National Forest Authority (NFA) manages the forest reserves. The UWA is situated under the Ministry of Tourism, Wildlife, and Antiquities, while the NFA operates under the Ministry of Water and Environment. Smaller district forest reserves are overseen by local district governments. Uganda does practice collaborative forest management with communities in some areas. This involves the signing of agreements between the NFA and communities, which result in communities taking charge of a particular area and managing it in collaboration with the NFA. Finally, water resources in Uganda are managed by the Ministry of Water and Environment, which has a dedicated Wetlands Management Department, in conjunction with the National Environmental Management Authority.

Government agencies are assisted by a number of private sector and NGO conservation partners in the region. In addition to collaborating with governments on protected area management, these organizations support community projects to increase the flow of benefits from nature-based tourism to local communities and to promote alternative livelihood strategies that reduce reliance on natural resources. African Parks currently manages Akagera and Nyungwe National Parks in collaboration with the Rwanda Development Board, which has helped to increase funding and effectiveness of conservation management. Rwanda also recently signed an agreement with Wilderness Safaris to manage tourism activities in Gishwati-Mukura National Park. Such management partnerships help to reduce pressure on over-stretched government agencies. Ugandan key informants also noted that the government plays an active role in providing technical support and even funding to local conservation organizations such as Uganda Women Birders and the Chimpanzee Sanctuary and Wildlife Conservation Trust. This extends to the local level, where conservation NGOs sign MoUs with local governments and receive technical support from them where required. Other active NGOs in the wildlife landscape include the Dian Fossey Gorilla Fund International, International Gorilla Conservation Programme, African Wildlife Foundation (AWF), and Wildlife Conservation Society.

Table 45. The national parks within the Albertine Rift Forests study region and their defining features

NATIONAL PARK	SIZE (HA)	MANAGEMENT	DEFINING FEATURES
Kibira	47,746	Established 2000, managed by the Burundian Office for the Protection of the Environment (OBPE)	Primary montane rainforest, more than 200 bird species, 10 primate species including chimpanzees, 98 mammal species, more than 600 species of plant.
Nyungwe	101,861	Established 2004, managed by Rwanda Development Board (RDB)	Largest tropical Afromontane rainforest in East Africa, more than 13 primate species (25 percent of Africa's total), more than 300 bird species, 85 mammal species, 1,000 plant species. Number of endemic species is greater than in any other forest in the Albertine Rift Mountains.
Volcanoes	14,592	Established 1925, managed by RDB	Afro-montane rainforest, encompassing five of the eight volcanoes in the Virunga Mountains. Home to about 300 endangered mountain gorillas and other primates such as the golden monkey.
Bwindi Impenetrable	32,721	Established 1991, managed by Uganda Wildlife Authority (UWA)	Both montane and lowland forest. Home to 400 mountain gorillas, half of the world's population. Also 120 species of mammals, 348 species of birds, 220 species of butterflies, 27 species of reptiles. One of the most diverse forests in East Africa, with more than 1,000 flowering plant species.
Queen Elizabeth	208,755	Established 1952, managed by UWA	A range of diverse ecosystems, including savanna, shady, humid forests, lakes, and wetlands. Home to big game (elephant, hippo, buffalo, lion, leopard), 10 primate species including chimpanzees and more than 600 species of birds.
Kibale	78,868	Established 1993, managed by UWA	Range of habitats from savanna and woodland to tropical forest. Famous for chimpanzee tracking. Home to more than 300 bird species, 13 primate species, and 75 mammal species.
Rwenzori Mountains	96,977	Established 1991, managed by UWA	Known as the *Mountains of the Moon*, the national park is home to 70 mammal species and 217 bird species including 19 Albertine Rift endemics, as well as some of the world's rarest plant species.

Source: Uganda Wildlife Authority, UNESCO, Rwanda Development Board

There are also some notable nature-based business and investment initiatives in the study region. Rwanda created a National Fund for the Environment (FONERWA) in 2012, which is the largest fund of its nature in Africa. Branded as the Rwanda Green Fund, FONERWA seeks to be the engine for green growth in Rwanda, providing funds to support the country's environmental, climate change-related and developmental needs and goals. Several successful tourism-related enterprises have also taken off in the region, particularly in the Rwandan and Ugandan portions of the landscape. These may employ local people as guides or pay communities for access to certain attractions, including both natural attractions and cultural tours. Recently, efforts have also been made to implement payment for ecosystem services (PES) initiatives in the region. For example, a four-year PES project was tested in Uganda, which involved paying private forest owners in Hoima and Kibaale Districts to conserve and sustainably manage their forest areas. Although key informants noted the research and monitoring showed the return on investment was favorable, the scheme ultimately failed to gain sufficient support from government and the private sector to carry the project forward. However, a key informant noted that a smaller-scale PES project focusing on the Rutaha River, again located in Uganda, has achieved very positive results. These include increased water flows as farmers continue to follow improved practices beyond the 18-month project duration. The key informant felt that opportunities for implementing PES schemes in the Ugandan portion of the landscape were significant, but noted that increased political buy-in and understanding of the concept is still needed to fully realize this potential.

Efforts have been made to promote cross-border collaboration in conservation of the landscape. At a general level, there are a number of EAC initiatives that seek to allow member states to cooperatively deal with issues around conservation and natural resource management. Additionally, all states in the landscape are part of the Nile Basin Initiative, which is high-level inter-governmental panel promoting integrated management and development of water resources in the Nile Basin. The Greater Virunga Transboundary Agreement has a more specific conservation focus. It seeks to formalize cross-boundary collaboration between the DRC, Rwanda, and Uganda in the conservation of the Greater Virunga Landscape. The process has culminated in the formation of the Greater Virunga Landscape Collaboration as an international institution with a fully functioning executive secretariat based in Kigali. These initiatives provide potential entry points for promoting increased transboundary collaboration in conservation matters. Key informants also noted that there have been discussions around the formation of a transboundary body to manage the Nyungwe-Kibira Transboundary landscape. However, those active in the Nyungwe-Kibira region noted that joint patrols between Burundi and Rwanda have stopped due to political tensions between the two countries, suggesting transboundary collaboration remains more challenging in this portion of the landscape.

Future plans to invest in nature in the landscape, according to key informants in Burundi and Rwanda, include efforts to market the Nyungwe-Kibira landscape as a nature-based tourism hotspot and to expand investment in tourist resorts and hotels in general. In Uganda, an NGO key informant noted that they have been providing training, technical advice, and registration assistance to communities living in areas attractive to tourism, in the hope that this will increase their capacity to provide tourism services. A few community-based tourism enterprises have reportedly since taken off, with potential for further growth. Key informants from Uganda suggested more pressure could be put on major private sector companies active in the region, including multinational petroleum companies, to invest more resources in habitat restoration and environmental conservation in order to offset their negative environmental impacts.

PEOPLE AND LIVELIHOODS

The exceptional diversity of species and habitats in the region continues to face severe threats from human activities. Population densities are very high across most of the study region, particularly in southwestern Uganda, Rwanda, and Burundi, putting heavy pressure on the region's extraordinary biodiversity (Brooks *et al.*, 2004). Indeed, Rwanda (512 people/km^2) and Burundi (449 people/km^2) are the most densely populated countries in mainland Africa (UN 2019). There are just under 10 million people living within this study region, an estimated 2 million rural households with an average household size of five people (Table 46). Smallholder farming and livestock rearing are the dominant livelihood activities in the region, which remains predominantly rural (97 percent) despite increasing urbanization across all countries in the study area (Salerno *et al.*, 2018). High population densities have resulted in significant deforestation, as land is cleared for livestock and cultivation, and trees are harvested to meet growing demands for fuel, construction materials, and other needs. Land shortages have also resulted in increasing cultivation of marginal and vulnerable areas such as steep slopes and riparian zones, resulting in substantial soil erosion. The mountainous areas receive substantial rain and have fertile soils, making rain-fed agriculture the dominant livelihood activity. Tea and coffee are important commercial crops in the region, with large tea plantations a common sight surrounding the forested national parks. Households grow a variety of fruit and vegetables for household consumption as well as for sale at market. Small-stock farming is important as is dairy cattle in some areas.

Community representatives interviewed as part of the study indicated high awareness of the benefits provided by nature. These include a range of important resources harvested from forests such as wood for fuel and construction, medicinal plants, herbs, grass, wild foods, bamboo, and other crafting materials. Many also indicated that the forests help to maintain high, consistent rainfall in the area, benefitting crop yields. Some respondents also noted benefits associated with revenues generated from nature-based tourism, including employment opportunities and the development of infrastructure such as schools and health facilities as part of benefit-sharing mechanisms. These benefits are described further and valued in the following sections.

Table 46. Population statistics for the Albertine Rift Forests study region

COUNTRY	TOTAL POPULATION	NUMBER OF RURAL HOUSEHOLDS	AVERAGE HOUSEHOLD SIZE	% RURAL
Uganda	4,134,479	882,613	4.5	97
Rwanda	3,561,893	766,745	4.4	96
Burundi	1,550,621	308,413	5.0	98
Total for study region	9,246,992	1,957,772	4.6	97

ECOSYSTEM SERVICES

NATURE-BASED TOURISM

The Albertine Rift Forests are among the most biodiverse forests in the world and one of the most important regions for conservation in Africa (Plumptre *et al.*, 2003). The forests and surrounding habitats are known for their richness in species and high levels of endemism (Plumptre *et al.* 2003). These natural attractions form the backbone of the tourism industry in Rwanda and Uganda, with both countries having tourism policies and strategic visions that promote high-end, responsibly managed ecotourism rather than mass tourism (Moyini, 2000; Rwanda Ministry of Trade & Industry, 2009; Nielsen & Spenceley, 2010). Encompassing all tourist activities related to nature, nature-based tourism is an important component of the overall tourism sector in these countries. Activities include visits to national parks, nature reserves, and game reserves, and outdoor activities such as hiking and trekking, biking, and birdwatching. The most popular nature-based tourism destinations in this study region are Queen Elizabeth and Bwindi Impenetrable Forest National Parks in Uganda and Volcanoes National Park in Rwanda. While tourism in Burundi is relatively underdeveloped, the sector has shown growth in recent years with international tourist arrivals increasing in 2017 and 2018. The Kibira National Park in northern Burundi is an important attraction, especially for birdwatchers.

Gorilla trekking is the most popular tourism activity for international leisure tourists in Rwanda (Spenceley *et al.*, 2010) and a major tourist attraction in Uganda. Rwanda and Uganda are currently the only two countries in the world where mountain gorillas *Gorilla beringei beringei* can be safely visited, in the forests of Volcanoes and Bwindi Impenetrable Forest National Parks. Mountain gorillas are confined to an area that spans approximately 447 km^2 (Spenceley *et al.*, 2010) in just two populations—one in the Virunga Massif that covers the borders of Uganda, Rwanda, and the DRC, and the other in the Bwindi Impenetrable National Park in Uganda. A census carried out by the Greater Virunga Transboundary Collaboration (a partnership of governments, non-profits, and conservationists) in 2018 shows that the population of mountain gorillas in Bwindi Impenetrable Forest National Park is estimated to be 459, an increase from the 400 counted in 2011 (Hickey *et al.*, 2018). During 2015 and 2016, the population of the Virunga Massif was estimated to be 604 gorillas, up from the 480 gorillas counted in 2010. This brings the total number to just over 1,000 gorillas within the wildlife landscapes of the Albertine Rift Forests. Within these parks, gorilla family groups have been habituated to tolerate visits by tourist groups at close range. A chance to track and spend up to one hour with a family group of gorillas will cost a tourist US$1,500 in Rwanda and US$700 in Uganda for a single permit, which does not include accommodation, travel, or food costs. Hatfield & Malleret-King (2007) estimated the economic value of viewing gorillas in Rwanda and Uganda using the travel cost method based on visitor numbers in 2001/2002. Their study estimated that the expenditure on travel specifically to see the gorillas was US$2.12 million for Volcanoes National Park, US$4.04 million for Bwindi Impenetrable Forest National Park, and US$1.59 million for Mgahinga Gorilla National Park.

Gorillas are not the only draw for tourists in the region. Other charismatic primate species such as chimpanzee are found in several forest areas, while rarer species like the golden monkey are an added attraction in Volcanoes and Mgahinga National Parks. Meanwhile, the forests of Nyungwe and Kibira hold 13 primate species, including unique large conglomerations of Angolan Colobus. Nyungwe National Park is also an attractive birdwatching destination, boasting an exceptional number of endemic species. Finally, the savannas of Queen Elizabeth National Park are home to significant populations of savanna elephant, lion, leopard, and buffalo. Gorilla trekking is often an important catalyst for these other tourism activities in the region, as tourists who primarily come to see the mountain gorillas also visit these other wildlife areas and tourist attractions as part of their itineraries.

Community-based tourism initiatives and revenue sharing processes are in place surrounding these parks, particularly in Uganda and Rwanda. The Rwanda Development Board (RDB) shares 10 percent of nature-based tourism revenues with communities living adjacent to parks. As of 2020, around US$2.3 million had been shared with communities across Rwanda as a whole, since the start of the revenue-sharing arrangement in 2005. Community projects surrounding Volcanoes National Park have included the construction of schools, tree planting initiatives, erosion control and fencing, water tanks, and income-generating activities such as beekeeping and basket weaving (Spenceley *et al.*, 2010). Additionally, Rwanda has a Special Guarantee Fund, which sets aside 5 percent of the revenues generated from nature-based tourism for the purposes of compensating victims of human-wildlife conflict. These revenue-sharing mechanisms appear to have contributed to more positive views of wildlife among communities living adjacent to parks in Rwanda, particularly around Volcanoes National Park where gorilla tourism generates significant revenue. In Uganda, 20 percent of park gate fees are shared with local communities around the parks. This revenue initially went directly to individuals and organized groups but is now channeled through the district and sub-county local governments, with the aim of funding development projects that benefit the community as a whole (*e.g.,* health facilities and schools).

In addition to revenue-sharing, communities can benefit from nature-based tourism through employment opportunities. For example, community guides hired by tourists are paid significant amounts of money in areas such as Kibale and Bwindi Impenetrable National Parks. Community informants from Rwanda similarly noted they benefit from employment as guides or porters for tourists visiting Volcanoes National Park. Communities can also earn revenue through community-owned lodges. For example, a key informant noted that Sabyinyo Community Livelihood Association's Sabyinyo Silverback Lodge generated US$2.7 million in revenue between 2007 and 2020 by providing accommodation to visitors to Volcanoes National Park. Nature-based tourism is also a catalyst for other tourism initiatives in the region, such as cultural tours to see traditional dancers or try local foods. According to key informants, these are popular around Volcanoes National Park in Rwanda and around Bwindi, Mgahinga, and Semuliki National Parks in Uganda. Some of these tours have been initiated and run by the communities themselves. Even in Burundi where visitor numbers are low, some community informants living around Kibira National Park note the beneficial effects of tourism and the need to maintain forest habitats and animal populations to ensure the area remains attractive to tourists. Some local communities around Kibira have organized themselves into cooperatives that acquire and share revenues generated from tourists paying to visit the forest and purchasing handcrafts. They also assist with protecting the forest, such as by creating and maintaining fire breaks along the forest edge. On the other hand, some community leaders from other parts of the Kibira region noted that wildlife had been extirpated and thus tourists no longer visited.

There has been significant progress in developing and managing tourism in this region over the past decade, particularly in Rwanda, where numbers of visitors to the national parks have increased steadily over time (Figure 60). The high annual growth rate in visitor numbers to Nyungwe National Park (16 percent) is also promising, higher than the 10 percent annual growth rate seen for Volcanoes National Park. In Uganda, visitor numbers to the national parks have also shown steady increases, but some parks have experienced drops in visitor numbers over some years (Figure 61). For example, visitor numbers to Queen Elizabeth National Park decreased between 2012 and 2015 but have since recovered, and numbers to Bwindi Impenetrable Forest National Park declined in 2014 and 2015 but have also shown signs of recovery. Rwenzori Mountains National Park has shown the largest growth in visitor numbers, with an annual growth rate of 19 percent between 2002 and 2017. This compared to an annual growth rate of 8 percent for Queen Elizabeth and 10 percent for Bwindi Impenetrable Forest National Parks over the same period.

Holiday tourists, who account for most of the expenditure on visiting tourism attractions, represent about 22 percent of tourists to Uganda but only 7 percent in the case of Rwanda and Burundi (Table 47). The percentage of holiday tourists is low when compared to other East African countries, such as Kenya and Tanzania, where they make up 74 percent and 64 percent of tourists, respectively.

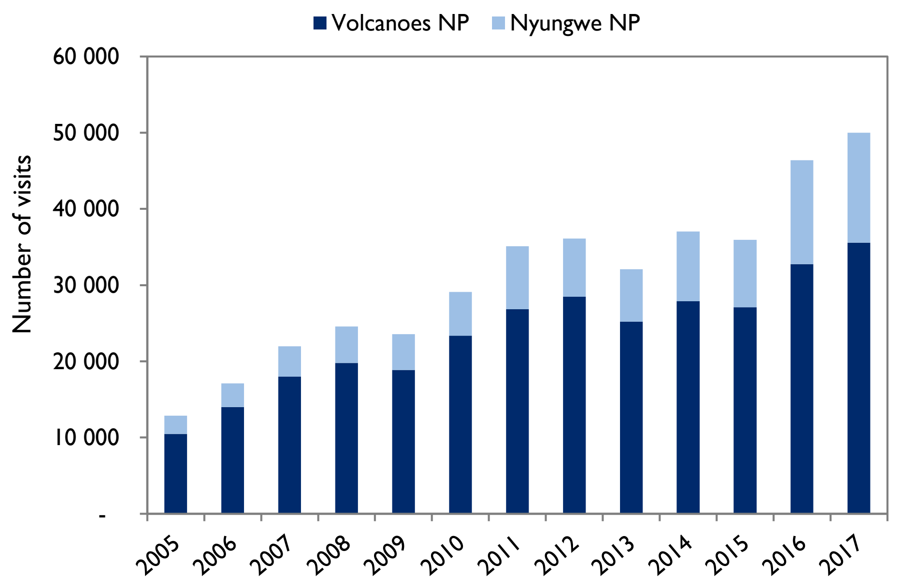


Figure 60. Total number of visits to Volcanoes and Nyungwe National Parks in Rwanda from 2005-2017

Source: NISR, 2019

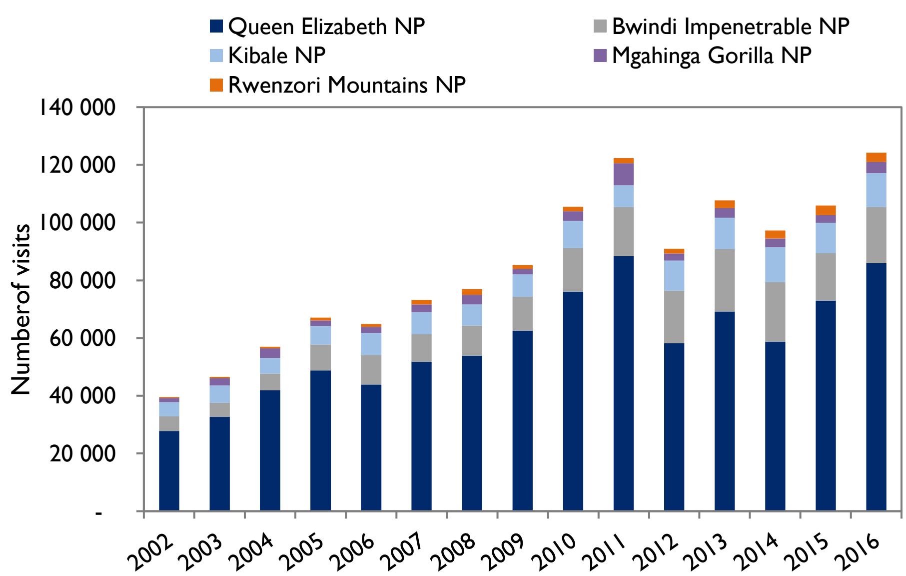


Figure 61. Total number of visits to Queen Elizabeth, Bwindi Impenetrable Forest, Kibale, Mgahinga Gorilla, and Rwenzori Mountains National Parks in Uganda from 2002-2016

Source: Uganda MTWA

Table 47. Typology of tourists to Uganda, Rwanda, and Burundi in 2018

PURPOSE OF VISIT	UGANDA (%)	RWANDA (%)	BURUNDI (%)
Holiday	22	7	7
VFR	39	30	30
Business	32	29	29
Other	7	33	33

Note that this data was not available for Burundi, so it was assumed that these estimates would be the same as neighboring Rwanda.

The total attraction-based tourism value (in terms of direct contribution to GDP) in 2018 for Uganda was estimated to be US$220 million, US$124 million in Rwanda, and US$15 million in Burundi (Table 48). Based on empirical evidence of tourist activity (photo densities) it was estimated that 16.6 percent, 10.7 percent, and 3.5 percent of these national values were attributed to the Albertine Rift Forests landscape in Uganda, Rwanda, and Burundi, respectively (Table 48, Figure 62). The total nature-based tourism value of the Albertine Rift Forests landscape was therefore estimated to be US$50.3 million in 2018: US$36.5 million in Uganda, US$13.3 million in Rwanda, and US$0.5 million in Burundi. Note that this direct value added is about 45 percent of the total expenditure in the country (Hatfield & Malleret-King, 2007) and that the economy-wide impacts are higher.

Table 48. The estimated total attraction-based tourism value for Uganda, Rwanda, and Burundi in 2018 and estimated nature-based tourism value of the Albertine Rift Forests wildlife landscape

COUNTRY	TOURISM DIRECT CONTRIBUTION TO GDP	LEISURE SPENDING AS A PROPORTION OF TOTAL SPENDING (%)	TOTAL ATTRACTION-BASED TOURISM VALUE PER COUNTRY	TOURISM VALUE OF WILDLIFE LANDSCAPE	% OF NATIONAL VALUE
Uganda	$715 m	87	$220 m	$36.48 m	16.6
Rwanda	$416 m	48	$124 m	$13.30 m	10.7
Burundi	$49 m	44	$15 m	$0.54 m	3.5

All values in 2018 US$ millions

In addition, wildlife tourists derive considerable consumer surplus from visiting the study area (Hatfield & Malleret-King, 2007). Consumer surplus of international visitors to see gorillas in three protected areas in Rwanda and Uganda was estimated to be more than $5 million per annum in 2001/2. Based on the relative value of consumer surplus and in-country expenditure in their study, it was estimated that the consumer surplus of international visitors to all the parks in the study was approximately $83.4 million a year in 2018.

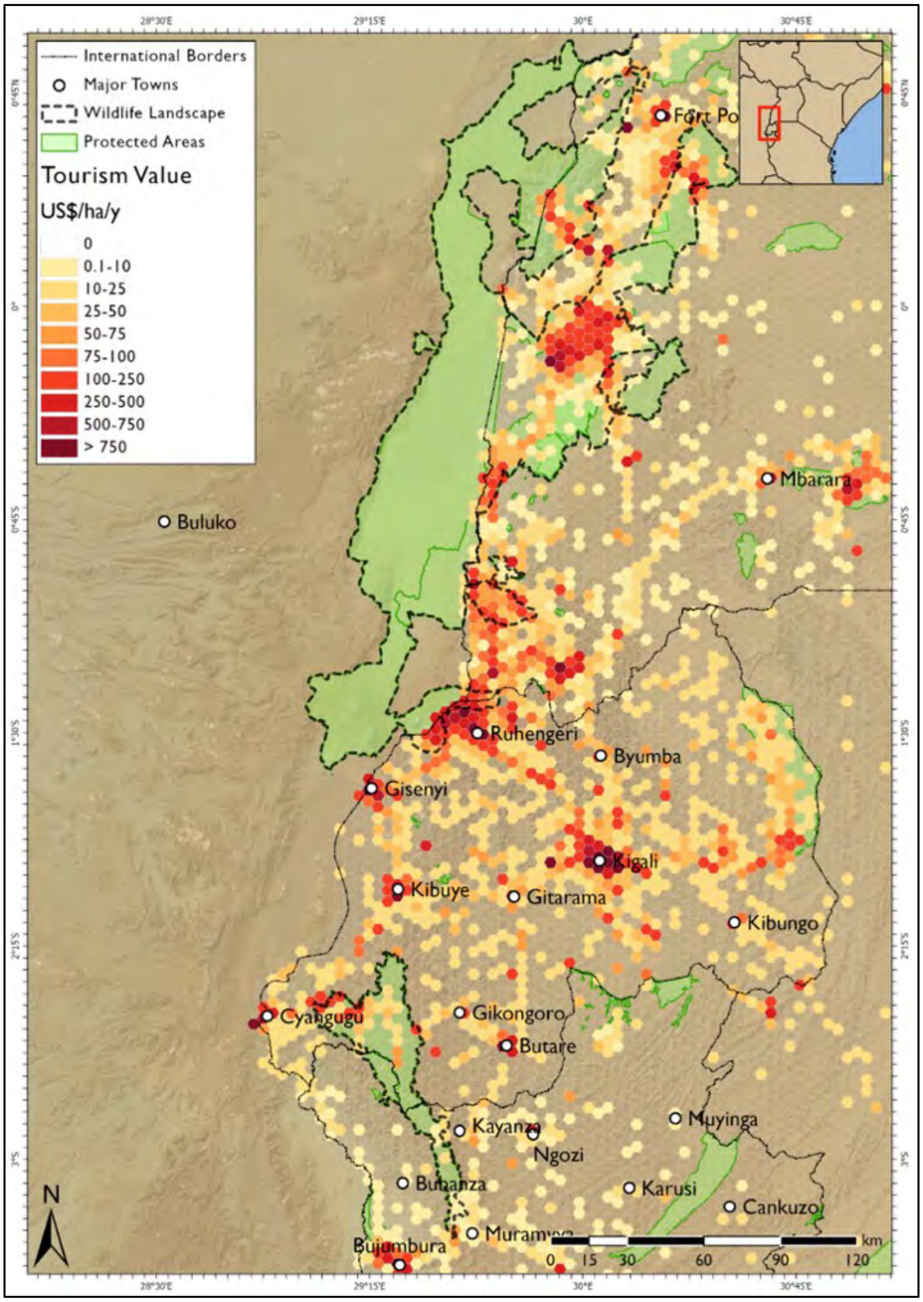


Figure 62. Nature-based tourism value (US$/ha/y) for 2018 across the Albertine Rift Forest wildlife landscape, based on the distribution of geo-referenced photographs uploaded to Flickr

The map of tourism value (Figure 62) clearly shows that Queen Elizabeth Park in Uganda has the highest total value of the all the national parks across the Albertine Rift Forests landscape, followed by Volcanoes National Park and Mgahinga Gorilla National Park (Table 49). However, Volcanoes National Park and Mgahinga Gorilla National Park have the highest per hectare tourism value, with values of up to US$837 per hectare per year. This highlights the conservation importance of these parks for the protection of mountain gorillas and the role they play in generating tourism income for these two countries. Rwenzori Mountains, Kibale, and Nyungwe National Parks have values ranging from US$19-27 per hectare per year, and unsurprisingly Kibira National Park in Burundi has a lower value of US$8 per hectare per year.

Table 49. The tourism value of the national parks of Uganda, Rwanda, and Burundi in the Albertine Rift Forests landscape

NATIONAL PARK	COUNTRY	TOURISM VALUE (US$ MILLIONS/Y)	TOURISM VALUE (US$/HA/Y)
Queen Elizabeth	Uganda	17.8	85
Volcanoes	Rwanda	9.1	564
Mgahinga Gorilla	Uganda	3.2	837
Bwindi Impenetrable	Uganda	2.7	83
Nyungwe	Rwanda	2,.7	26
Kibale	Uganda	2.0	27
Rwenzori Mountains	Uganda	1.9	19
Kibira	Burundi	0.4	8

All values 2018 US$

FLOW REGULATION

Natural ecosystems regulate seasonal surface flows through infiltration of rainfall into groundwater flows, and in so doing reduce the seasonal variation in flows by slowing down water through the landscape and contributing to river base flows during the dry season. This reduces the size of reservoirs that are needed to meet water demands, as well as affecting the availability of water to people who draw water directly from streams and rivers. In this study, the flow regulation service was evaluated as the difference in the contribution to baseflow (*i.e., water* that reaches a stream) between current land cover and a scenario in which all land cover is converted to bare ground.

The Albertine Rift Forest landscape was estimated to have an average baseflow contribution of 4,125 m^3 per hectare per year, the highest of all the wildlife landscapes studied (Figure 63). This amounts to a total baseflow contribution of 3,692 million m^3 per year across the landscape. Recharge, and thus contribution to baseflow, is generally higher in areas under natural vegetation and higher rainfall, although soil characteristics are another moderating factor. The highest local recharge values in the modelled Albertine Rift Forests study region were associated with the steep forested protected areas, especially Nyungwe National Park and the Rwenzori Mountains. However, the difference in baseflow when current land cover was compared to a denuded landscape was estimated to be negligible overall. Increased quickflow runoff with the removal of vegetation in the denuded scenario would potentially reduce infiltration and baseflow. However, this appears to be negated by the high evapotranspiration requirements of forest vegetation under current land use, resulting in little change in baseflow between the current and denuded land cover scenarios.

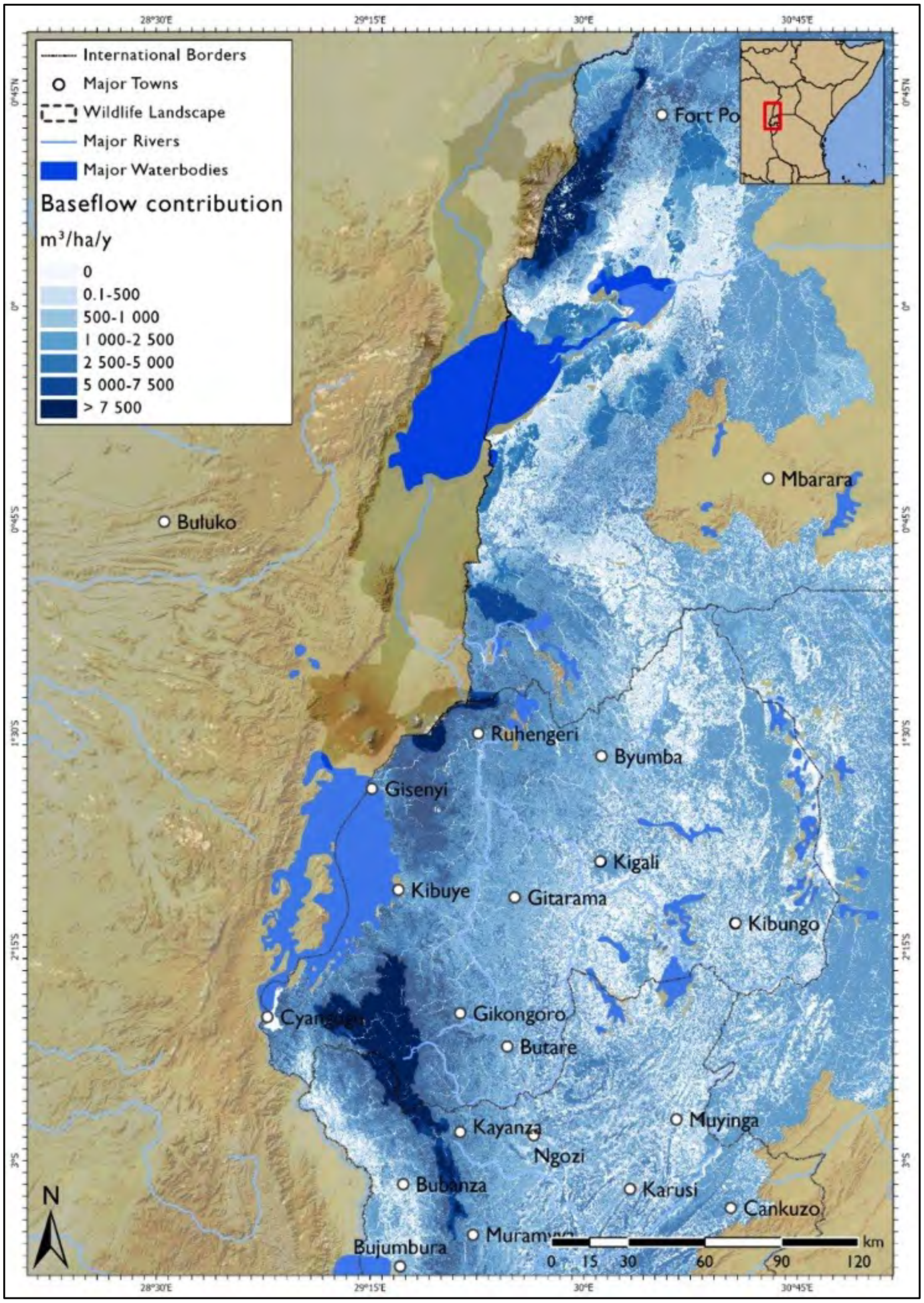


Figure 63. Baseflow contribution (m^3 per ha per year) by ecosystems of the Albertine Rift Forests landscape relative to a barren landscape

WATER QUALITY AMELIORATION

Levels of phosphorus export are generally low across the Albertine Rift Forests wildlife landscape (Figure 64). Outside of the protected area wildlife landscape, higher nutrient export values are noticeable. These are associated with cultivated areas, which cover most of this region outside of the protected areas. High nutrient export values are associated with more intensively cultivated land adjacent to the wildlife landscapes of Rwanda and Uganda, and to a lesser extent in Burundi where farmers apply less fertilizer on average.

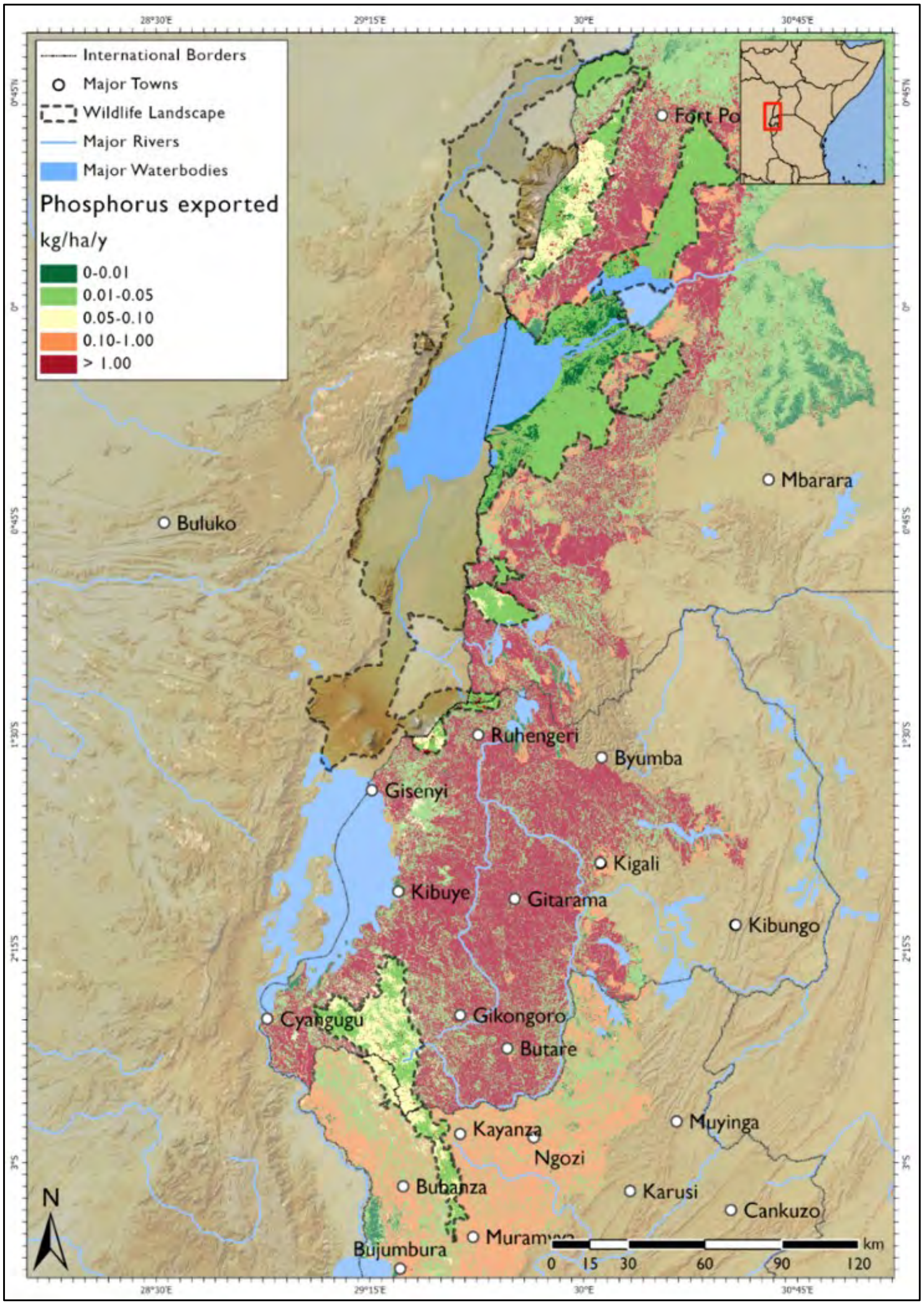


Figure 64. Average phosphorus exported (kg per ha per year) by ecosystems of the Albertine Rift Forest wildlife landscape

For valuation of the nutrient retention service, we focused on the entire wildlife landscape, which drains into the Rift Valley Lakes, as well as into the Akagera catchment and then Lake Victoria. From our InVEST model, we estimated current phosphorus export from the wildlife landscape to be on the order of 62 tons. We estimated both the active and passive nutrient retention service provided by the wildlife landscape. The active service refers to the current retention of nutrients by vegetation. If the natural vegetation in the wildlife landscape stopped retaining phosphorus, we estimated the replacement cost of the retention service to be on the order of US$331,360. This value of the active service is fairly small, as most of the cultivated areas exporting large loads of phosphorus into the lakes are located downstream of the wildlife landscape (Figure 64). Since nutrient export is generally low across the wildlife landscape, active retention of nutrients by vegetation in the wildlife landscape is in turn low. However, export of phosphorus into the Rift Valley Lakes and Akagera catchment would be much higher if the wildlife landscape were converted to agriculture, as demonstrated by the high nutrient export values associated with cultivated areas. The nutrient export avoided by maintaining natural vegetation, at the expense of cultivation, is the passive service provided by the wildlife landscape. We estimated the replacement cost of this passive service to be around US$682,469.

EROSION CONTROL

Natural habitats reduce soil erosion and transport of sediment to downstream habitats. This can occur through both *in situ* retention of soil due to vegetation cover as well as through the trapping of sediments that have been eroded from elsewhere in the landscape. By doing so, natural vegetation can reduce the negative impacts of excess sediment loads in watercourses, such as reduced water quality and loss of reservoir storage capacity. In this study, the sediment retention service was evaluated by the difference in sediment export between current land cover and a scenario in which all land cover is converted to bare ground. This difference provides a measure of the amount of sediment currently being retained by the landscape.

The Albertine Rift Forests wildlife landscape was estimated to have an average sediment retention value of 619 tons per hectare per year, much higher than any of the other wildlife landscapes (Figure 65). Average sediment retention was highest in Burundi, where the forests retain close to 1,000 tons of sediment per hectare per year. These values were lower for Rwanda (782 t/ha/y) and Uganda (552 t/ha/y). The high values across the region can be attributed to high rainfall across most of this region, often in combination with steep slopes. Together, these result in high potential soil erosion in the absence of natural vegetation across much of the landscape. Particularly high sediment retention values were estimated for Rwenzori and Volcanoes National Parks, with natural vegetation retaining more than 700 t/ha of sediment over a large portion of these protected areas. Other mountainous forested protected areas retaining large amounts of sediment per hectare include Kibale, Bwindi Impenetrable, Nyungwe, and Kibira National Parks. In contrast, more moderate sediment retention values were estimated in flatter areas, such as Queen Elizabeth National Park and surrounding protected areas, where sediment retention was often calculated to be less than 40 t/ha. As much of the wildlife landscape is situated within the Albertine Rift Valley, natural vegetation in these protected areas makes a substantial contribution to reducing sediment exports to Rift Valley lakes such as Lakes George, Edward, Kivu, and Tanganyika. A smaller portion of the Albertine Rift Forests wildlife landscape also drains into the Lake Victoria Basin. Of note here is Nyungwe National Park, the western portion of which forms the headwaters of the Akagera River. By reducing sediment export to the Akagera River in a high-risk area for soil erosion, this portion of Nyungwe's forests provide important benefits to crucial downstream ecosystems like the Akagera Wetlands for both people and biodiversity.

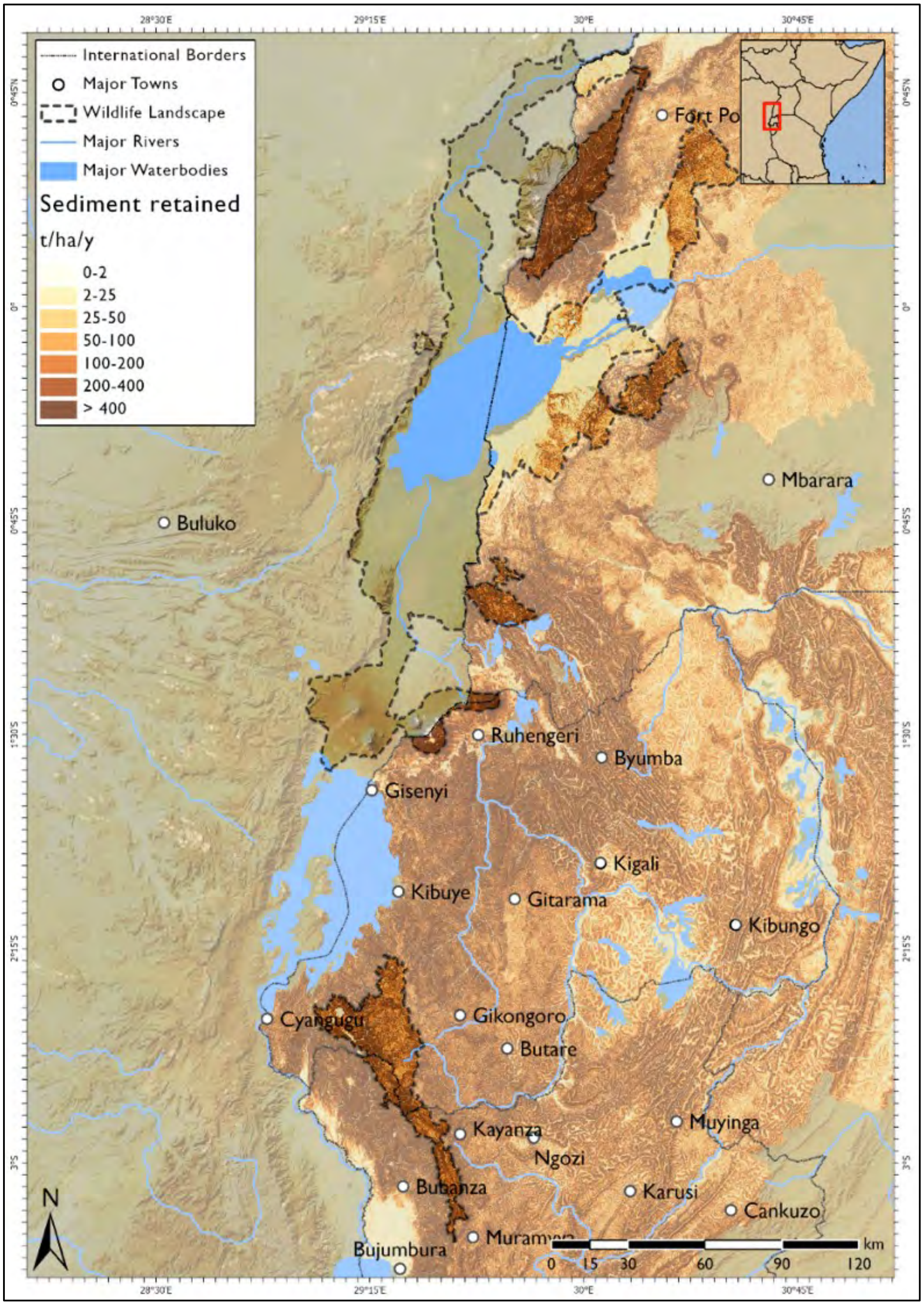


Figure 65. Average sediment retained by ecosystems in the Albertine Rift Forests landscape (tons per ha per year) relative to a barren landscape

It was estimated that current land cover across the Albertine Rifts Forests wildlife landscape retains a total of 498 million tons of sediment per year, relative to a scenario in which all land cover is converted to bare ground (Table 53). Most of this total retention falls within Uganda, where 340 million tons of sediment is retained by the natural vegetation each year. If this sediment was not being retained by the landscape, the replacement cost of this service, in terms of the construction of sediment check-dams, was estimated to be US$612 million per annum.

Table 50. Total sediment retained, mean sediment retained per hectare per year, and the total annual cost of sediment retention (US$ million/y) for the Albertine Rift Forests wildlife landscape

COUNTRY	TOTAL SEDIMENT RETAINED (MT/Y)	MEAN SEDIMENT RETAINED (T/HA/Y)	TOTAL ANNUAL VALUE (US$ MILLION/Y)
Burundi	54	989	65.9
Rwanda	105	782	128.8
Uganda	340	552	417.1
Total	**498**	**619**	**611.8**

CARBON STORAGE

Natural ecosystems make a significant contribution to global climate regulation through the sequestration and storage of carbon. About half of all vegetative biomass comprises carbon. In addition to accumulation in woody biomass, carbon accumulates in soils and peat as a result of the collection of leaf litter and partially decayed biomass. Degradation of vegetated habitats releases carbon and contributes to global climate change with impacts on biodiversity, water supply, droughts and floods, agriculture, energy production, and human health, whereas restoration or protection of these habitats mitigates or avoids these damages, respectively. The conservation and restoration of natural systems thus helps to reduce the rate at which greenhouse gases collect in the atmosphere and the consequent impacts of climate change.

Tropical forests play a critical role in the global carbon cycle (Glenday, 2006; Lewis, 2006). While they only cover about 10 percent of the earth's surface, they are carbon-dense and highly productive, storing approximately half of all carbon in terrestrial vegetation and processing six times as much carbon as emitted through anthropogenic fossil fuel use each year (Lewis *et al.*, 2009). Therefore, even small changes in the extent and intactness of the forest biome can have significant global impacts. Indeed, it has been estimated that forest loss accounts for 12-17 percent of global greenhouse gas emissions (Nakakaawa, Vedeld & Aune, 2011). It has been estimated that the carbon stocks in Uganda decreased by some 850 million tons between 2006 and 2010 as a result of the conversion of forested land to other land uses (Zhang *et al.*, 2017).

Based on global datasets derived from satellite data (see FAO & ITPS, 2018; Spawn & Gibbs, 2020), we estimated that approximately 643 million tons of carbon are stored within the vegetation and soils of the Albertine Rift Forests landscape (Table 51, Figure 66). Of this, 69 percent is stored within Uganda, 22 percent within the forest landscape of Rwanda, and 9 percent in Burundi. The highest densities occur within the protected national parks, in particular Nyungwe Forest, Kibira, Bwindi Impenetrable Forest, and Kibale, where the storage of carbon exceeds 1,000 tons per hectare across most of these protected areas. Densities are lower in Queen Elizabeth, Volcanoes, and Mgahinga Gorilla National Parks and across the Rwenzori Mountains where the forest habitat is less dense and interspersed with other vegetation types. Outside of the wildlife landscape, the carbon storage is significantly lower as remaining forest patches have been converted to agriculture and other land uses. The amount of carbon stored within the forest landscape ranged from as low as 40 tons per hectare to as much as 2,704 tons per hectare in Uganda (Table 51). Mean carbon storage was highest in Rwanda at 1,213 tons per hectare and lowest in Uganda at 826 tons per hectare. In Rwanda, these values align with previous work that has measured the above-ground carbon storage (only) to be between 428 and 659 tons per hectare in Nyungwe Forest alone (Nsabimana, 2009; Cohn, 2011).

Table 51. The total amount of carbon stored within the Albertine Rift Forest landscape and summary statistics (tons carbon per hectare) per country

COUNTRY	TOTAL STOCK OF CARBON (METRIC TONS)	MEAN T/HA	MIN T/HA	MAX T/HA
Uganda	444,936,721	826.07	40.71	2,704.45
Rwanda	143,038,858	1,213.71	252.79	1,968.61
Burundi	55,464,765	1,173.29	135.08	1,624.78

It has been estimated that a ton of carbon released into the atmosphere will cause global damages in the order of US$417 (net present value over 80 years), of which Uganda's share is US$0.84 per ton; Rwanda's share is US$0.16 per ton, and Burundi's share is just US$0.04 per ton. The total global damage costs avoided by retaining the total stock of biomass carbon is substantial, at just over US$56 billion per year (Table 52). The avoided damage cost to Uganda is estimated to be just under US$60 million per year, while the avoided damage costs to Rwanda and Burundi are lower at US$4.6 million and US$0.1 million per year, respectively.

Table 52. The avoided damage costs to local countries and the rest of the world by retaining the total stock of biomass carbon in the Albertine Rift Forests landscape (US$ million/y)

	BURUNDI	RWANDA	UGANDA	REST OF THE WORLD
Carbon storage value (damage costs avoided, US$ million/y)	0.1	4.6	57.8	42,216

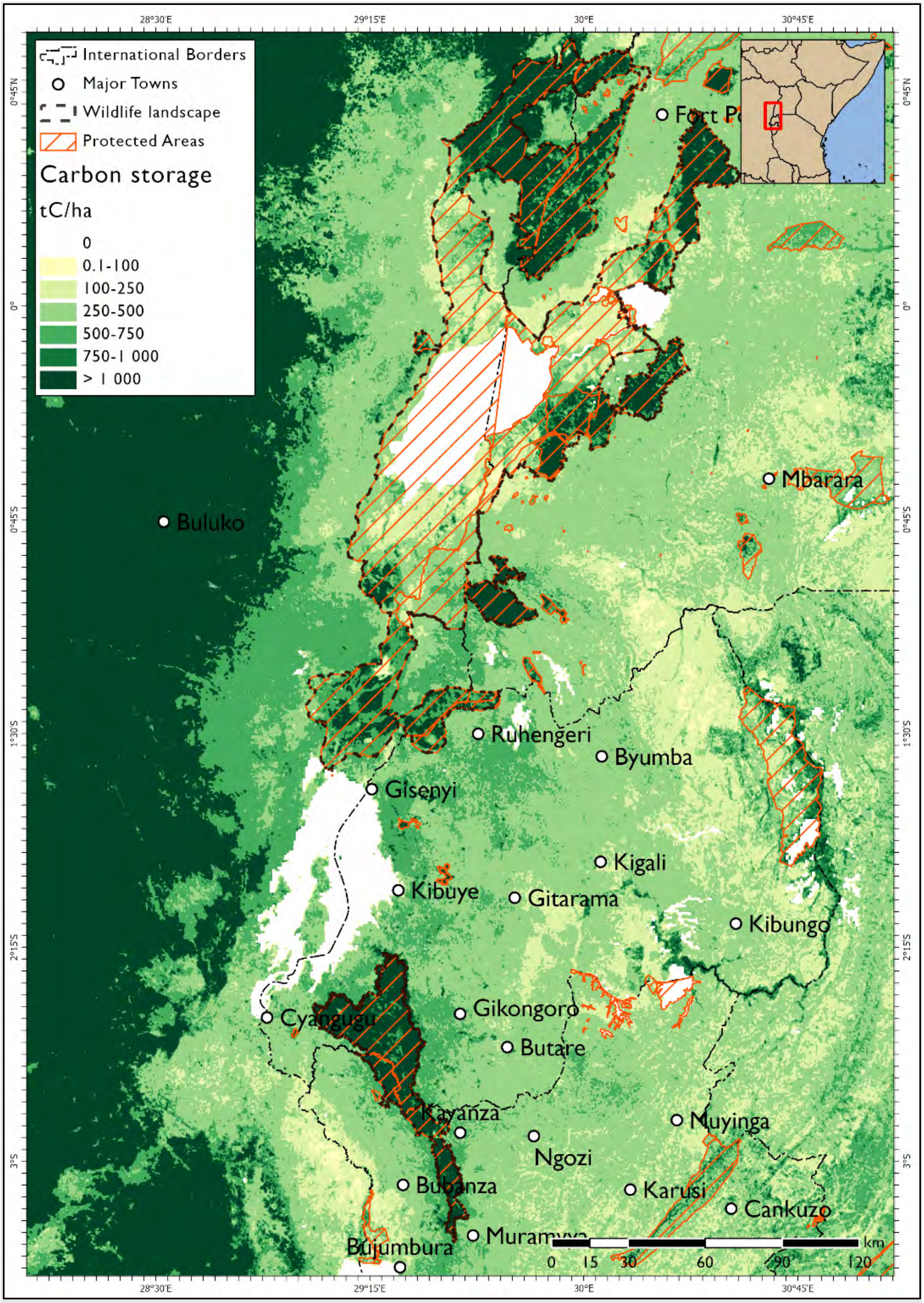


Figure 66. Total carbon storage (tons/ha) across the Albertine Rift Forests wildlife landscape

POLLINATION OF CROPS

Pollination services are widely recognized as critical for human wellbeing and survival given their vital role in ensuring food security. However, the value of wild pollinators remains unclear. This is of concern for sub-Saharan Africa, a region highly dependent on subsistence agriculture as a main source of livelihood (Tibesigwa *et al.*, 2019). The presence of wild pollinators is directly linked to natural vegetation (Kremen *et al.*, 2004), which plays an essential role in certain life cycle stages of pollinator species, such as through the provision of nesting sites or forage at certain times of year. Insects are responsible for 80-85 percent of all pollinated commercial crops, which represents about one-third of global food production (Allen-Wardell *et al.*, 1998; Klein *et al.*, 2007).

Outside of protected areas, smallholder agriculture is the dominant livelihood activity across this study region. The dominant food crops grown across the region are maize, cassava, beans, and bananas, which are typically grown for household consumption, and coffee, pineapples, bananas, Irish potatoes, cabbages, and other vegetables, which are largely sold at market for cash. In such systems, farmers are completely reliant on wild species for the successful pollination of their crops, supported by nearby natural ecosystems. While not all of these crops require insect pollination (*e.g.*, rice and maize), the majority of them do (*e.g.*, pawpaw, passionfruit, sunflower, coffee, beans, groundnuts), either experiencing reduced yields (of up to 90 percent) or a reduction in seed/breeding yield without wild pollination (*e.g.*, cassava and cocoyams).

The value of wild pollination services (contribution to production from crops in smallholder cultivated land) is summarized in Table 53. Based on the percentage share of natural vegetation within a 1,000 m buffer distance of all cultivated land surrounding the wildlife landscape, we estimate the value of wild pollination services to nature-dependent smallholder cultivated land in this study region to be US$36.2 million per year. Just under 80 percent of this value falls within Uganda, 13 percent in Burundi, and just 9 percent of the value in Rwanda. The mean per hectare value was estimated to be US$256 across the landscape, highest in Burundi (US$284 per hectare) and lowest in Rwanda (US$204 per hectare). The very limited natural vegetation outside of protected areas means that the pollination service in this study region is most valuable to smallholder farmers that cultivate crops immediately adjacent (~1 km) to the protected areas. Uganda has the highest density and largest extent of protected areas in this region and as a result the pollination service is highest here.

Table 53. The value of the wild pollination services within the Albertine Rift Forests wildlife landscape (2018 US$)

	TOTAL HA OF NATURAL VEG IN BUFFERS	TOTAL POLLINATION SERVICE VALUE (US$)	MAX VALUE (US$/HA)	MEAN VALUE (US$/HA)
Uganda	108,314	28,145,883	917	260
Rwanda	16,784	3,422,170	969	204
Burundi	16,481	4,674,923	874	284
Total	**141,579**	**36,242,976**	**969**	**256**

Note that the minimum value in all cases was zero.

HARVESTED RESOURCES

In addition to their conservation value, natural resources play an important role in supporting the livelihoods of people. Wild plant and animal resources are harvested for food, medicine, energy, and raw materials, particularly where there are limited economic opportunities. The capacity of the landscape to supply different types of wild resources is related to vegetation type and condition, availability of water, and other factors. However, a number of other factors determine their use and value, and these vary in space and time. The accessibility of wild resources is determined by regulations such as land tenure and harvesting rights, social norms and informal agreements, geographic features such as topography and rivers, and human-made features such as roads. The demand for wild resources is influenced by the socio-economic circumstances of households and the prices of alternatives.

Wildlife habitats usually require full, no-take protection, not only because of the risk associated with overharvesting that changes the nature and functioning of wildlife habitats, but also because of the disturbance that it can cause, especially affecting the shier and more vulnerable wildlife species. The people that live around these wildlife habitats are largely dependent on wild resources, particularly during times of economic stress such as crop disease, drought or floods (which are likely to only worsen with climate change), and international pandemics. During these times, people fall back on nature to fill livelihood needs. However, this is a potentially vicious cycle of unsustainability as more people rely on nature for food and raw materials and stocks become depleted. The stocks of resources protected within parks and reserves help to maintain the stocks utilized outside of these wildlife habitats. The more resources are harvested unsustainably, the fewer there will be available in the future and the less people can rely on nature to fill this need. As resource stocks outside of wildlife habitats become degraded, there will be a higher demand for the resources on the edge of these landscapes as well as on the inside.

Efforts have been made to increase stocks of key natural resources in areas surrounding protected areas. In Uganda for example, the UWA and NFA have supported tree planting and beekeeping projects among communities living around protected areas. At times, these initiatives have been criticized for being top-down interventions that do not foster a sense of community ownership. Furthermore, these initiatives are sometimes conducted as a one-time activity, resulting in a lack of long-term success. Nevertheless, a local leader key informant noted that efforts to promote tree planting have reduced the number of people extracting wood from forests. Multiple key informants from Burundi indicated similar tree planting and sensitization initiatives are taking place around Kibira National Park through the government, NGOs, and community groups. These efforts have achieved some success at reducing harvesting pressures on the forest.

Entrance to and harvesting of resources in most protected areas in the landscape is prohibited by law. Nevertheless, significant harvesting likely still takes place, especially where law enforcement effectiveness is low. Notably, Uganda does have some provisions for limited access to national parks for harvesting purposes through so-called multiple-use programs. Under these arrangements, the UWA allows communities limited access to parks to collect resources such as bamboo, thatching grass, medicinal plants, and honey (Tolbert *et al.*, 2019). Harvesters are only granted permission following registration and authorization, and access is limited to certain hours or days of the week.

THE DEMAND FOR NATURAL RESOURCES

The people that live in the areas bordering the highland forests of Uganda, Rwanda, and Burundi belong to a number of ethnic groups but have in common that their livelihoods are centered largely on rain-fed, hand-tilled agriculture (Figure 67, FEWS NET 2010a, 2012a). In some areas, activities also include small-scale livestock rearing with dairy cattle, goats, and sheep important for supplementing food and cash needs and used as an economic safety net. In Burundi, the production of small stock is favored. These mountainous areas receive between 1,200 and 2,000 mm of rainfall per year, and coupled with relatively fertile soils, allow for the cultivation of a variety of food crops; most households are able to produce surplus food in normal years. Maize, cassava, beans, and bananas are typically grown for household consumption and coffee, pineapples, bananas, Irish potatoes, cabbages, and other vegetables are largely sold at market for cash. In some areas, such as around the Rwenzori Mountains National Park in Uganda, Nyungwe National Park in Rwanda, and the northern highlands of Burundi surrounding Kibira National Park, tea production is an important cash crop. Those living closer to the lakes and larger rivers in the study region also engage in small-scale fishing, but this appears to be limited (NISR, 2010). In many parts of this study region, the mountainous terrain, poor road networks, and remoteness limit market access, particularly in the rainy season when roads are impassable and landslides are common. Poorer households are known to engage in firewood collection and charcoal production for sale and also depend on the collection of wild foods, especially when cash income is low as a result of shocks such as prolonged dry spells or crop losses due to crop disease (FEWS NET, 2010b, 2012).

Figure 67. Subsistence agriculture is the most important livelihood activity in the region. Cultivated fields adjacent to Bwindi Impenetrable National Park in Uganda (left) and on the boundary of Rwanda's Volcanoes National Park (right)
Credit: Jason Houston for USAID (left), John Cooke (right)

A wide variety of wild resources are harvested for nutrition and health, energy, and raw materials from the forested habitats that remain in this region. Each of these are discussed in turn below. Woody resources are particularly important for households in these areas (Table 54). Fuelwood is by far the dominant energy source across the study region, with over 95 percent of households using firewood or charcoal as a main fuel source (Kayanja & Byarugaba, 2001; Wong, Roy & Duraiappah, 2005; Hatfield & Malleret-King, 2007; NISR, 2012; Mwageni, Shemdoe & Kiunsi, 2015; Nyamuyenzi, 2015; Bitariho, Sheil & Eilu, 2016; Ntiranyibagira et al., 2017). This results from limited electrical infrastructure, as well as the higher price of alternative energy sources like paraffin, kerosene, and electricity (where available). A substantial portion household energy demand may be met from plantations and cultivated woodlots in parts of the study area. In Rwanda, for example, where little natural forest remains outside of protected areas, planted forests provide 80 percent of the national wood supply for energy and construction

(Drigo et al., 2013). Firewood is generally the primary fuel in rural areas of the study region, with only about 5-15 percent of rural households using charcoal as their main energy source (NISR, 2012; Mwageni et al., 2015; UBOS, 2018). Estimates for annual per capita fuelwood demand across the region vary from 409 kg/year to 675 kg/year (Kayanja & Byarugaba, 2001; Walter, 2001; Drigo et al., 2013; Gianvenuti & Vyamana, 2018). While it is known that bamboo is also sometimes used for firewood, there is limited information on this.

There is limited subsistence demand for charcoal in the region, but substantial commercial harvesting may take place in certain areas to meet the high demand from urban areas. This may be undertaken by local rural people or by commercial harvesters travelling from other regions. Despite the lower numbers of people involved, charcoal harvesting can have a much more detrimental impact on wooded habitats than subsistence firewood harvesting. For example, in a study conducted near Uganda's Kibale National Park, Naughton-Treves, Kammen & Chapman (2007) found the average charcoal producer harvested 59 m^3/ha pf wood per year, compared to an average annual rural household firewood consumption of around 5 m^3/ha. Furthermore, charcoal harvesters often target slow-growing, indigenous hardwood trees, which adds to the unsustainability of the practice. Hence, charcoal harvesting to meet growing urban demand presents a significant threat to woody habitats in this study region, particularly in Uganda where more wooded habitats remain outside of protected areas than in Rwanda and Burundi.

Table 54. Proportion of rural households harvesting woody resources for wood fuel and raw materials within each country in the Albertine Rift Forests study region and the estimated demand per average household per year

COUNTRY	FUELWOOD		CHARCOAL		POLES & WITHIES		TIMBER	
	% RURAL HH	M^3/HH/Y	% RURAL HH	M^3/HH/Y	% RURAL HH	M^3/HH/Y	% RURAL HH	M^3/HH/Y
Uganda	78	3.8	18	0.2	60	0.1	7	0.1
Rwanda	94	4.7	10	0.1	33	0.1	4	0.1
Burundi	80	3.9	10	0.1	33	0.1	4	0.1

Households also make use of a range of raw materials for house construction, furniture, and mats, as well as ornamental items. Timber and poles are used to varying degrees across the study area (Table 54, Figure 68). Across Rwanda and Burundi, brick houses generally dominate. However, there is still an indirect demand for wood associated with these houses, as brick production requires firewood to heat the kilns. In contrast, half or more of rural houses are still constructed using wooden poles across much of the Ugandan portion of the study region (Rwamahe, 2008; Hartter, 2010; NISR, 2012; Harrison *et al.*, 2015; UBOS, 2018). Poles are also widely harvested for use in other structures, such as fencing and kraals for livestock. Few estimates of the subsistence or small-scale production and harvest of timber and poles could be found for the study area. Based on fieldwork in the Bwamba region of Uganda, just north of Rwenzori Mountains National Park, Howard (1991) estimated annual wood demand for poles to be 0.27 m^3 per household. In addition to subsistence harvesting of timber, some communities living adjacent to forest reserves in Uganda benefit from collaborative forest management arrangements signed through agreements with the National Forest Authority (NFA). As part of these arrangements, the NFA has created a benefit-sharing scheme where 5 percent of revenues generated from timber harvesting in forests is transferred to local communities.

Figure 68. A house on the outskirts of Volcanoes National Park in Rwanda constructed from mud and wooden poles and thatch grass (left). A house in southwestern Uganda constructed from mud bricks and wooden poles with a corrugated iron roof (right)

Credit: John and Melanie Kotsopoulos (left), Nick Ribaudo (right)

The forest habitats are also a source of bamboo, which is utilized by many households in regions where it occurs (

Table 56). Localized populations of African mountain bamboo *Yushania alpina* occur in high-altitude forest areas of southwest Uganda, Rwanda, and Burundi (Nzigidahera, 2006; van der Hoek et al., 2019). The species is prized for a range of uses including handicrafts, furniture, rope, poles, and firewood (Bitariho & Mosango, 2005; Zhao et al., 2018). With subsistence agriculture the dominant livelihood activity in these areas, there is high demand for bamboo baskets during harvesting and post-harvest processing of farm produce. Sale of bamboo products is the primary livelihood option for some individuals, with an average annual income of about US$30 reported for bamboo harvesters using Echuya Forest Reserve in Uganda (Kalanzi et al., 2017). Stocks of bamboo in protected areas like Bwindi and Mgahinga National Parks provide a source of rhizomes, which can be legally collected for on-farm planting by authorized community members (Bitariho & Mosango, 2005).

Similar to bamboo, various liana or vine species are used to make a variety of handicraft products, including important household items like baskets, winnowing trays, and granaries (Muhwezi, Cunningham & Bukenya-Ziraba, 2009; Bitariho & Emmanuel, 2019). They are also woven into stretchers, essential for transporting sick people across the rugged mountainous terrain of the region where few roads exist (Cunningham, 1996). Virtually all households around Bwindi National Park have at least some household items woven out of lianas or bamboo, indicating the importance of these resources to local people (Cunningham, 1996).

Reeds and sedges (*e.g.*, papyrus) are used for the construction of household craft items as well as furniture and sometimes thatching. However, they are not abundant in this region and are not particularly important. This is probably because bamboo is more abundant, providing the necessary weaving material for household items such as baskets and mats. Thatching grass is not widely used in the study area, with fewer than 3 percent of households having thatched roofs across the forested regions of the study area (Table 55) according to Rwandan and Ugandan household census data (NISR, 2012; UBOS, 2018).

Table 55. Proportion of rural households harvesting non-woody raw materials within each country in the Albertine Rift Forests study region and the estimated demand per average household per year

COUNTRY	BAMBOO		REEDS AND SEDGES		THATCHING GRASS	
	% RURAL HH	CULMS/HH/Y	% RURAL HH	KG/HH/Y	% RURAL HH	KG/HH/Y
Uganda	64	15	15	2.6	3	1.0
Rwanda	35	8	15	2.6	0.1	0.1
Burundi	48	11	15	2.6	0.1	0.1

Collection of wild fruits, vegetables, and mushrooms is a commonly mentioned activity for rural households across the study area (

Table 56). However, studies providing quantitative estimates of demand appear to be scarce. As in other regions of Africa, edible wild plants provide a valuable dietary supplement or a subsistence income source for households that sell harvested plant foods (Cunningham, 1996). Collection of wild plant foods appears to be limited across some of the forested areas of the study area. For example, around Uganda's Bwindi National Park, only a few wild plant species are harvested for food, mostly carried out by the poorest households (Cunningham, 1996). Limited consumption of wild fruits around Kibira National Park in Burundi has also been noted (Nzigidahera, 2006). Consumption of wild fruits and vegetables appears to be higher in the miombo woodland in the far south of the study area (Rwamahe, 2008). Wild mushrooms also provide a valuable protein-rich nutritional supplement in the study area (Table 56, Degreef *et al.,* 2016), but quantitative data on household consumption was limited to only a few studies. Ndayambaje (2002) reported 32 percent of respondents harvested wild mushrooms around Rwanda's Nyungwe National Park.

Use of plant medicines is thought to be relatively high in some parts of the study area (Table 576), where access to modern medicine is generally lacking. Additionally, many in the region regard traditional plant medicines to be more effective than modern ones (Twinamatsiko *et al.,* 2014). In Uganda, it has been estimated that 80 percent of the population relies on traditional medicines (Kanabahita, 2001). However, not all households harvest their own medicines, as many instead rely on traditional medicine practitioners. This may underlie high variability in estimates of proportions of households harvesting medicinal plants across different areas of the study region, from as low as 16 percent around Bwindi National Park (Harrison *et al*., 2015) to 33 percent around Volcanoes (Nahayo, Ekise & Niyigena, 2013) and 49 percent around Kibale (Hartter, 2010) National Parks.

Table 56. Proportion of rural households harvesting wild plants foods and medicines within each country in the Albertine Rift Forests study region and the estimated demand per average household per year

COUNTRY	WILD PLANT FOODS		MEDICINES		MUSHROOMS	
	% RURAL HH	KG/HH/Y	% RURAL HH	KG/HH/Y	% RURAL HH	KG/HH/Y
Uganda	20	13.6	26	1.3	39	3.8
Rwanda	48	32.4	20	0.1	17	1.7
Burundi	58	39.2	2	0.1	21	2.1

Honey is harvested by communities across the study region, though proportions of households involved in the harvesting are generally low (Table 57), with estimates ranging from 5-25 percent across different parts of the study area (Ndayambaje, 2002; Hatfield & Malleret-King, 2007; Rwamahe, 2008; Harrison *et al.*, 2015). Some protected areas in the region permit beekeeping, such as Bwindi and Mgahinga National Parks, where individuals with permits are legally allowed to harvest honey from designated multi-use zones (Harrison *et al.*, 2015; Bitariho *et al.*, 2016). Honey sales have been found to provide significant supplementary income to beekeeping households around Bwindi National Park (Bitariho *et al.*, 2016).

Although largely illegal, bushmeat hunting is also carried out across the study area. Notwithstanding concerns around the sustainability of bushmeat consumption, it does provide an important source of protein to poor rural households who cannot afford meat from domestic animals, and a potentially valuable income source to households that sell it (Tumusiime *et al.,* 2010; Harrison, 2013; Twinamatsiko *et al.,* 2014). Excessive hunting has led to declines and local extinctions of wildlife species in protected areas like Nyungwe National Park (Masozera & Alavalapati, 2004). Estimates of household involvement in bushmeat consumption are highly variable. The illegal nature of bushmeat is likely a contributing factor of this variability, leading to under-reporting of consumption and hunting. Cultural differences, varying availability of bushmeat species, and different levels of law enforcement effectiveness are likely to also underlie the variation in bushmeat consumption estimates. The lowest estimate came from Nyungwe National Park, where 14 percent of surrounding households admitted to consuming bushmeat (Ndayambaje, 2002). Estimates were higher from other areas, with 26 percent of households around Bwindi National Park (Harrison et al., 2015) and approximately 50 percent of households around Volcanoes National Park (Hill, Osborn & Plumptre, 2002a) consuming or hunting bushmeat.

Small-scale fishing is undertaken by only a small number of households, but this activity is locally important in the areas adjacent to Lake Edward and Lake Kivu, as well as along some of the larger rivers. However, catches are reported to be relatively low (NISR, 2010).

Table 57. Proportion of rural households harvesting wild animal resources within each country in the Albertine Rift Forests study region and the estimated demand per average household per year

COUNTRY	MAMMALS, PRIMATES, BIRDS		WILD HONEY		FISH	
	% RURAL HH	KG/HH/Y	% RURAL HH	LITERS/HH/Y	% RURAL HH	KG/HH/Y
Uganda	19	5.3	6	2.0	1.1	0.8

COUNTRY	MAMMALS, PRIMATES, BIRDS		WILD HONEY		FISH	
	% RURAL HH	KG/HH/Y	% RURAL HH	LITERS/HH/Y	% RURAL HH	KG/HH/Y
Rwanda	14	3.9	5	1.1	0.7	4.6
Burundi	14	3.9	5	2.0	1.1	3.3

THE SUPPLY, USE, AND VALUE OF HARVESTED WILD RESOURCES

To briefly recap, the resource use results are the combined product of natural resource stocks, the availability of these resources for harvesting (protected area status), and the local demand for the various resources. Stocks of natural resources per unit area varied according to habitat type and condition. The supply of natural resources was also moderated by protected area status, as we reduced the proportional availability of natural resources where they occurred within protected areas. The magnitude of this reduction varied according to the level of protection. Finally, the data for available stocks per hectare was combined with estimated household demand per hectare. Demand is a function of both the average quantity of resources used per household, and the number of households in the area (population density).

In general, the Albertine Rift Forests region had relatively high use per unit area for most harvested natural resources (Table 58; Figure 69-Figure 73) for two reasons. Firstly, the areas surrounding the wildlife landscape have some of the highest rural population densities in Africa, resulting in high demand for resources. Secondly, forest habitats harbor the highest stock per unit area of many of the harvested resources considered in the study. Hence, more resources are available for harvesting per hectare. The total value of wild harvested resources was estimated to be US$352.2 million across the landscape: US$162.9 million in Uganda, US$139.9 million in Rwanda, and US$49.3 million in Burundi (Table 58).

Sizeable continuous areas of natural resources are mostly limited to the protected areas comprising the wildlife landscape of the region (Figure 69-Figure 73). This reflects the general dominance of cultivation outside of protected areas, with only small patches of more natural habitats remaining. As a result, zero values for natural resource harvesting are widespread outside of the wildlife landscape. Where natural vegetation remains outside of protected areas, substantial demand from dense human populations mean use of natural resources is often very high, indicating severe harvesting pressure on remaining natural habitats.

Our model also estimated relatively high natural resource harvesting in several protected areas within the wildlife landscape (Figure 69-Figure 73). Notable examples include Volcanoes National Park in Rwanda and adjacent Mgahinga Gorilla National Park in Uganda. Both parks have particularly high tourist value due to the presence of gorillas. However, our model suggests that dense surrounding populations, and the relative accessibility of resources due to the small size of these parks, mean natural resources would come under heavy pressure in the absence of adequate protection. Similar reasons underlie the high estimated resource use in Burundi's Kibira National Park. The park's narrowness means the entirety of Burundi's portion of the wildlife landscape is within relatively easy reach of surrounding rural populations, giving rise to much higher average use per hectare values for Burundi when compared to the other countries (Table 58). The lowest values for resource use/ha were associated with Queen Elizabeth National Park and surrounding forest reserves in Uganda and the interior of Nyungwe National Park in Rwanda. Despite high pressure on natural resources on the periphery of these parks, low to no harvesting can be expected in their interior regions, due to the large travel distances that would be required for people living outside of them. The presence of these larger protected areas, with lower harvesting rates, underlies the generally lower average use per hectare values in Uganda and Rwanda compared to Burundi (Table 58).

The Albertine Rift Forests were estimated to have the highest amount of fuelwood and timber harvesting per hectare of all the wildlife landscapes analyzed (Figure 69; Table 58), as would be expected given high population densities and the high woody biomass values of the forest habitats. Fuelwood also had the highest monetary value per hectare of the harvested resources considered, reflecting its importance as an energy source for cooking across most rural households in the area. In contrast, harvesting of wood for poles did not have an especially high per hectare value relative to other wildlife landscapes. This can be attributed to the relatively high use of brick or concrete in house construction.

Bamboo has high use values where it occurs, but its distribution in the region is limited to pockets in Uganda's Rwenzori Mountain National Park, several pockets in southern Uganda and northern Rwanda, and Kibira National Park in Burundi (Figure 71). Burundi has a much higher average value for bamboo use per hectare because bamboo covers a much greater proportion of Burundi's wildlife landscape than it does in Rwanda and Uganda. Similarly, we estimated limited occurrence of reeds and sedges across the region (Figure 70), resulting in low average use per hectare (Table 58). While isolated harvesting areas are dotted across the region, the largest reed and sedge harvesting zones are associated with wetland habitats in the northern part of the region. Use values of thatching grass are low in the Albertine Rift Forests region (Figure 71; Table 58), particularly in Rwanda and Burundi where only a minority of houses have thatched roofs. Furthermore, since we assumed that thatching grass does not grow in intact forest habitats, zero values for thatching grass use prevail across many of the region's protected areas where forest dominates. Hence, no thatching grass use was predicted for the Rwandan and Burundian portions of the wildlife landscape, though some use outside of the wildlife landscape was estimated.

Use of wild plant foods and medicines is relatively high and was estimated to have the next highest monetary value per hectare after fuelwood (Table 58). Honey is not a particularly popular resource in the region, but still has a relatively high use value/ha due to the aggregate demand from the high population densities. Finally, the largest continuous areas of fish use were predicted along the shores of the Rift Valley lakes, along with isolated areas of fish stocks in wetland areas throughout the region (Figure 73).

Table 58. Average quantities, monetary values per hectare, and total value (US$ millions) for subsistence harvesting of wild resources in the Albertine Rift Forests wildlife landscape

RESOURCE		*UGANDA*			*RWANDA*			*BURUNDI*		
	UNIT	USE (UNIT/HA)	US$/HA	TOTAL US$ MN	USE (UNIT/HA)	US$/HA	TOTAL US$ MN	USE (UNIT/HA)	US$/HA	TOTAL US$ MN
Fuelwood	m^3	0.93	29.04	100.1	1.02	31.85	103.7	2.10	65.74	34.3
Poles & withies	m^3	0.03	0.61	2.1	0.01	0.35	1.0	0.03	0.75	0.4
Timber	m^3	0.04	4.86	15.9	0.01	2.87	6.7	0.05	6.46	2.6
Thatching grass	kg	0.11	0.04	0.3	-	-	>0.01	-	-	>0.01
Reeds & sedges	kg	0.46	0.31	7.4	0.19	0.13	8.0	0.47	0.31	1.3
Bamboo	culms	1.60	1.12	1.0	1.31	0.92	0.3	10.28	7.20	0.3
Wild plant foods & medicines	kg	8.97	18.23	35.0	12.63	13.61	19.5	38.95	28.80	10.1
Bushmeat	kg	0.93	1.07	0.6	0.76	0.88	0.1	1.30	1.49	0.1
Honey	l	0.17	0.27	0.5	0.11	0.18	0.2	0,23	0.37	0.1
Fish	kg	0.07	0.04	0.15	0.05	0.03	0.42	0.02	0.01	0.01

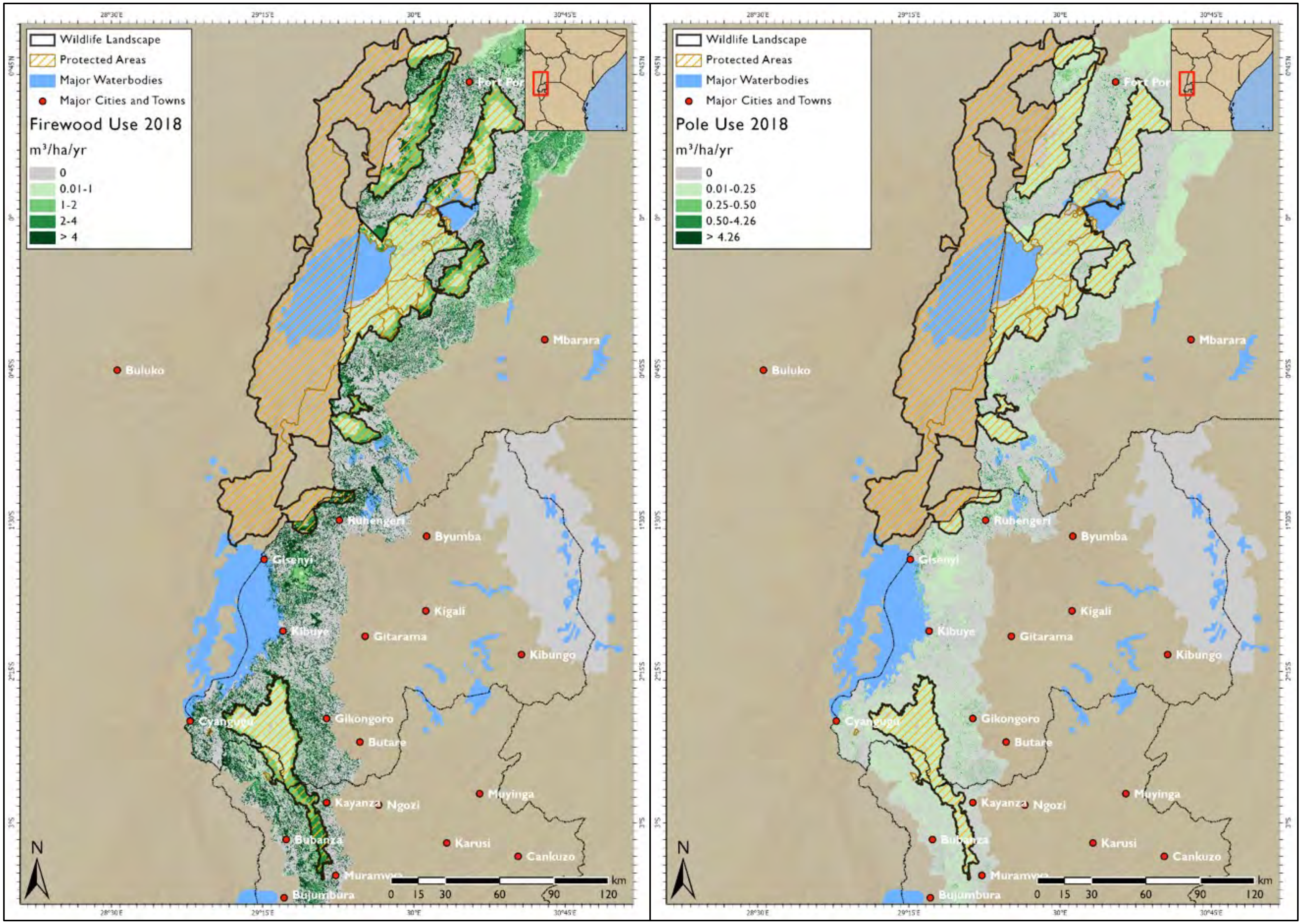


Figure 69. Estimated variation in the subsistence harvesting of fuelwood (left) and poles (right) across the Albertine Rift Forests region

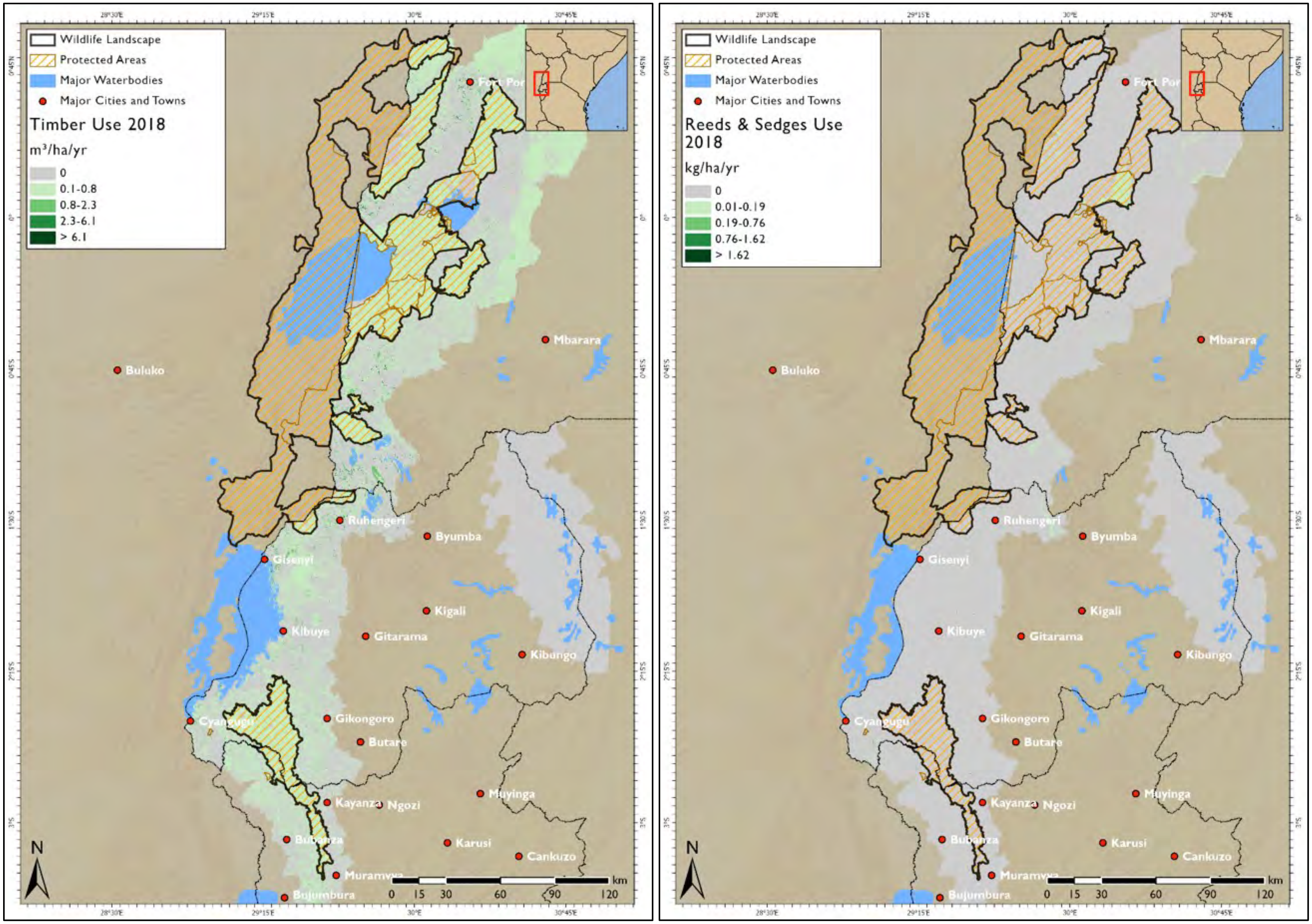


Figure 70. Estimated variation in the subsistence harvesting of timber (left) and reeds and sedges (right) across the Albertine Rift Forests region

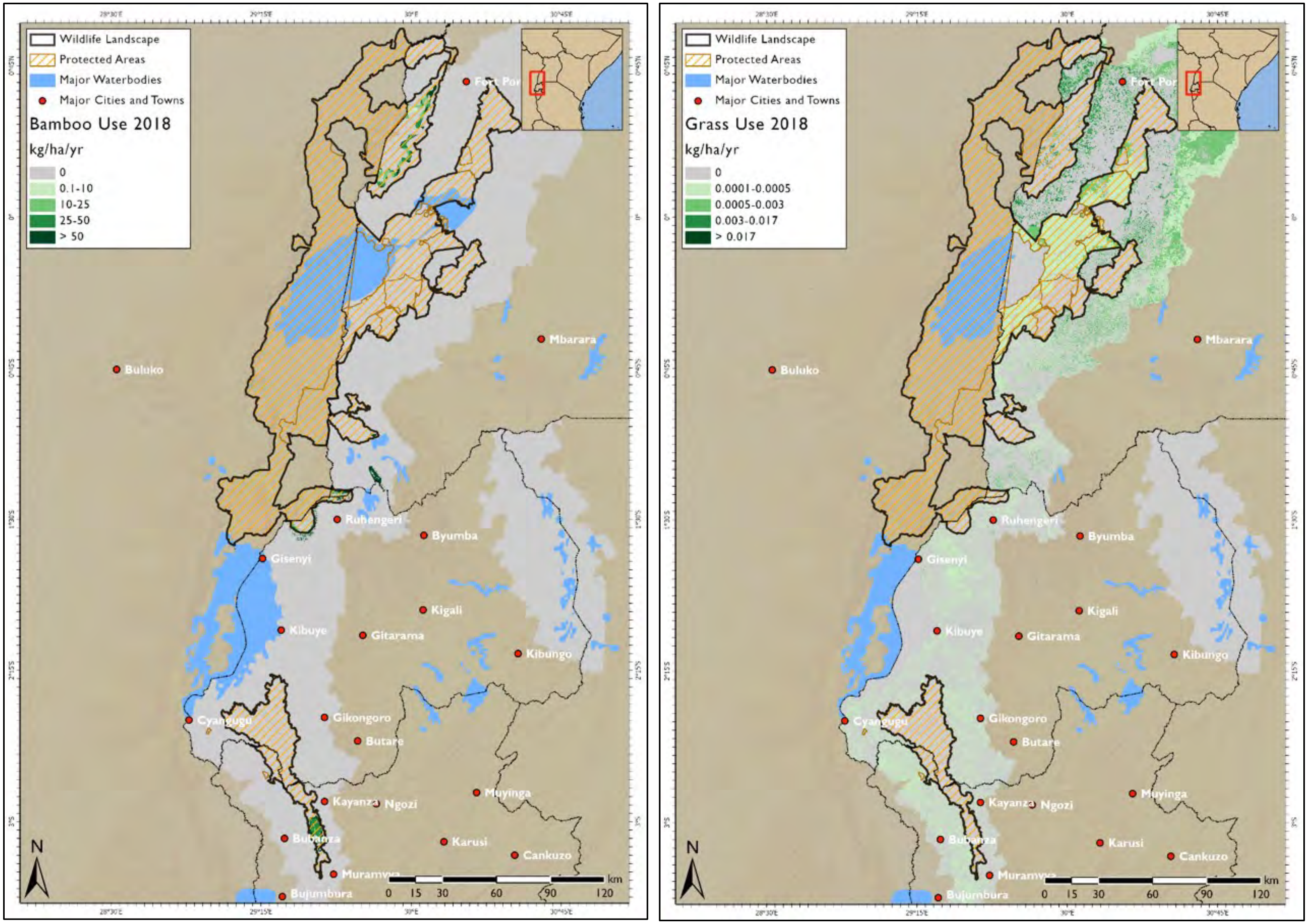


Figure 71. Estimated variation in the subsistence harvesting of bamboo (left) and thatching grass (right) across the Albertine Rift Forests region

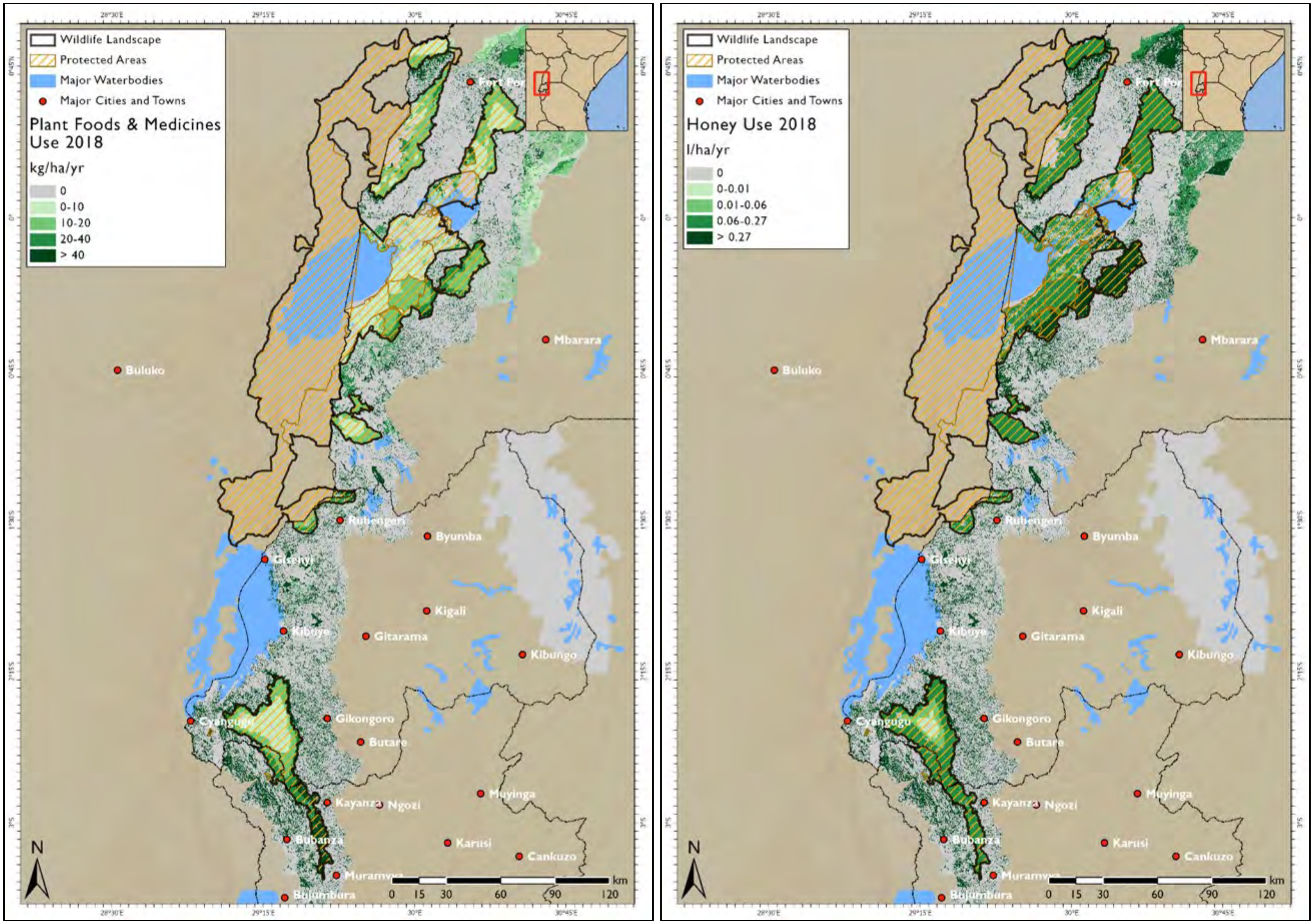


Figure 72. Estimated variation in the subsistence harvesting of plant foods and medicines (left) and honey (right) across the Albertine Rift Forests region

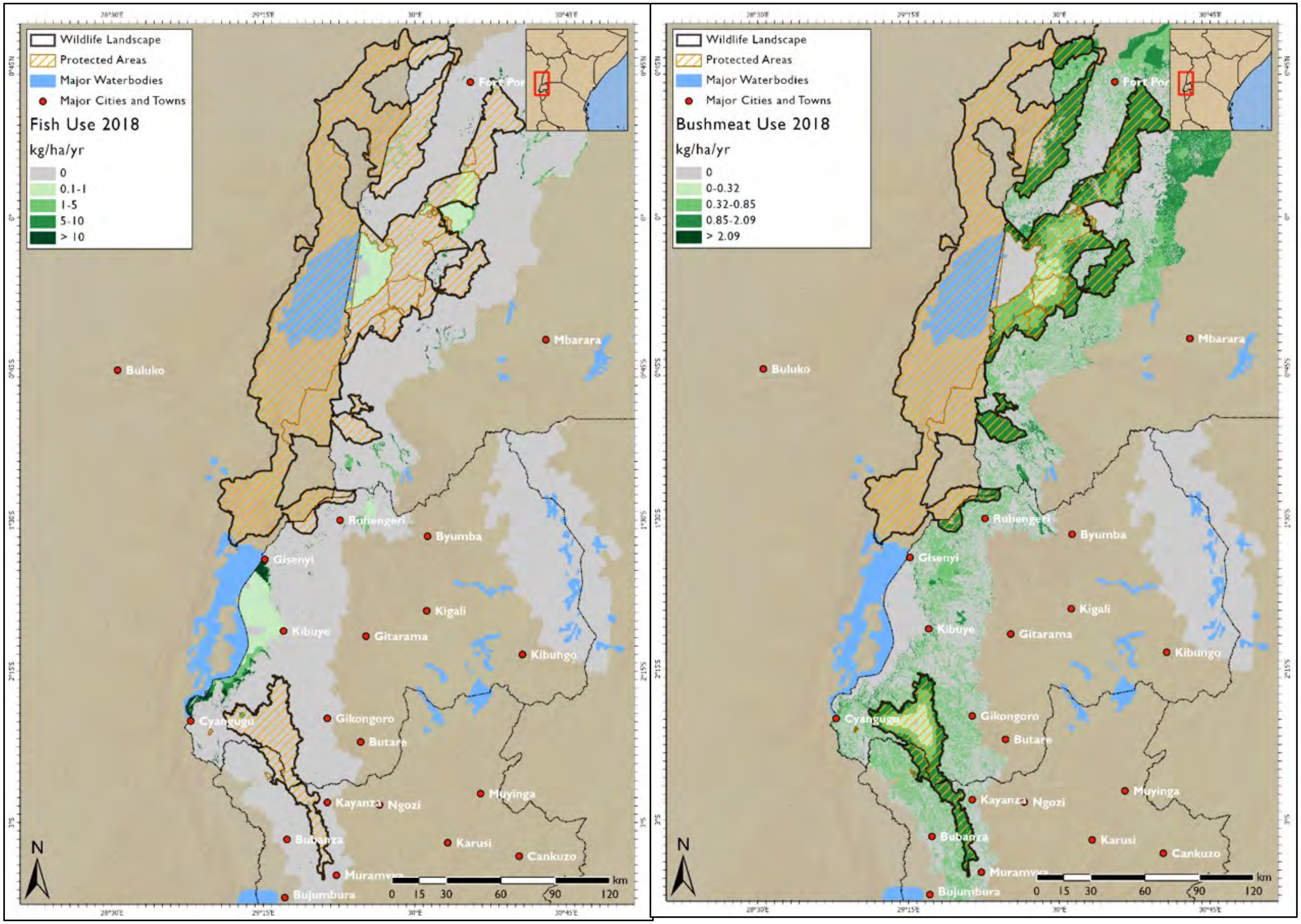


Figure 73. Estimated variation in the subsistence harvesting of fish (left) and bushmeat (right) across the Albertine Rift Forests region

SUMMARY

The total direct contribution to GDP of nature-based tourism was estimated to be US$50.3 million in 2018, mostly associated with the Queen Elizabeth, Mgahinga Gorilla, and Volcanoes National Parks. The national parks in this region generate US$27.6 million or 4 percent of tourism value in Uganda, US$11.8 million or 3 percent in Rwanda, and US$0.4 million or 1 percent in Burundi. This excludes multiplier effects. Nature-based tourism also generates an estimated $83 million in net benefits to international visitors.

Keeping the forest habitats of this landscape in their current natural condition generates costs savings for the region that could be worth about US$2.6 billion per year through regulation of hydrological processes and atmospheric carbon. Based on our high-level modeling exercise, these habitats are estimated to retain about 498 million tons of sediment per year, this service having a replacement cost value of US$612 million per year. The forest habitats also reduce phosphorous loadings by an estimated 165-2,244 tons per year (depending on the alternative land use), with a replacement value of between US$331,360 and US$682,469 per year. It appears that these forests do not have a smoothing effect on baseflows, however. These estimates should be refined in future with more detailed modeling at finer scales, and with the provision of reliable monitoring data on environmental processes in Uganda, Rwanda, and Burundi, and should ideally be extended to incorporate the DRC. The Albertine Rift Forests also store an estimated 643 million tons of carbon, the retention of which, according to the most recent estimates, would avoid local climate change damages of US$63 million per year. In addition, retention of these carbon stocks avoids damages of about US$42 billion per year at a global scale.

Table 59. Summary of the benefits derived from ecosystem services of the Albertine Rift Forests wildlife habitats. All values in US$ millions per year

	BURUNDI	RWANDA	UGANDA	REGION	REST OF WORLD	TOTAL
Nature-based tourism	0.5	13.3	36.5	50.3	83	134
Biodiversity existence	-	0.0	0.1	0.1	322	322
Erosion control	65.9	128.8	417.1	611.8	-	612
Water quality amelioration	0.2	0.2	0.2	0.5	-	0.5
Carbon storage	0.1	4.6	57.8	62.6	42,216	42,279
Crop pollination	4.7	3.4	28.2	36.3	-	36
Harvested resources	49.3	139.9	162.9	352.1	-	352
Total value $ millions per year	**120.7**	**290.2**	**702.8**	**1,113.7**	**42,622**	**43,736**
Total value $ per ha per year	**2,554.5**	**2,462.6**	**1,148.1**	**1,432.9**	**54,838**	**56,271**

The forest habitats also contribute to agricultural production around the margins of this landscape. Wild pollinators were estimated to increase crop production by some US$36.2 million per year. The wild resources harvested from these habitats are worth an estimated US$352.2 million per year. Including conservative estimates of the existence value of biodiversity, the wildlife landscape is estimated to be worth at least $1,430/ha/year on average to East Africa, and over $56,000/ha/year globally. For validation, it is interesting to compare our estimate of existence value with that of Hatfield & Malleret-King (2007). Their study estimated the WTP for biodiversity in the same area to be $1,870 million, and the authors opted to assume that the figure might more realistically be 10 percent of this. Our estimate falls within that range, on the more conservative side.

IMPLICATIONS FOR THE FUTURE UNDER A BAU SCENARIO

The Albertine Rift region has been substantially transformed by human activities. Nevertheless, the remaining natural habitats that comprise the transboundary landscape still retain a wealth of biodiversity, including a host of endemic species. The severe competition for land and resources between wildlife and people, combined with substantial impacts predicted under future climate change, make the future of this transboundary landscape particularly precarious. The following sections describe some of the expected impacts of a range of existing pressures based on past trends, as well as some of the expected impacts of future climate change derived from modelling studies. Finally, there is a discussion on existing pressures and future climate change impacts to predict the future of wildlife, habitats, and ecosystem services provided by the transboundary landscape under a BAU scenario.

HABITAT CONVERSION

Favorable climatic conditions for agriculture have promoted high population densities and extensive conversion of natural habitat to cultivation throughout the Albertine Rift region (Kayanja & Byarugaba, 2001; Plumptre *et al.*, 2016; Salerno *et al.*, 2018). Confronted with ever increasing food demand, farmers across the region have been forced to cultivate more land in an effort to improve their productivity (Salerno *et al.*, 2018). Expansion of large-scale industrial agriculture such as sugar, rice and tobacco has also become a major cause of forest loss in recent decades, particularly in the Ugandan Albertine Rift (Twongyirwe *et al.*, 2015). The controversial decision to allow 5 500 ha of Uganda's Bugoma Forest Reserve to be cleared for sugarcane provides a well-publicized recent example of the threat posed by these activities. Intensive cropping has already expanded right to the edges of protected areas in much of the landscape. In the Burundian and Rwandan portions of the landscape in particular, very little natural habitat remains outside of protected areas (Plumptre *et al.*, 2016). Meanwhile, clearance of buffer zones and unprotected forests in Uganda is driving disappearance of remaining wildlife corridors and migratory routes outside protected here too. Key informants also noted immigrants from the DRC settling in forests as an additional cause of habitat loss, particularly as such people are generally poor and thus highly dependent on forest resources. Overharvesting of woody biomass for fuel and construction purposes places further pressure on woody habitats, representing another major threat to the Albertine Rift wildlife landscape. Oil and gas exploration also poses a threat in the Ugandan portion of the landscape especially, where large-scale land acquisitions have occurred in support of these activities. Key informants from local communities in the landscape also noticed that there has been some breakdown of traditional norms and taboos surrounding the protection of sacred spaces and certain wildlife species. This has reportedly been driven in part by immigrant populations with different values and belief systems.

With little room left for further expansion of agriculture in much of the landscape, pressure to convert protected areas to cultivation will intensify with future population growth. While the larger protected areas in the landscape generally remain resilient to substantial habitat loss currently, this has not always been the case. For example, multiple armed conflicts over the past decades have taken a toll on natural habitats in the region, starting with war in Uganda in the 1970s and 1980s, followed by the Rwandan civil war in the 1990s (Kanyamibwa, 1998; Petursson *et al.*, 2013). In Rwanda, a lack of conservation law enforcement during the civil war, and the influx of returning refugees following the war, resulted in substantial loss of forest to agriculture in Gishwati Forest Reserve, Nyungwe National Park, and other, smaller protected areas (Kanyamibwa, 1998). Worryingly, the Global Forest Change dataset indicates a clear upsurge in deforestation rates in the Albertine Rift Forests landscape since 2014 (Figure 74), indicating that habitat loss is once again on the rise in the region. From 2001 to 2013, annual deforestation rarely exceeded 500 ha across the landscape. In contrast, deforestation from 2014-2019 exceeded 1,000 ha per year in all but one year.

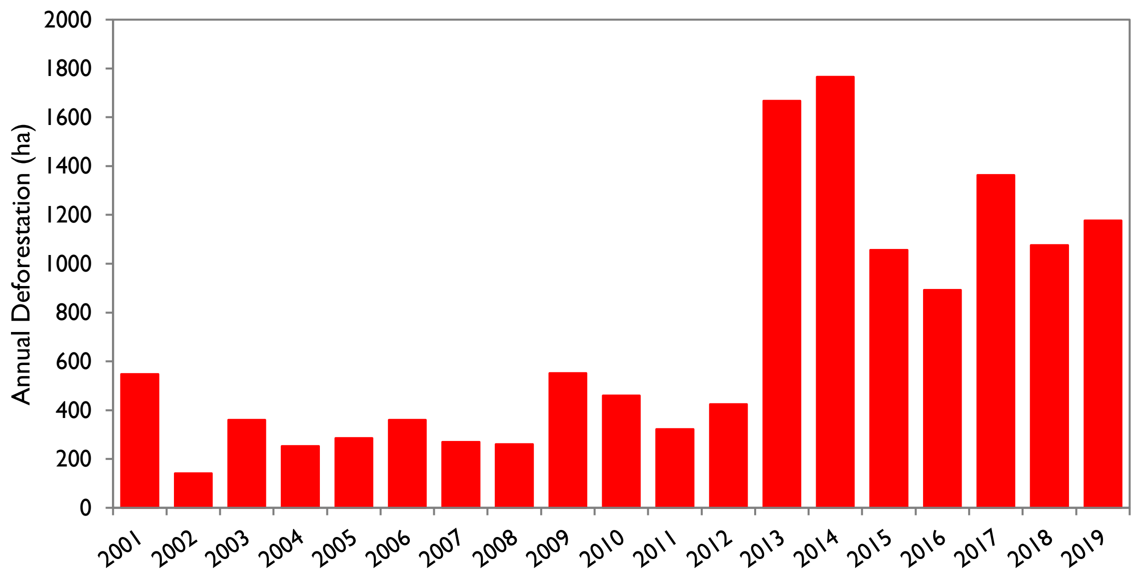


Figure 74. Annual forest loss in the Albertine Rift landscape from 2001 to 2019

Source: Hansen/UMD/Google/USGS/NASA

Habitat loss has led to the isolation of wildlife populations within protected areas, disrupting historical migration and dispersal routes. A number of protected areas have effectively become islands in a "sea of agriculture" (Caro & Davenport, 2016; Salerno *et al.*, 2018). This is particularly pronounced for parks like Nyungwe/Kibira and Bwindi Impenetrable National Parks, which are surrounded by agriculture on all sides and thus completely disconnected from other forests in the region. The resulting inability for species to move in response to climate change will be of increasing concern in the future (Ponce-Reyes *et al.*, 2017), as is discussed further below. Furthermore, the importance of retaining remaining landscape connectivity has already been demonstrated. For example, the connectivity between protected areas like Volcanoes and Queen Elizabeth National Parks with the DRC appears to have been vital for helping to maintain populations of large mammals in this part of the landscape. It is thought that wildlife used this connectivity between Rwanda, Uganda, and the DRC to move in response to increased anthropogenic pressures, like the various wars that have occurred in the region (Plumptre *et al.*, 2007, 2016). Unfortunately, substantial habitat loss is occurring in the DRC's Virunga National Park due to armed conflict, the expansion of cultivation and settlement, mining, and oil and gas exploration (Plumptre *et al.*, 2016, 2017; Christensen & Arsanjani, 2020). This threatens to further reduce landscape connectivity in the region in the future.

Overall, Plumptre *et al.*, (2017) estimate that, on average, species ranges have declined by 55 percent from habitat loss due to agriculture and infrastructural development in the region. Furthermore, since this estimate includes the DRC where more extensive natural habitat remains, average reduction in species ranges is likely to be higher when only the Burundi, Rwanda, and Uganda portions of the landscape are considered.

The ESA CCI land cover data at 300 m resolution suggest that the area under crops expanded until the mid-2000s, then again during 2010 to 2015, but has decreased since then (Table 41). The Copernicus 100 m landcover data series, which goes back to 2015, suggests that there has been an increase in cropland in the study area from 2015 to 2018 of 456 hectares per year. This highlights the potential inaccuracy of land cover data products and the need for ground-truthing. Given the information in the literature, the latter trend is more likely. The built-up area has increased steadily, which is a concern that most of this landscape is ostensibly under protection.

Table 60. Extent and annual rates of change of land cover classes in the Albertine Rift Forests region from 1992 to 2004 and from 2004 to 2018

LAND COVER CHANGE	1992 TO 1998	1998 TO 2004	2004 TO 2010	2010 TO 2015	2015 TO 2018
Average annual change in area under crops (ha/year)	16	123	-304	352	-40
Average annual change in built-up area (ha/year)	8	5	4	2	6

Source: Based on ESA CCI Land Cover 300m resolution (European Space Agency, 2018)

OVERHARVESTING OF NATURAL RESOURCES AND OVERGRAZING

Dependence on natural resources among the dense populations living in the region has also driven degradation of remaining habitats. High dependence on fuelwood and scarcity of wood in unprotected lands drives people living near protected areas to harvest wood illegally inside protected areas (Harrison *et al.*, 2015; Plumptre *et al.*, 2016). Additionally, charcoal production for growing urban markets is expanding (Naughton-Treves *et al.*, 2007). Due to its reliance on slow-growing, indigenous hardwood species, rising charcoal demand is of particular concern for the health of the region's forests. Brick production, which depends on wood as a fuel, is also rising to meet the construction demands of expanding urban and semi-urban areas (Naughton-Treves *et al.*, 2007). In the absence of alternative fuel sources, these harvesting pressures will likely increase as populations continue to grow and wood outside protected areas becomes scarcer. There is evidence that wood supply is already substantially lower than demand. For example, Drigo *et al.*, (2013) estimated the total demand for woody biomass for fuel and construction in Rwanda to be 4.8 million tons, while the sustainable supply potential was estimated to be just 3.2 million tons. Such a supply deficit would need to be addressed to avoid further degradation of forests. Illegal harvesting of timber is another driver of degradation of forest habitats in the landscape (Kayanja & Byarugaba, 2001).

Over-harvesting of other forest resources has also caused habitat degradation. For example, harvesting of forest climbers and bamboo has modified habitat in some forests (Muhwezi *et al.*, 2009; Plumptre *et al.*, 2016). Encroachment into protected areas to harvest resources appears to be particularly serious in the Burundi portion of the landscape, based on key informant interviews conducted around Kibira National Park. Use of forests by local people has also resulted in fires, which degrade and destroy forest habitat. For example, about 12 percent of Nyungwe's forests were burned in the early 2000s, leading to a replacement of forest with invasive open bracken fern-dominated habitat (Plumptre *et al.*, 2016). Fire was also widely reported as a major threat to the forest by key informants around Kibira National Park. Human disturbance of canopy cover in Nyungwe has also promoted the spread of invasive species. The most notable of these is a climber (*Sericostachys scandens*) that smothers trees and regenerating forest, resulting in increased tree mortality and slower forest regeneration (Plumptre *et al.*, 2016)

Livestock grazing is a cause of habitat degradation in parts of the landscape. This occurs most notably in Queen Elizabeth National Park, where the savanna habitats are more attractive for cattle grazing than the forests that cover much of the rest of the landscape. A 2010 aerial census revealed that cattle were the most abundant species in Queen Elizabeth National Park, raising concern that they are reducing food availability for wild grazing species like Ugandan kob and buffalo (Plumptre *et al.*, 2010a).

HUNTING PRESSURE

Excessive hunting pressure has had a severe impact on wildlife populations in the landscape. In particular, periods of armed conflict have been associated with rampant poaching, facilitated by the use of automatic weapons and lack of conservation law enforcement (Kanyamibwa, 1998; Plumptre *et al.*, 2007). Wars also leave people in the area poor and destitute, forcing them to rely heavily on bushmeat to supplement their diets with animal protein and their incomes with illegal trade in wildlife (Hill, Osborn & Plumptre, 2002b; Plumptre *et al.*, 2017). While hunting pressures are more severe during period of war, hunting pressures are still a serious threat to wildlife in the landscape during peacetime (Plumptre *et al.*, 2016).

Excessive hunting has resulted in the extirpation of several species from the region's protected areas, even where sizeable suitable habitat remains. For example, buffalo in Nyungwe/Kibira were hunted to extinction some time ago, while the last elephant was recorded in the forest in the early 2000s (Plumptre *et al.*, 2002). Similarly, in Bwindi Impenetrable National Park, buffalo and leopard were hunted to extinction in the 1970s and 1980s, while the elephant population was drastically reduced (Butynski, 1984). Commercial hunting of wildlife for the illegal wildlife trade is another threat to certain species in the landscape, primarily targeting elephant and hippopotamus for their ivory, as well as pangolin for their scales (Plumptre *et al.*, 2016; Travers, 2017). While not highly targeted for bushmeat hunting in the region, gorilla and chimpanzee may still be caught in snares incidentally, while young individuals are targeted for sale to zoos and the pet trade (Plumptre *et al.*, 2010b; Rossi, 2018).

Although poaching in protected areas in Rwanda and Uganda has been significantly reduced in recent times, it was still widely noted as a threat by key informants. Furthermore, there is concern that recent gains could be reversed by poachers crossing the border from the DRC, where enforcement has been hampered by insecurity and armed conflict. Meanwhile, key informants from Burundi consistently flagged hunting as a key threat to Kibira, with community leaders in certain areas around the park noting that wildlife species had totally disappeared. Community informants in Uganda also noted wildlife had virtually disappeared from their areas, indicating increases in isolation of wildlife within protected areas. A number of key informants suggested that rangers were too few in number and/or underpaid, reducing their capacity to enforce regulations against bushmeat hunting. Improved patrolling and stricter enforcement of anti-poaching laws was thus recommended as a way of controlling unsustainable bushmeat harvesting. Other recommendations for addressing the issue included diversification of household income generation to reduce reliance on hunting and improved transboundary cooperation in anti-poaching patrols and law enforcement.

HUMAN WILDLIFE CONFLICT

Human wildlife conflict (HWC) has also contributed to the decline and disappearance of large wildlife. Due to the expansion of cultivation and dense human populations up to the borders of protected areas, HWC can be a serious problem for local people, potentially leading to retaliatory killing of problem wildlife species (Hill *et al.*, 2002b; Tolbert *et al.*, 2019). For example, in the savanna portion of the landscape, conflict with livestock owners is said to be the chief cause of mortality for large predators like lion and hyena (Plumptre *et al.*, 2016). There can be serious HWC in the forest portions of the landscape too. For example, around Volcanoes National Park, 91 percent of respondents living adjacent to the park said they were affected by crop damage from wild animals (Hill *et al.*, 2002b). However, Rwanda's Special Guarantee Fund, which allocates 5 percent of nature-based tourism revenue to compensation for victims of HWC, appears to have improved perceptions of wildlife in some communities living adjacent to parks (Tolbert *et al.*, 2019). Conversely, interviews with community informants in Uganda suggest perceptions of wildlife are more negative among communities affected by HWC. Key informants complained that the UWA can take too long to respond to problem animal reports, forcing communities to take action themselves to avert further loss of crops or livestock. While Uganda has proposed laws around the compensation of people affected by HWC, these are yet to be gazetted. However, some compensation of affected individuals is provided by conservation NGOs working in the area. Human-wildlife conflict is likely to worsen in the future, as population growth and shortage of land prompts more people and livestock to live adjacent to and encroach inside protected areas.

OVERALL IMPACT ON WILDLIFE POPULATIONS

The combined effects of these pressures have already led to substantial loss of wildlife across the landscape. Due to extensive habitat transformation, large wildlife populations are now limited to isolated pockets in protected areas. Even in these, certain wildlife species are often absent, despite the presence of suitable habitat. For example, the extinction of buffalo from parks like Nyungwe/Kibira and Bwindi Impenetrable has already been noted (Butynski, 1984; Plumptre *et al.*, 2002). However, where wildlife species have been able to survive past habitat loss, war, and rampant poaching, populations have often shown stability or even recovery in recent decades. A good example of this is that of mountain gorilla numbers, which have increased consistently since the 1980s despite their range having been substantially reduced by habitat loss (Plumptre *et al.*, 2007; Gray *et al.*, 2013; McGahey *et al.*, 2013). This recovery has been attributed to intensive transboundary and international conservation efforts, as well as the economic value of the species. Similarly, the isolated elephant population in Bwindi Impenetrable National Park has recovered from the brink of extinction, increasing from just 25-30 individuals in 1986 (Babaasa, 2000) to 106 in 2019. Nevertheless, the pressures mentioned above mean the future of large wildlife populations remains precarious, especially those in isolated protected areas. Furthermore, climate change is predicted to pose an increasing threat to wildlife in the future, as discussed further below.

PROJECTED CHANGES IN TEMPERATURE AND RAINFALL

Tropical forests are considered resilient to climate change, provided they remain intact (Huntingford *et al.*, 2013). The ability of tropical forests to produce the rainfall required to sustain themselves declines following deforestation; once this tipping point has been reached, these forests will begin to change into a savanna-type ecosystem. Apart from releasing stored carbon into the atmosphere, deforestation renders tropical forests more susceptible to climate change. Indeed, the significant temperature increase around Kibale National Park in Uganda has been attributed to the largescale deforestation and drainage of wetlands in the area, rather than global climate change, which shows that the impacts of climate change will be significantly exacerbated by land cover conversion from tropical forests (Plumptre *et al.*, 2017).

Notwithstanding further land cover changes and changes in forest cover, total annual precipitation across the Albertine Rift Forests for the period 2040-2060 is expected to increase by 1.9 percent relative to historical (1960-1990) precipitation (Figure 75), and mean annual temperature is expected to increase by 2.7°C (Figure 76), in line with observed changes (Plumptre *et al.*, 2017) and climate predictions (*e.g.*, Phillips & Seimon, 2009; Seimon & Phillipps, 2010; Seimon, Picton-Phillips & Plumptre, 2011). Precipitation across the Albertine Rift Forests features a bimodal distribution with wetter periods occurring from March to May and August to November (Taylor *et al.*, 2008; Figure 75). The bimodal pattern results from the regional movement of air masses associated with the Intertropical Convergency Zone. Predictions are for the August to November period, the short rainy season, to get wetter with increased risk of flash floods and landslides, while the long rainy season, March to May, is predicted to get marginally drier (Figure 75). Geographically, the northern and western parts of the landscape are expected to experience the largest increases in annual precipitation relative to the remainder of the landscape (Figure 77). The entire landscape stands to experience a similar increase in mean annual temperature.

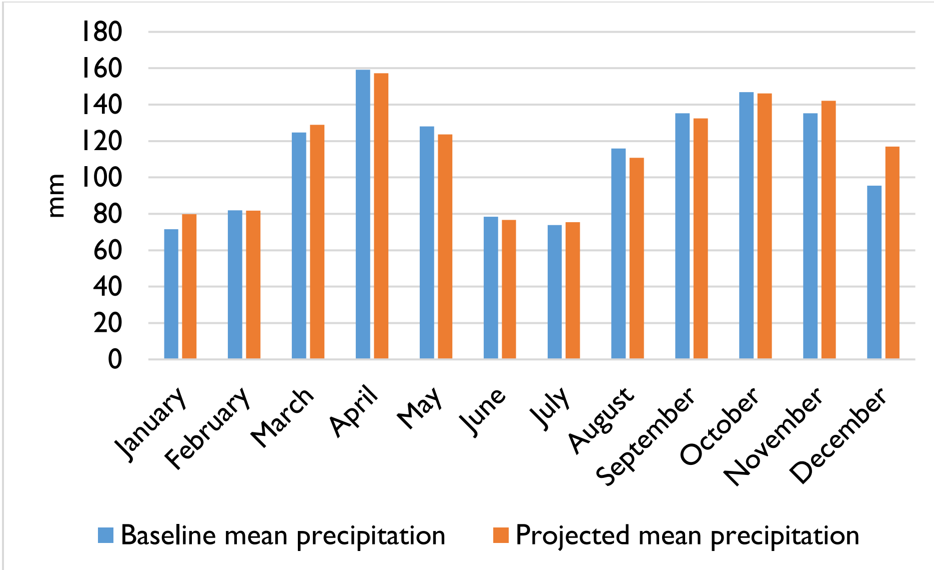


Figure 75. A comparison between historic and projected monthly precipitation (mm) for the Albertine Rift Forests

Source: Based on data from WorldClim Version2 and CMIP5

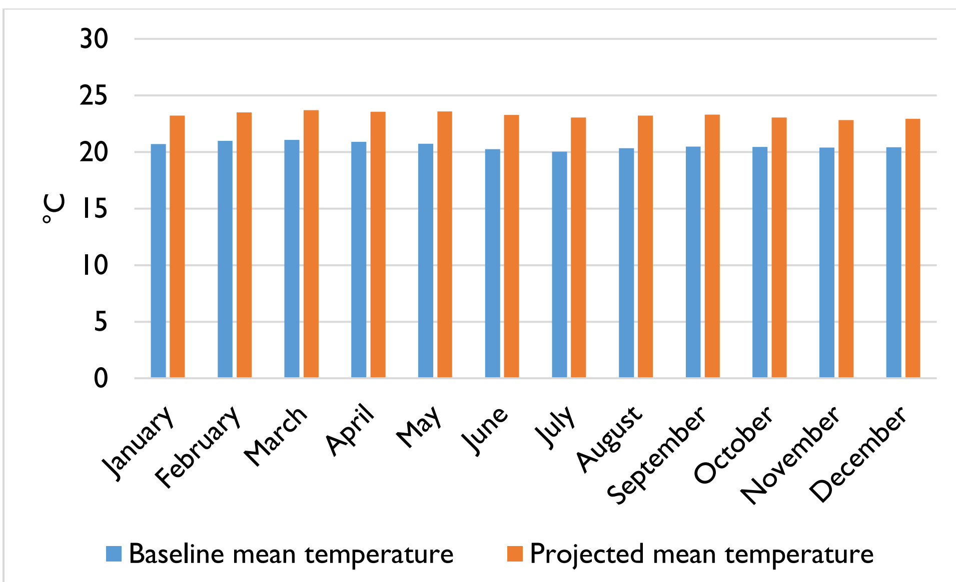


Figure 76. A comparison between historic and projected mean monthly temperature (°C) for the Albertine Rift Forests

Source: Based on data from WorldClim Version2 and CMIP5

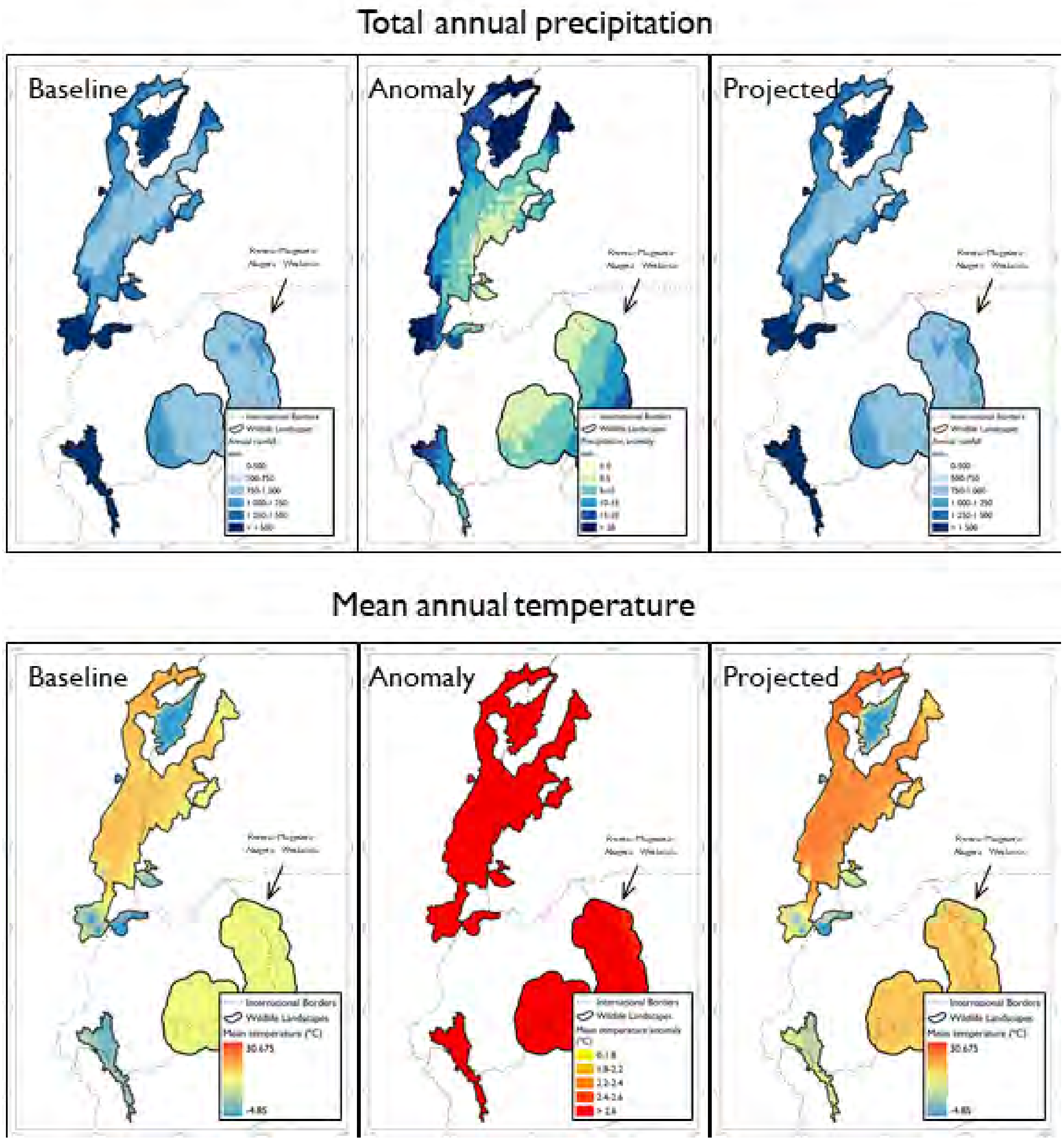


Figure 77. Baseline/historic (1960 – 1990) and projected (2040 – 2060) total annual precipitation (mm) and mean annual temperature (°C) across the Albertine Rift Forests landscape

Source: Based on data from WorldClim Version2 and CMIP5

Although at the landscape scale the annual precipitation and temperature projections are for a marginal increase in total annual rainfall of 1.9 percent and a considerable increase in mean annual temperature difference of 2.7°C, the change in precipitation could differ markedly across the landscape in a few decades time. Changes in mean annual temperature is consistent across the landscape, possibly due to its proximity to the equator. Table 61 provides the projected change in mean annual temperature (°C) and total annual precipitation (mm) for important protected areas in the wildlife landscape. The expected increase in mean annual temperature is 2.7°C across all protected areas. Those protected areas with lower mean annual temperatures experience greater relative increases. The protected areas in mountainous areas, namely Volcanoes, Rwenzori Mountains, and Mgahinga Gorilla are examples of these. The expected change in total annual precipitation ranges from an increase of 0.3 percent for Bwindi Impenetrable and Mgahinga Gorilla National Parks in Uganda to an increase of 1.9 percent for Semuliki National Park in Uganda.

Table 61. Historic, projected, and percentage changes for mean annual temperature (°C) and total annual precipitation (mm) for key protected areas in the Albertine Rift Forests study area

	MEAN TEMPERATURE (°C)			MEAN PRECIPITATION (MM)		
PROTECTED AREA	HISTORIC ANNUAL AVG.	PROJECTED ANNUAL AVG.	CHANGE	HISTORIC ANNUAL TOTAL	PROJECTED ANNUAL TOTAL	% CHANGE
Queen Elizabeth	23.0	25.7	2.7	940	946	0.6
Nyungwe	15.5	18.2	2.7	1,653	1,669	0.9
Rwenzori Mountains	11.6	14.4	2.7	1,657	1,686	1.8
Kibale	21.1	23.7	2.7	1,201	1,222	1.7
Kibira	16.4	19.2	2.7	1,669	1,679	0.6
Bwindi Impenetrable	17.1	19.8	2.7	1,330	1,334	0.3
Kigezi	22.4	25.1	2.7	1,025	1,029	0.4
Semuliki	24.4	27.1	2.7	1,142	1,163	1.9
Volcanoes	11.0	13.7	2.7	1,871	1,883	0.6
Kyambura	23.1	25.7	2.7	918	924	0.6
Mgahinga Gorilla	12.9	15.6	2.7	1,690	1,695	0.3
Kazinga	23.3	26.0	2.7	880	885	0.6

Source: Based on data from WorldClim Version2 and CMIP5. Protected areas are listed in descending order of area.

PROJECTED CLIMATE CHANGE IMPACTS ON HABITATS AND WILDLIFE

Climate change is a major threat to biodiversity, affecting both individual species and overall ecosystem functioning (Scheffers *et al.*, 2016). To survive a shift in suitable climate, species may need to either adapt to their changed environment or relocate to more suitable areas (Moritz & Agudo, 2013). However, opportunities to move may be restricted by anthropogenic or natural barriers such as cultivated land, mountain ranges, or water bodies. This challenge is particularly pronounced in the Albertine Rift landscape, where protected areas are often surrounded by intensive cultivation. In montane habitats like much of the Albertine Rift, species ranges are generally predicted to move upslope in response to future climate change (Ayebare *et al.*, 2018). These ecological shifts will exacerbate the challenge of conserving species and ecosystems in the landscape. Such shifts have already been observed in bird distributions in the region (Byrne, 2016). Due to the broader climatic envelope they provide, mountainous areas with a high altitudinal range such as Rwenzori Mountains, Volcanoes, Bwindi Impenetrable, and Mgahinga Gorilla National Parks are thought to be the areas where the most species are likely to persist in the landscape (Plumptre *et al.*, 2017; Ayebare *et al.*, 2018; Bagchi *et al.*, 2018). Nevertheless, significant range reduction is predicted for most endemic species due to a loss of suitable climatic conditions and associated habitats, even in mountainous regions (Ayebare *et al.*, 2018).

Using existing SDM outputs, the expected combined species richness of mammals, birds, reptiles, and amphibians is shown in Figure 78. The maps indicate expected species richness under current conditions and under the projections of three different climate models for 2070 (models ac, bc, and cc), showing the range in results depending on which future climate model one uses. Note we use the term "expected," because potential species distributions have been interrupted by anthropogenic land use and other pressures. In particular, the real species richness is likely to be substantially different to the expected species richness in the areas surrounding the wildlife landscape, due to extensive habitat transformation. The model results indicate areas with high current expected species richness are often associated with wetter and more mountainous areas like Rwenzori and Bwindi Impenetrable. Species richness is lower in the central savanna portion of the landscape. Species richness is predicted to remain high in species-rich montane regions in the future though notable declines are expected, particularly under the bc and cc future climate scenarios (Figure 78). Furthermore, these aggregate trends likely mask significant range contractions of threatened and endemic species, the ranges of which have been predicted to decline substantially in other modelling studies of the region (Ayebare *et al.*, 2018). In contrast, species richness across much of the central savanna portion of the landscape is predicted to increase under these climate scenarios, though still falling short of species richness in montane areas. This pattern is also reflected when species richness is broken down into the broad taxonomic groupings (birds, mammals, etc.) of animals (see Appendix 5).

To get a more detailed understanding of climate change impacts, the SDM model predictions of the ranges of key charismatic wildlife species under future climates were also analyzed. The model predictions for the future distribution of mountain gorilla (Figure 79), the most important species in the landscape from a wildlife tourism perspective, are also presented. The expected current distribution encompasses Volcanoes/Mgahinga Mountain Gorilla and Bwindi Impenetrable National Parks where gorilla currently occur, as well as surrounding regions where gorilla may have occurred in the past. Substantial reductions in expected range are predicted under all future climate scenarios, though the severity of these varies depending on the scenario in question. Conditions are most favorable under the cc scenario, where expected range persists in a small portion of Volcanoes and most of Bwindi Impenetrable, but not Mgahinga Mountain Gorilla National Park. Even more worryingly, no suitable range is predicted in any of these parks under the ac and bc scenarios, with expected range limited to the Rwenzori region where gorilla do not currently occur (and an isolated patch to the west of Virunga National Park in the DRC under the bc scenario). The total lack of overlap between current and predicted ranges under some climate scenarios is of great concern for the species, as well as for the lucrative gorilla tourism industry in Rwanda and Uganda.

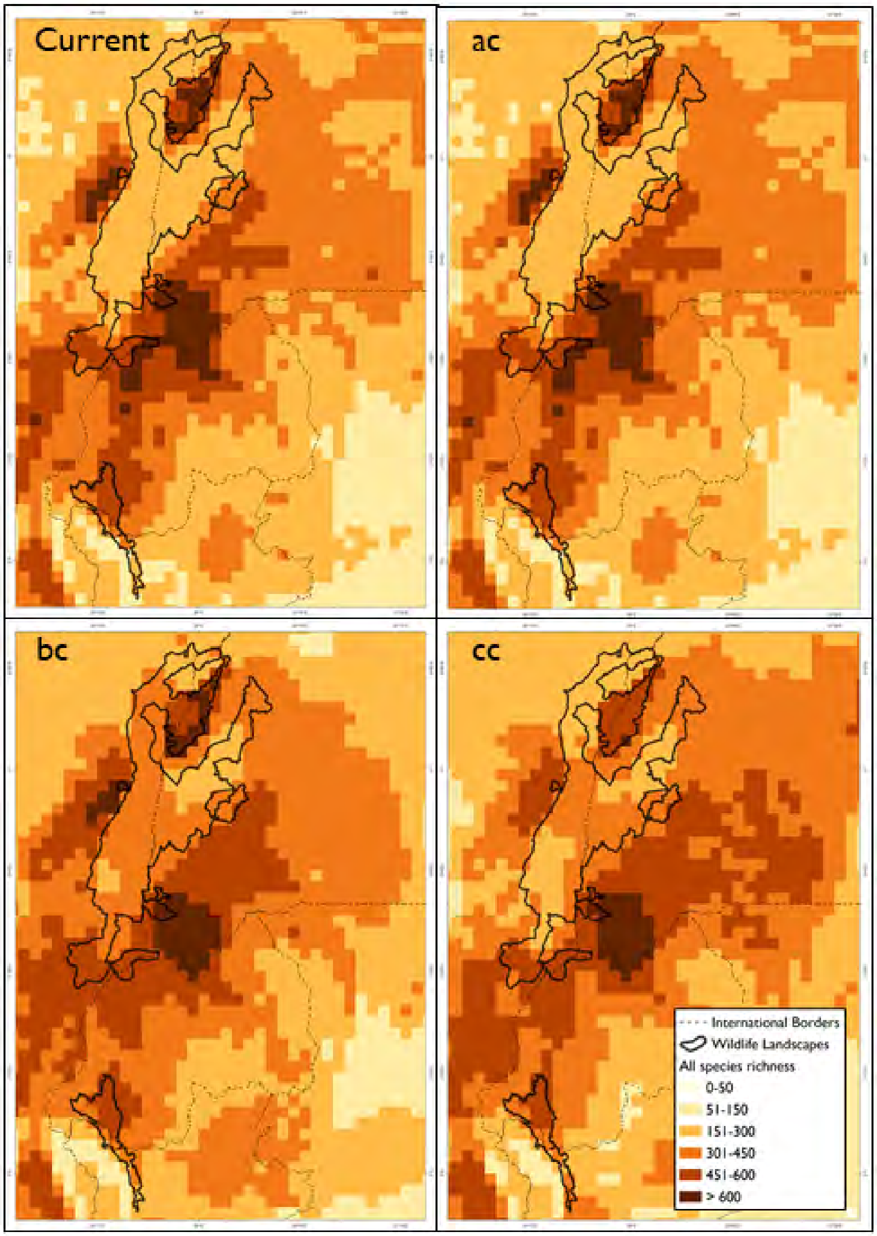


Figure 78. Current geographic variation in expected species richness (amphibians, birds, mammals, and reptiles) for the Albertine Rift Forests landscape, followed by the projected expected species richness pattern under each of the three future climate scenarios used

Source: Based on modelled species distributions from Conservation International

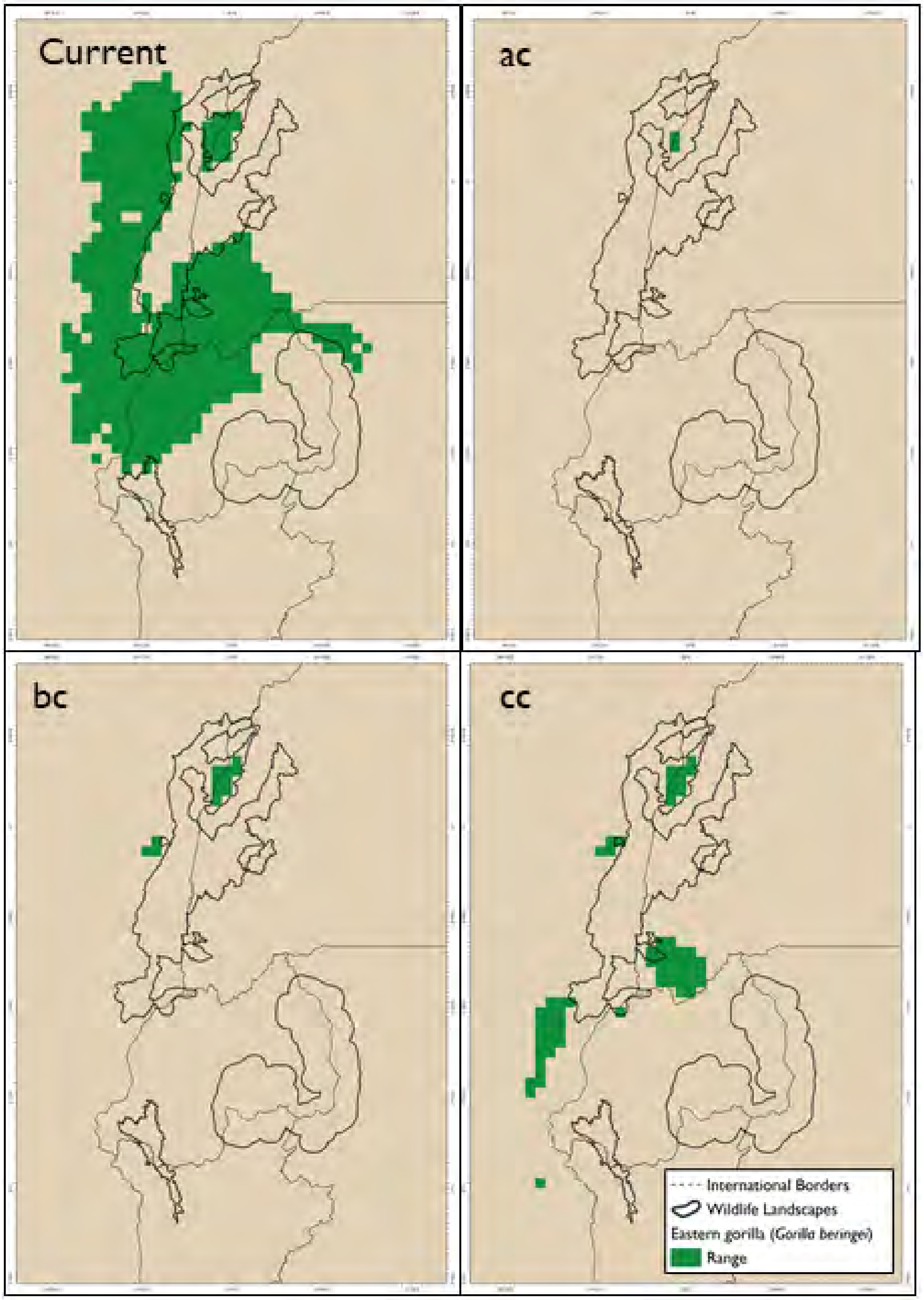


*Figure 79. Current habitat suitability of mountain gorilla (*Gorilla beringei beringei*) for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)*

Source: Based on modelled species distributions from Conservation International

PROJECTED IMPACTS ON SUITABILITY FOR CROPS

To determine the impact of a change in climate on the suitability of crop production, we used FAO's EcoCrop analytical tool (EcoCrop, 2010). Suitability is described in terms of the suitable area for a given crop (*i.e., the* region with a suitability score of greater than 0), as well as the relative suitability score, which ranges from 0 (unsuitable) to 1 (optimum conditions). For most crops, the suitable area is either similar or expands by 2050 with future climate change (Table 62). Additionally, suitability scores increase for many crops under future conditions, with some exceptions. A general upslope migration of suitability occurs for most crop species, suggesting that conversion pressures may increase in higher-lying protected areas that currently have low suitability for cultivation. Changes in suitability are described for individual crop species in Table 62, based on the maps shown in Figure 80.

Table 62. Summary of the expected changes in the suitable area and suitability scores for crops in the Albertine Rift Forests landscape and immediate surrounds, based on the maps shown in Figure 80

CROP	CURRENT SITUATION	IMPACT
Cassava	Most of the landscape suitable, aside from higher-altitude mountainous areas.	Suitable area expands upslope considerably in parts of the landscape (*e.g.,* Nyungwe/Kibira, Bwindi), suitability scores remains generally similar to present across the rest of the suitable area.
Banana	Limited suitability in the landscape, generally more suitable area in the DRC.	Suitable area expands notably in certain regions, such as Nyungwe/Kibira. Suitability scores generally increase across the suitable area.
Beans	Entire landscape suitable aside from highest-lying areas, and suitability scores very high across the entire suitable area range.	Suitable area is predicted to expand into high-altitude areas, while suitability scores are predicted to decrease in the central and northern parts.
Coffee	Suitable area widespread across the landscape, but areas of high suitability associated with moderate elevation mountainous areas (*e.g.,* Nyungwe/Kibira).	Suitable area is predicted to expand upslope in high-altitude areas (*e.g.,* Rwenzori, Volcanoes) and suitability is predicted to increase marginally across the landscape.
Maize	Most of the landscape suitable aside from highest-lying areas and drier areas around Lakes Edward and George.	Suitable area expands slightly upslope, while suitability scores increase substantially across much of the suitable area, with notable increases in suitability in Nyungwe/Kibira and Bwindi.
Plantain	Suitable area limited to central and northern parts of the landscape.	Suitable area expands upslope into higher-lying areas like Nyungwe/Kibira and Bwindi, while suitability increases substantially across much of the suitable area.
Potato	Entire landscape suitable aside from highest-lying areas, and suitability scores very high across the entire suitable area.	Suitable area is predicted to expand into high-altitude areas, while suitability scores are predicted to decrease in the central and northern parts.
Tea	Much of the landscape suitable, aside from drier regions around Lakes Edward and George. Few areas with high suitability.	Suitable area expands upslope, while suitability scores increase significantly. Areas like Nyungwe/Kibira, Bwindi, and Kibale become highly suitable.

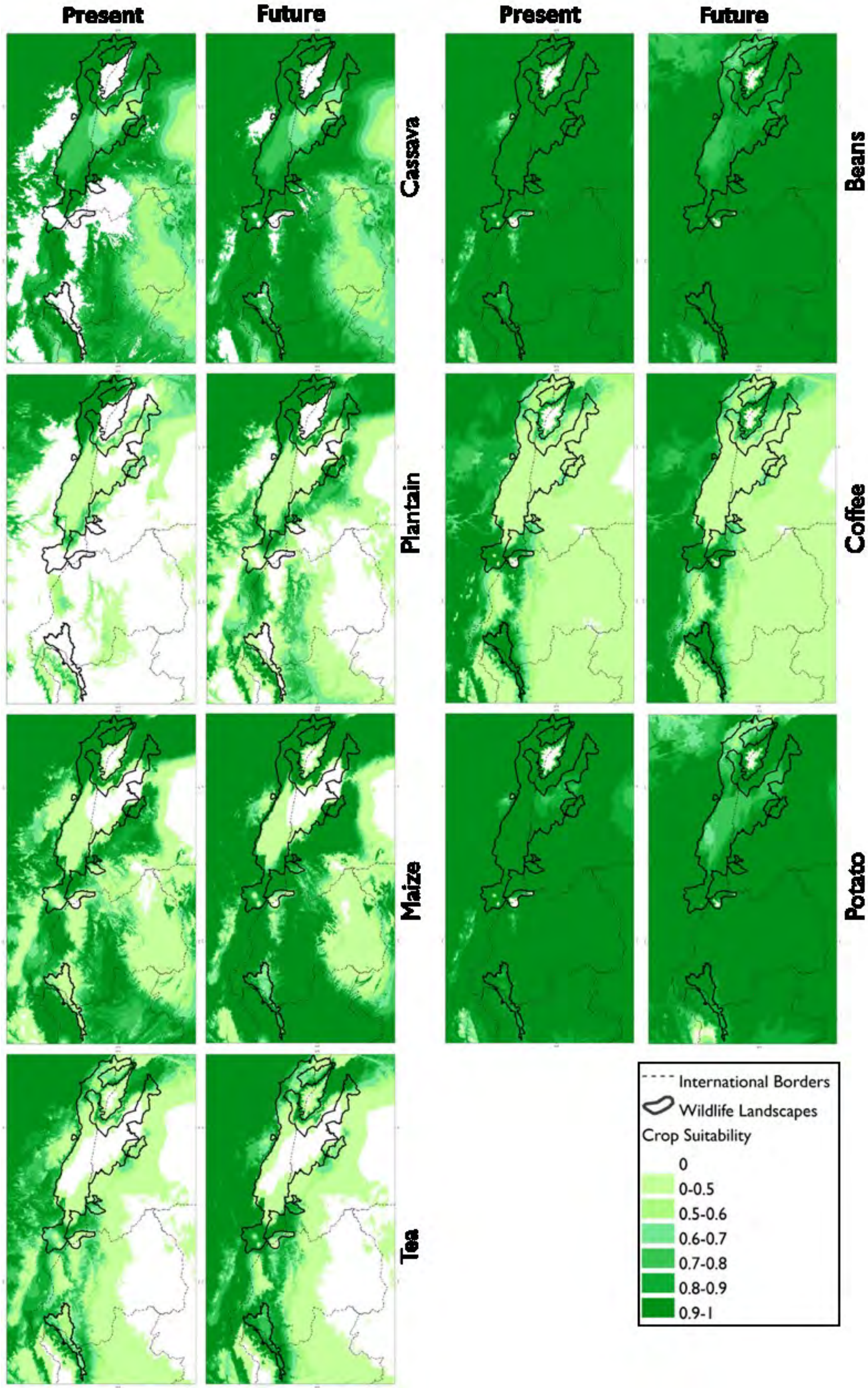


Figure 80. Estimated present and future suitability for some of the key crops grown in and around the Albertine Rift Forests landscape. Model outputs generated using the FAO EcoCrop database and model and climate projections for 2040-60

POTENTIAL IMPLICATIONS OF A BUSINESS-AS-USUAL SCENARIO

This section provides an integrated, qualitative assessment of the impacts of a business-as-usual scenario on wildlife, ecosystem services, and human wellbeing over the period from the baseline (2018) to 2030. The combination of 1) increasing population and demand for land and resources and 2) the impacts of climate change on habitats, species, and agriculture need to be considered. There is a great deal of uncertainty in this. Notwithstanding these caveats, the following impacts could be expected.

Deforestation could accelerate as a result of charcoal production and clearing for cultivation, shrinking available intact wildlife habitat. As population growth continues and the fuelwood supply deficit increases, greater pressure on woody resources in remaining forests can be expected. This will be accentuated by growing charcoal demands from rapidly expanding urban areas, given the higher woody biomass demands for charcoal production. Based on population growth trends, it was estimated that **demand for woody resources could increase by about 75 percent by 2050** in a BAU scenario. This represents severe additional pressure on woody habitats in the landscape. Worsening land scarcity and declining land productivity will drive greater encroachment of cultivation into forests. Governments may resort to partial or full degazettement of protected areas to address land scarcity, as has occurred in the past (*e.g.*, Akagera National Park, Volcanoes National Park). Projection based on deforestation rates between 2001 and 2019 (Figure 81) derived from (Hansen *et al.*, 2013) supports the view that deforestation will increase in the region, due to the increase in deforestation rates since 2014. Based on the current trajectory, we estimated that a further **89,000 ha of forest could be lost by 2050** in a BAU scenario. This amounts to **a loss of 15.5 percent of existing forest cover**. Habitat loss of this magnitude could have serious consequences for the exceptionally high number of IUCN-listed species found in the landscape, many of whose ranges have already been substantially compressed by human activities (Plumptre *et al.*, 2016). Golden monkey, Angolan colobus, and Grauer's swamp-warbler are examples of globally threatened species whose populations are already in decline due to habitat loss and other pressures (IUCN, 2020), while a future upsurge in deforestation rates could jeopardize the current stability of the landscape's iconic and vulnerable gorilla populations.

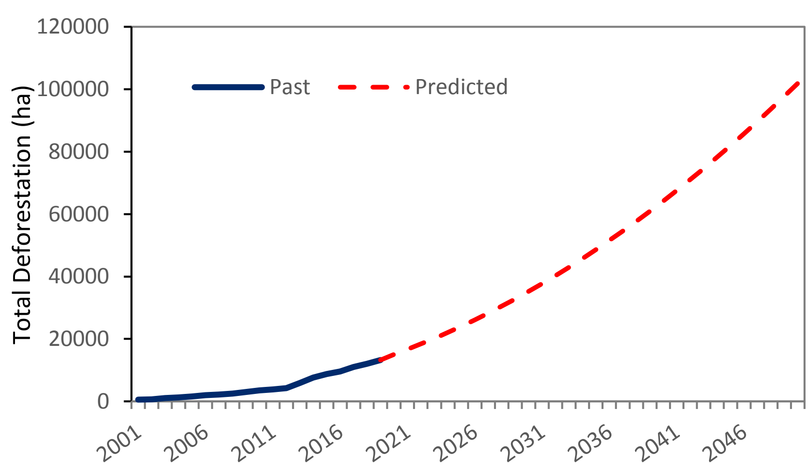


Figure 81. Cumulative deforestation since the year 2000 in the Albertine Rift Forests landscape. The solid blue line shows past deforestation derived from Global Forest Change 2000-2019 data, while the red dotted line shows predicted deforestation under business as usual, based on trends in deforestation between 2001 and 2019

Landscape connectivity could be further compromised, threatening the viability of wildlife populations. Certain protected areas are already totally isolated due to cultivation (*e.g.,* Bwindi Impenetrable, Nyungwe/Kibira). Ongoing habitat conversion in the DRC may erode the critical linkage between Rwanda, Uganda, and the DRC currently provided by the DRC's Virunga National Park. This will lead to further isolation of wildlife populations in the landscape, reducing their ability to adapt to local pressures and potentially increasing genetic erosion.

Key wildlife species could disappear due to shrinking suitable climatic ranges. Modelling studies have revealed that montane habitats and wildlife are particularly vulnerable to climate change. The severe lack of connectivity of some protected areas in the landscape will prevent species from moving in response to climate change, which could result in declining populations and even local extinctions. This includes the highly localized mountain gorilla, whose entire existing range may become unsuitable under some future climate change scenarios.

Tolerance for wildlife could decrease. HWC is likely to worsen as populations living around protected areas become increasingly dense. Where local communities have historically gained some benefits from wildlife tourism, a decline in these could also increase resentment of protected areas and wildlife, especially where human wildlife conflict and opportunity costs of lost cultivation and grazing land are high.

The potential overall effects of the above pressures on wildlife and wildlife habitats on ecosystem services under a BAU scenario can be summarized as follows (see Table 63).

Wildlife tourism revenue has been significantly affected by COVID-19, but may recover well if key attractions can be maintained. Tourism in the landscape has been substantially reduced by the COVID-19 pandemic and associated restrictions on international travel, which has had a significant impact on the financial status of wildlife areas. In Uganda, for example, key informants report that reduced budgetary allocations for conservation have had a severe impact on the operations of national parks. Meanwhile in Rwanda, tourist numbers fell by 75 percent in 2020 and tourism revenue declined by 85 percent. However, it is predicted that tourism could increase and recover if the pandemic eases. Rwanda has already taken measures to adapt to declining tourism revenue, such as entering into partnership with African Parks for management of Nyungwe National Park, which should increase the available budget and effectiveness of conservation in the park. Due to the high value of gorilla tourism in particular and successful conservation of the species, tourism was predicted to fully recover and even increase beyond present-day values by 2050 in Rwanda and Uganda. For **Rwanda, annual tourism value as predicted to increase by US$5.3 million by 2050 (40 percent of current value),** while a **US$4.2 million increase in tourism was predicted for Uganda (11 percent of current value).** This assumes that management of key tourist sites remains effective in the face of increasing encroachment pressures, and that populations of charismatic species can be maintained. Furthermore, it was predicted that tourism growth in these countries would begin to reach a ceiling around 2040, due to ongoing population growth and encroachment pressures on remaining habitat. In addition to habitat loss, poaching, and illegal trade, the threat of climate change to the extremely localized gorilla populations is of much concern. Given the importance of gorilla tourism to the region, the positive tourism growth trajectory will likely change should climate change or other threats start to have a negative impact on the species. In contrast to Rwanda and Uganda, Burundi does not have a well-developed tourist industry. Due to limited tourism products and facilities currently, the potential for increased encroachment into Kibira National Park and the insecurity of this part of the country, it was predicted that tourism in the Burundi portion of the landscape would decline by US$400,000 by 2050 (7 percent of the current value) in a BAU scenario.

Total annual runoff might increase but so will flood risk, while dry season flows are expected to decline. Further deforestation may increase total annual runoff, due to reduced evapotranspiration from forest vegetation. However, loss of the buffering effect of forest vegetation cover would increase flood risks as surface runoff increases. Dry season flows might also be decreased as more rainfall runs off immediately rather than being captured and released more gradually over time. To investigate further, the prediction of deforested area by 2050 in a BAU scenario was extrapolated from deforestation trends between 2001 and 2018. It was assumed that 50 percent of this deforested area would be converted to agriculture (with the remainder staying as more open natural land cover classes, for example, shrubland). If this were to occur, **baseflow was predicted to decline by 3.1 percent by 2050.** If the capacity of the landscape for infiltration and release of flows was reduced in this way, **the cost of reservoir storage to retain this amount of water is estimated to be US$13 million by 2050.**

Soil erosion and sedimentation are expected to increase. Steep slopes and high rainfall mean much of the Albertine Rift region is highly prone to soil erosion. The spread of agriculture to increasingly marginal areas, particularly high gradient, forested areas, will reduce the erosion protection provided by vegetation cover. This will cause a substantial increase in soil erosion. Using the predicted rate of deforestation by 2050 under BAU and again assuming 50 percent of deforested area is converted to cultivation, it was estimated that the **capacity of the landscape to retain sediment and control erosion could decrease by 1.3 percent by 2050**, with an additional 6.5 million tons of sediment entering rivers and waterbodies. If the capacity of the landscape to control soil erosion was reduced in this way, the cost to the region in terms of lost reservoir storage capacity and the greater need for sediment clearance was estimated to be around **US$8 million per year by 2050.**

Nutrient pollution of lakes and watercourses is expected to worsen. Further conversion of forests to agriculture and settlement will reduce the ability of the landscape to retain nutrients in surface runoff. Nutrient inputs across the landscape will also increase as natural habitats are converted to agriculture and fertilizer is applied to these areas. Based on the predicted rate of agricultural expansion in a BAU scenario, it was estimated that **the amount of phosphorus exported from the landscape could increase by a factor of 3.9 by 2050,** representing an additional 179,000 t of phosphorus export over the current landscape. If nutrient export from the landscape increased in this way, **annual water treatment costs to the region would increase by US$338,000 by 2050.**

Deforestation is expected to significantly worsen local and global climate change. Conversion of forest to agriculture and settlement and harvesting of wood for firewood, charcoal, and building materials will lead to a loss of above ground carbon storage. Habitat conversion will also lead to a loss of carbon stored in soils. These losses could make a particularly pronounced contribution to climate change, due to the high amounts of carbon stored in tropical forest vegetation and soils. Based on the predicted rate of deforestation and potential encroachment of built-up land, it was estimated that **carbon storage in the landscape could decline by 7.6 percent (28.7 MtC) by 2050** in a BAU scenario. This would represent an annual cost to the region of US$4.7 million in terms of climate-change-related damages.

Table 63. Estimated changes in the value of ecosystem services and water treatment costs by 2050 caused by land use changes under a BAU scenario for the Albertine Rift Forest landscape. For services with a global value, both total value to the world and value to the East African region only are shown (latter value in parentheses).

ECOSYSTEM SERVICE	CURRENT VALUE (US$)	2050 VALUE (BAU) (US$)	% CHANGE
Nature-based tourism	83.4m (50.3m)	99.1m (59.7m)	+18.7
Biodiversity existence	322.2m (87.6k)	296.7m (80.7k)	-7.9
Flow regulation		-12.7m	-3.1
Erosion control	611.8m	603.7m	-1.3
Carbon storage	42.2b (62.6m)	39.0b (57.9m)	-7.6
Water treatment costs	261.3k	364.3k	+39.4

THE RWERU-MUGESERA-AKAGERA WETLANDS

FEATURES AND LOCAL CONTEXT

WILDLIFE AND WILDLIFE HABITATS

The fourth study region centers on the Rweru-Mugesera-Akagera Wetlands complex, which straddles southeastern Rwanda, northeastern Burundi, and northwestern Tanzania border regions (Figure 82). This wetland complex is made up of the Rweru-Mugesera Wetlands on the Rwanda-Burundi border and the Akagera Wetlands, which straddle Rwanda and Tanzania, forming part of the proposed Kagera Transboundary Conservation Area (TFCA), (IUCN ESARO, 2020).

The Wetlands area is dominated by the Akagera River valley. The flat valley floor reaches 20 km or more in width across much of the region, facilitating extensive lake and wetland systems in the area. The Akagera flows from south to north through the wetlands complex before eventually turning east to drain into Lake Victoria. The wetlands and valley floor are bounded by a series of ridges running roughly parallel to the course of the river.

Large areas of papyrus swamps occur in the wetland complex, as well as several open water lakes. The Rweru-Mugesera Wetland complex is made up of Rweru, Kanzigiri, Cyohoha South, Rwihinda, and Gacamarinda Lakes in Burundi; Rweru, Cyohoha South, Cyohoha North, Gaharwa, Kilimbi, Miravi, Rumira, Kidogo, and Gashanga Lakes in Bugesera District, Rwanda; and Mugesera, Birara, and Sake Lakes in Ngoma District, Rwanda. The Akagera River (also known as the Kagera River) flows out of Lake Rweru along the Rwanda-Burundi border and then north along the Rwanda-Tanzania border through Akagera National Park. (Ndayisaba *et al.*, 2017).

This study focuses on the Rweru-Mugesera-Akagera Wetlands and the ecosystem services and benefits that these habitats provide, but also includes the terrestrial area of the Akagera National Park. The study area for estimation of population, livelihood activities, and the use of harvested resources is depicted in Figure 82 by the dotted grey line (about 20 km around the wetlands). Evergreen bushland is the dominant natural vegetation type in the terrestrial areas around the wetlands, interspersed with grassland; only small patches of forest occur.

Wildlife and natural resources in the study region have come under heavy pressure in recent decades. Rwanda's civil war from 1991-1997 resulted in a lack of law enforcement in Akagera National Park, leading to substantial losses of wildlife (Apio, Plath & Wronski, 2015). The influx of vast numbers of Rwandan refugees also placed high pressure on wildlife and natural resources in northwest Tanzania, including in the Kimisi and Ibanda Game Reserves. Substantial deforestation and poaching took place around refugee camps in Tanzania (Masalu, 2008). Following the cessation of the Rwandan civil war, the return of thousands of refugees, coupled with already very high population densities in that country, resulted in the settling of large numbers of returnees inside Akagera National Park and other protected areas (Apio *et al.*, 2015). These demographic pressures led the Rwandan government to degazette large areas of Akagera National Park and the whole of the adjacent Mutara Game Reserve in 1997, leading to substantial habitat conversion and losses of wildlife. However, following improved protection and fencing in more recent years, the remaining portion of Akagera National Park supports a rich, recovering wildlife population, which includes reintroduced populations of lion and black rhinoceros (Apio *et al.*, 2015; Lindsey *et al.*, 2016; Gross, 2018). Despite degazettement, the park still encompasses a substantial portion of the region's lakes and papyrus swamps, as well as bushland and grassland areas in the western part of the park. Populations of large wildlife species such as elephant and buffalo also remain in Tanzania's Ibanda and Kimisi Game Reserves (Masalu, 2008).

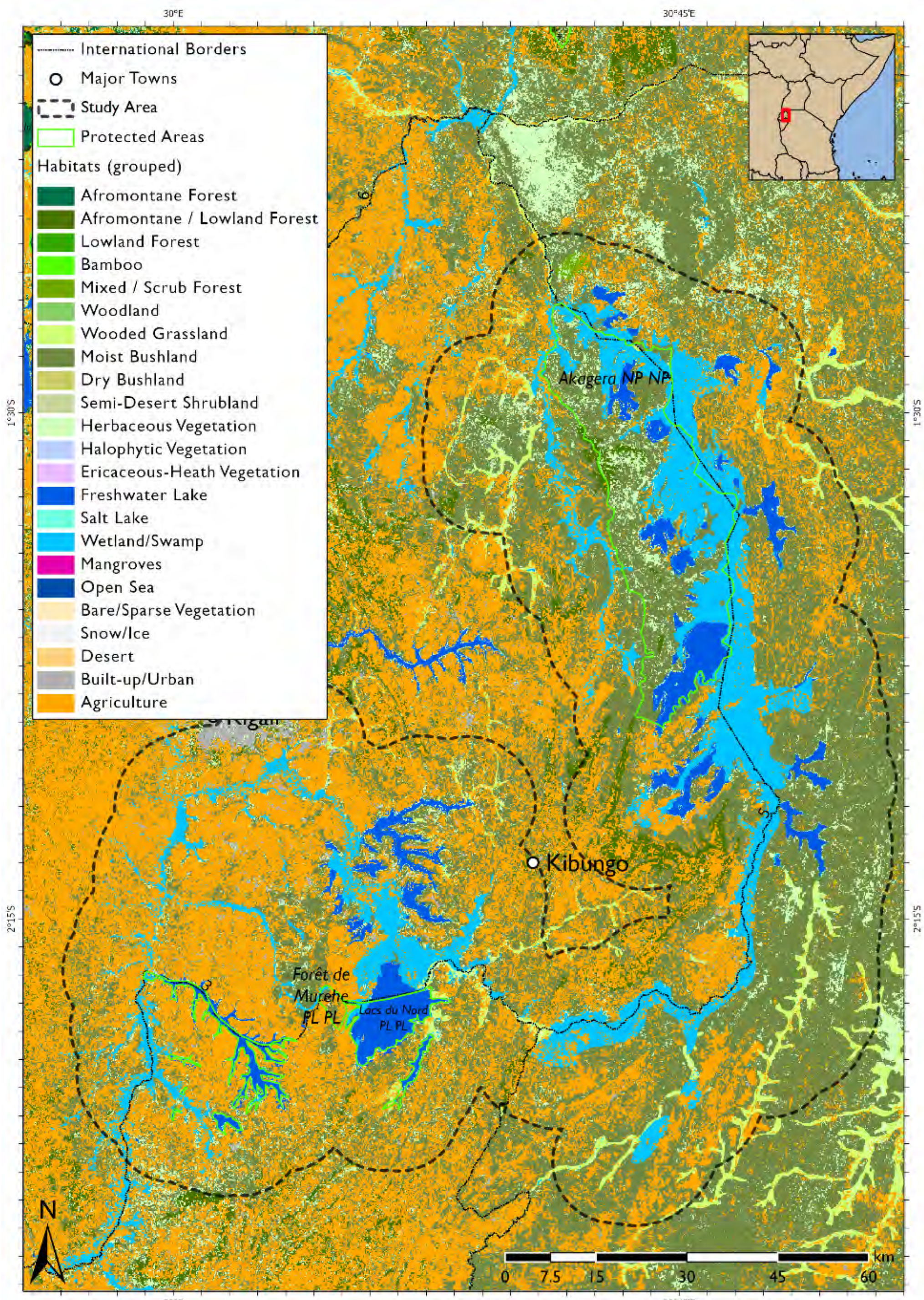


Figure 82. The Rweru-Mugesera-Akagera Wetlands and surrounding land cover

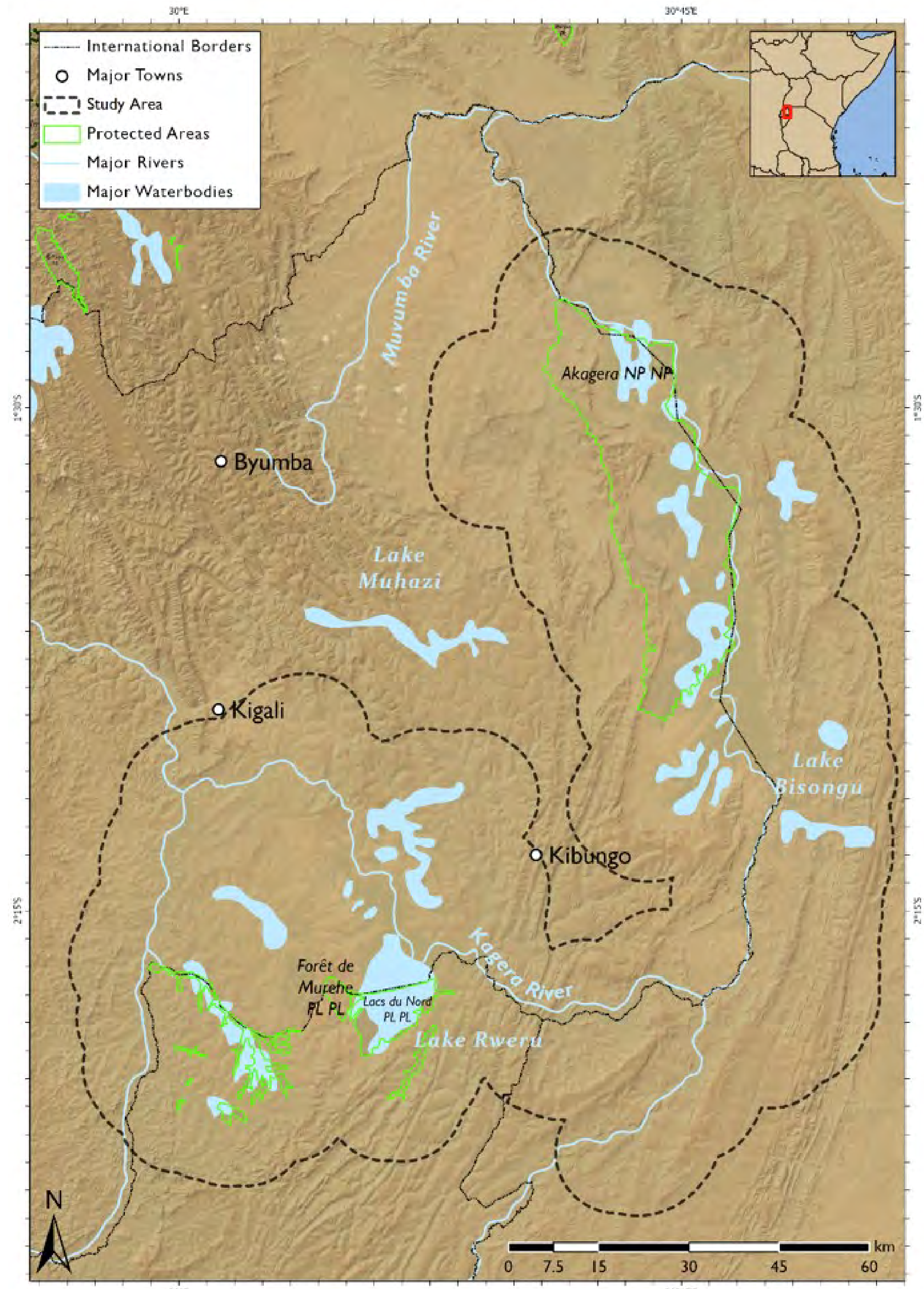


Figure 83. The protected areas of the Rweru-Mugesera-Akagera Wetlands area

Akagera National Park (Figure 83) is tropical Africa's largest protected wetland. It is also the last remaining refuge for savanna-adapted species in Rwanda and the country's only Big Five park (home to lion, leopard, elephant, rhino, and buffalo). Just 20 years ago, the park was on the verge of collapse with most of the wildlife either hunted to local extinction or displaced by cattle. In 2010, African Parks (see https://www.africanparks.org/the-parks/akagera) assumed management of Akagera in partnership with the Rwanda Development Board (RDB). Effective law enforcement and management has significantly reduced poaching, and as a result there has been a complete revival of the park. The park contains incredible biodiversity and rare species, such as the iconic shoebill stork. There are thousands of individual large mammals and more than 480 bird species. The numbers of black rhinoceros and lion are growing. A 120 km solar powered predator-proof fence was constructed along the park boundary, and this has significantly reduced human-wildlife conflict situations. Just downstream of the study area, the recently established Ibanda-Kyerwa National Park in Tanzania (formerly Ibanda Game Reserve) is situated in the western corner of the country bordering Rwanda and Uganda. The rolling plains of *Acacia* savanna cover 20,000 ha and are traversed by seasonal rivers and scattered with permanent lakes and swamps. The park is home to hippo, buffalo, giraffe, leopard, and a number of rare antelope species. It is also teeming with bird life.

The lakes and wetlands of Rweru-Mugesera are surrounded by numerous villages, both in Rwanda and Burundi, where agriculture is the dominant livelihood activity, encroaching on the wetland and lakes' protected buffer zones (Karame, Alvares & Faustin, 2017). The Rweru-Mugesera Wetland complex has diverse bird life as well as crocodiles and monitor lizards, and a number of endemic species of snakes and chameleons. In Rwanda, these wetlands are reportedly the second-richest wetland habitat for mammals outside of national parks (Karame *et al.*, 2017). However, being unprotected, the biodiversity of this wetland complex is threatened by increasing habitat loss (largely through agriculture). A number of bird and mammal species previously recorded in this wetland complex are now rarely seen as a result of the fragmentation and destruction of wetland habitat. The sitatunga (*Tragelaphus spekii*), an antelope associated with swamp and marsh habitat, was commonly seen across the wetland complex only ten years ago. It is now very rarely seen, with decreases in the population attributed to hunting and habitat loss (Karame, Alvares & Faustin, 2017). In parts of the complex, bird species such as the papyrus gonolek (*Laniarius mufumbiri*) and the papyrus yellow warbler (*Calamonastides gracilirotris*), which are restricted to the dense papyrus swamps, have previously been recorded but in more recent surveys were not seen (Karame, Alvares & Faustin, 2017). In Burundi, the lakes of the Rweru-Mugesera Wetland complex, known as "Lacs du Nord," cover 187 km^2 and are a protected landscape according to the World Database on Protected Areas (www.protectedplanet.net, Figure 83). However, there is very little information provided about what this description means.

MANAGEMENT OF PROTECTED AREAS AND NATURAL RESOURCES

Various government institutions are responsible for conservation and environmental management in the area. In Rwanda, the Ministry of Environment (MoE) is responsible for environmental management at the highest level, which includes defining laws and strategies for wetland conservation and management. Protected area management is the responsibility of the RDB. The Rwanda Environmental Management Agency (REMA) is a regulatory body that plays an important role in wetlands management. Its role includes enforcing the laws and regulations set by the MoE. It is also tasked with establishing strategies for issues such as control of invasive species, which are a serious threat to the wetlands complex. Finally, the district governments are in charge of wetland management at the local level.

In Burundi, the Burundian Office for the Protection of the Environment (OBPE) is tasked with environmental protection in general, as well as management of protected areas. It works under the supervision of the Ministry of the Environment, Agriculture and Livestock.

In Tanzania, Tanzania National Parks (TANAPA) is responsible for management of national parks, while the Tanzania Wildlife Management Authority (TAWA) manages other protected areas including game reserves and wildlife management areas (WMAs). These agencies operate as parastatals under the Ministry of Natural Resources and Tourism. TANAPA took over management of Ibanda-Kyerwa from TAWA following its regazettement from a game reserve to a national park. In doing so, it is supported by relevant local district authorities. Despite the transboundary nature of issues like poaching and water pollution, key informants noted that harmonized cross-border conservation policies and strategies are lacking.

Access to and harvesting of resources from wetlands and surrounding terrestrial habitats in the aforementioned protected areas is restricted. Additionally, buffer zones around wetlands and lakes are protected by laws that prohibit or limit cultivation and livestock grazing. In Burundi, for example, cultivation and livestock are not permitted within 50 m buffer zones around some of the lakes and wetlands in the complex. Similarly, Rwanda has created buffer zones around some lakes and wetlands, in which agriculture is not permitted (Karame *et al.*, 2017). However, these measures are not always accepted and understood by communities or enforced by government authorities. As a result, much encroachment of human activities into these buffer areas still occurs, as was confirmed by key informants.

In addition to the various government bodies mandated with management of protected areas and wetlands and the enforcement of relevant laws, NGOs and communities also play an important role in conservation of the wetlands. The key role of African Parks and their successful partnership with the RDB in managing Akagera National Park has already been noted. Elsewhere, NGOs can play an important role in contributing outside of protected areas. For example, Nature Rwanda is an NGO that conducts wetland conservation initiatives in the Mugesera-Rweru section of the wetlands. Private sector actors also play a role in conservation of the wetlands. For example, a bird tour operator noted that they plan to advocate for conservation and alternative livelihood activities among local communities, since human encroachment into wetlands presents a threat to their business activities.

Community involvement in conservation and natural resource management varies across the landscape. Community interactions and involvement with conservation appear to generally be more positive in the Rwandan portion of the landscape, particularly around Akagera where communities benefit from sharing of nature-based tourism revenues. Communities also collaborate with park management here, while community cooperatives benefit from getting contracts to fish in the lakes found within the park (Ndayisaba *et al.*, 2017). This is positively received by community key informants, who reported that these measures have increased their sense of ownership of the park. Efforts by park authorities to reduce HWC through fencing, digging trenches, and collaborating with communities in problem animal management were also appreciated by community key informants here. Due to generally positive perceptions of wildlife, communities around Akagera now reportedly cooperate with parks management in providing information on wildlife crimes. In the Tanzanian portion of the landscape, TAWA reported that they involve communities through providing employment opportunities, particularly for non-technical positions. However, there are no WMAs in this part of Tanzania, meaning the potential for community-based natural resource management is limited. Key informant interviews suggested less interaction occurs between conservation authorities and local communities in Burundi.

Community informants revealed that there are some community-imposed restrictions and protection measures surrounding natural resources, in addition to official government restrictions. For example, an informant from Lake Rweru in Burundi noted that local leaders (chiefs) restrict activities such as tree cutting and illegal fishing. A community informant reported that permission is required to go fishing, and that there are restrictions on the number of boats and nets that can be used. Meanwhile, only collection of dead wood from forests is permitted. People are also prohibited from washing clothes in the lakes in an effort to reduce pollution.

Both Burundi and Rwanda have active community groups that seek to improve environmental protection in the wetlands complex. Several community associations in the Burundian portion of the landscape have been established to protect the environment and enforce laws against hunting and fish poaching. These measures have reportedly increased the availability of fish. In Rwanda, Watersheds Management Committees (*Commités de Gestion des Bassins Versants*) put in place guidelines for the sustainable use of water resources and wetland management. These committees comprise local farmers who work in and around wetlands and rivers. They are appointed by districts and work in conjunction with local agronomists.

The development of payment for ecosystem services (PES) schemes has been proposed as a way of increasing incentives to conserve the wetlands environment. While no PES schemes appear to be in place in the landscape, one key informant felt there was a clear opportunity for their development in the Rwandan portion of the wetlands. This was based on the observation that many businesses benefit from services provided by the wetlands, suggesting an opportunity for these businesses to pay back into the conservation of the ecosystems that provide these services.

PEOPLE AND LIVELIHOODS

Approximately 7.5 million people live within 20 km of the wetlands. This population is almost entirely rural in Burundi and Tanzania (99 percent) but less so in Rwanda (88 percent) due to the position of the capital city Kigali to the northeast of the wetlands (Table 64). The average household size is 4.4, and there are just over 1.3 million rural households, 84 percent of which are within Rwanda. Fertile soils and ready availability of water mean the wetlands remain an attractive site for agriculture and livestock grazing. Indeed, agriculture is the dominant livelihood activity in the study region; pastoral and agropastoral groups are present and most households own some livestock (FEWS NET, 2012). In both Rwanda and Burundi, cattle are owned by the wealthier households and goats and poultry by the poorer households, and in Tanzania cattle, sheep, and goats are owned by wealthier households with poorer households usually owning just goats. This southeastern region of Rwanda, northeastern region of Burundi, and northwestern region of Tanzania has become known for the large-scale production of bananas, which provides a source of food and income for most households. Market access is good throughout this region, and other commonly grown crops include beans, maize, cassava, peas, and in some areas, coffee. The large proportion of the Akagera Wetland complex contained within a protected area means human activities are limited across much of this system, particularly in Rwanda. Nevertheless, park management has put in place coordinated frameworks to permit and regulate fishing in some of the lakes (Ndayisaba *et al.*, 2017). Fishing on the lakes in Burundi is an important activity, with total catch from these lakes in 2018 totaling 3,600 tons (Ministry of Environment Agriculture and

Livestock, 2020). However, in general, fish stocks are declining in the region (FEWS NET, 2012; REMA, 2019).

Table 64. Population statistics for the Rweru-Mugesera-Akagera Wetlands study region

COUNTRY	TOTAL POPULATION	NUMBER OF RURAL HOUSEHOLDS	AVERAGE HOUSEHOLD SIZE	% RURAL
Rwanda	5,391,081	1,128,956	4.2	88
Tanzania	1,010,984	212,913	4.7	99
Burundi	1,158,191	261,872	4.3	99
Total for study region	**7,560,255**	**1,603,741**	**4.4**	**95**

ECOSYSTEM SERVICES

NATURE-BASED TOURISM

The Rweru-Mugesera-Akagera Wetlands include some of the largest protected wetlands in Africa and are home to a wide array of bird and animal life. However, habitat degradation and poor accessibility are barriers to tourism development across much of the complex. Large wildlife populations in the area are thus largely limited to Rwanda's Akagera National Park and Ibanda-Kyerwa National Park in Tanzania. In addition to wildlife-viewing opportunities, birdwatching is a tourism drawcard. As well as large populations of waterbirds, the vast papyrus swamps harbor unique habitat specialist bird species such as the Papyrus gonolek, Papyrus canary, and Papyrus yellow warbler.

Rwanda's Akagera National Park is a tourism success story. In just under ten years, the park has been rehabilitated, wildlife numbers have increased, and tourism has thrived. In 2009, the Rwanda Development Board and African Parks signed a joint agreement establishing a management company for the park. An electric fence was erected to keep poachers out and to address human-wildlife conflict. A team of 80 rangers and a canine unit protect the wildlife within the park, and rangers are also involved in community outreach to develop trust and promote conservation. The park has also been restocked with lion and black rhino, making it the only Big Five park in Rwanda. An aerial wildlife census count in 2015 revealed that wildlife populations for many species have increased, most notably buffalo, waterbuck, zebra, topi, and warthog (Akagera Management Company, 2015). The population of roan antelope, considered to be vulnerable to extirpation in this area, was found to be recovering well, and numbers of eland had also increased (Akagera Management Company, 2015). Investment in the tourism facilities in the park has included the construction of two upmarket tented camps. The park is also only a two-and-a-half-hour drive from Kigali, allowing easy access for tourists and providing the opportunity for daytrips.

While gorillas undoubtedly continue to be the biggest drawcard for tourists to Rwanda, the rehabilitation of Akagera has diversified the industry and provided an opportunity for tourists to see large game, including the Big Five. This has not gone unnoticed by tour operators who are increasingly adding Akagera to their itineraries. Indeed, since 2010, tourism revenue has increased by 1,150 percent and the park is now 90 percent self-financing (African Parks). Visitor numbers to Akagera National Park have been increasing steadily since 2010 (Figure 84). In 2017, a total of 44,054 people visited the park, a 5 percent increase on the year before. It is the most visited park in Rwanda, accounting for 47 percent of all park visitors in 2017. It has thus been a catalyst for the development of nature-based businesses in the area. Tourism is lower in the Rwandan portion of the wetlands away from Akagera National Park, as was confirmed by key informants. Since the Bugasera portion of the wetlands has limited wildlife viewing opportunities, papyrus-restricted bird species are one of the main attractions, as the area is reportedly one of the best sites in the country to see these.

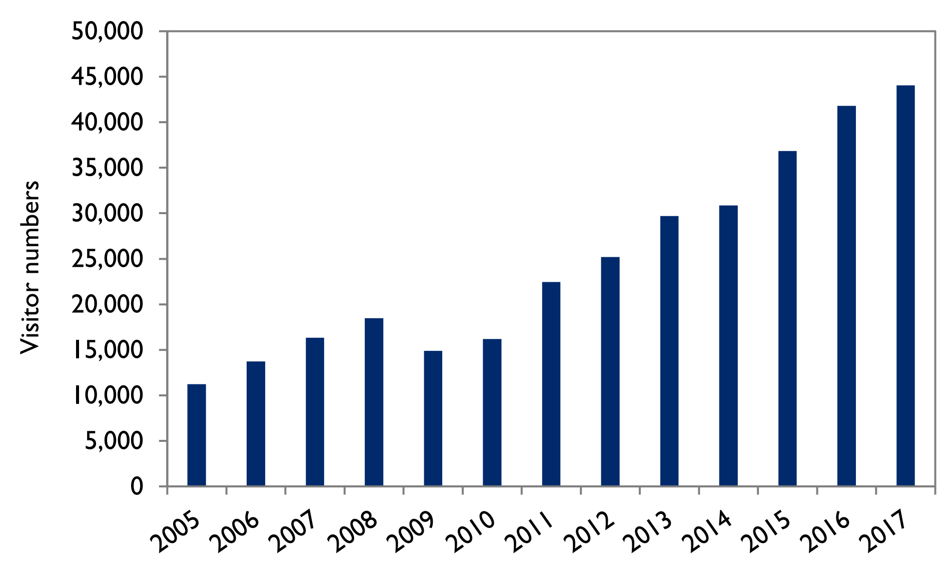


Figure 84. Visitor numbers to Akagera National Park in Rwanda from 2005-2017

Source: NISR, 2019

The protected areas over the border in Tanzania and Burundi are relatively unknown to tourists. In Tanzania, the recent gazettement of Ibanda-Kyerwa National Park (formerly Ibanda Game Reserve) forms part of the Resilient Natural Resource Management for Tourism and Growth (Regrow) Project, with the objective being to strengthen management of protected areas and promote tourism in this part of Tanzania. The area was previously used primarily for sport hunting, but this no longer takes place now that it has been designated as a national park. Following its reclassification as a national park, TANAPA has taken over management of the area from TAWA. A key informant confirmed that visitor numbers are low as the park is still relatively unknown. Notably, some key informants were not positive around the regazettement of Ibanda as a national park, as it resulted in the withdrawal of the safari hunting company that had invested in the area. On the other hand, key informants reported that poaching has declined since the park was upgraded from game reserve status. The park covers an area of 200 km^2 and is mostly covered by grassland and moist bushland. Wildlife include buffalo, hippo, leopard, and a number of antelope species. Facilities in the park appear to be limited, with basic open camping sites on offer. Activities include game drives, birdwatching, and walking safaris. There is little information on the numbers of visitors to this newly established park. However, we assume based on Figure 85 that tourist numbers are still very low. Despite the withdrawal of the safari hunting operator, key informants report that there are multiple joint ecotourism ventures in the area between private companies and community groups, as well as a number of tourist lodges.

In Burundi, the protected area of Lacs du Nord appears to cover only the large lake systems and their buffer areas, and there is very little information on the wildlife and tourism in this area of the country. Key informants noted that the large hippo population in Lake Rweru, which straddles across the Burundi-Rwanda border, is a potential drawcard for tourists, but is hampered by a lack of accessibility. They also lamented the total lack of tourism facilities at the lake. One key informant also noted that the number of tourists coming to visit from Rwanda had declined substantially in recent years due to political tension between Burundi and Rwanda. Infrastructure such as roads and water supply were also reported to be very poor, presenting a significant barrier to tourism. The low contribution of tourism was confirmed by key informants, who reported that the Lacs du Nord landscape generated just 80,000 BIF in 2019, or only 0.13 percent of the total tourism revenue generated by protected areas in Burundi as a whole.

There are some revenue-sharing mechanisms in place to increase the flow of nature-based tourism benefits to local communities. In Rwanda, the RDB allocates 10 percent of nature-based tourism revenues from national parks to local communities. Beneficiaries are reportedly selected through collaborative agreements between the government and local communities, with money going to community cooperatives. Membership of a cooperative is thus reportedly a prerequisite for benefitting from the revenue sharing arrangements. However, all community members benefit from other measures such as infrastructure development projects funded by the park. According to key community informants, revenue-sharing has helped to improve attitudes toward wildlife among communities living alongside Akagera National Park. Some community members also benefit from selling handicrafts to tourists. Unfortunately, key informants reported that these revenues have declined substantially with the COVID-19 pandemic due to the severe reduction in tourism. Key informants confirmed that benefits from nature-based tourism are currently very low. In addition to revenue sharing, community key informants around Akagera reported that they benefit from employment as guides and rangers, and as labor for maintaining park infrastructure. Revenue sharing from nature-based tourism is more limited in the Rwandan portion of the wetlands away from Akagera due to the lower visitor numbers in the rest of the wetlands complex. Community informants from Burundi lamented the lack of tourist facilities around their portion of the wetlands, feeling this results in missed opportunities for employment. They thus expressed hope that the state or private sector might invest in the development of tourist facilities in the area. They also noted that there are no mechanisms to allow them to benefit from the tourists who do visit the area, prompting them to suggest visitors should pay a fee when they come to the lakes. Tanzania also does not have a formalized policy for sharing of revenues from national parks with local communities. However, community informants living around Ibanda-Kyerwa National Park in Tanzania reported that TANAPA allocates a portion of its revenues to fund community development projects in the area. Key informants also reported that 25 percent of tourism revenue is meant to go to the responsible district, which then ostensibly trickles down to villages living around national parks in that district.

Holiday tourists make up 7 percent tourists to Rwanda and Burundi and 64 percent to Tanzania, respectively (Table 65). The total attraction-based tourism value in 2018 for Rwanda was estimated to be US$124 million, US$1.7 billion for Tanzania, and US$15.4 million for Burundi.

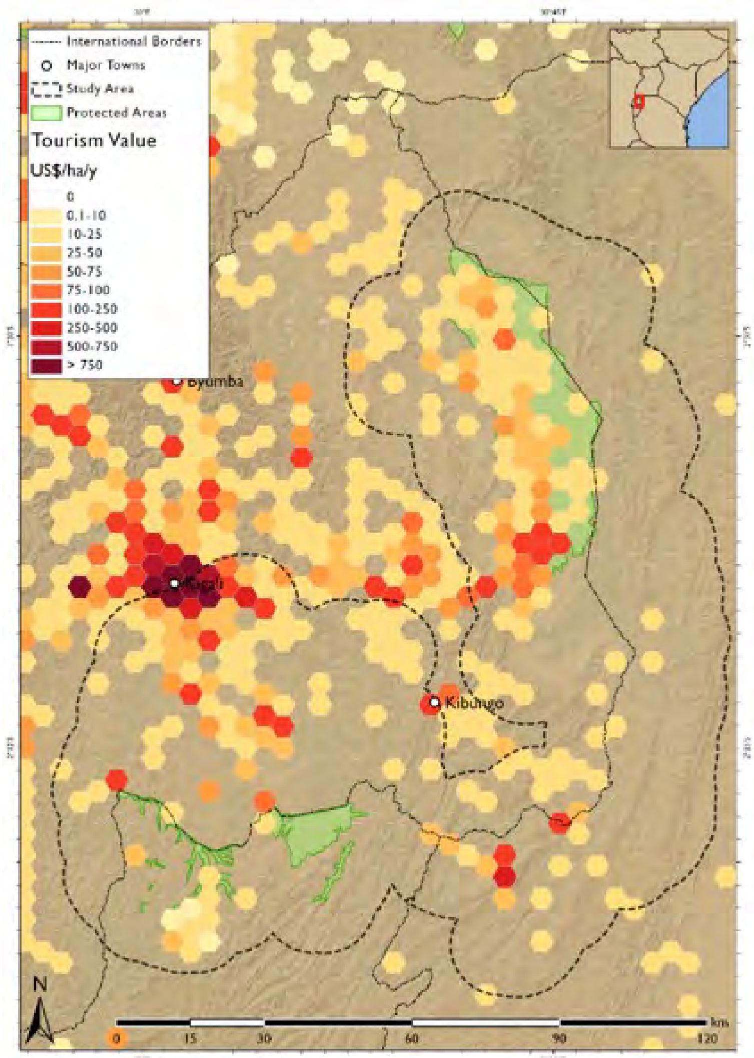


Figure 85. Tourism value (US$/ha/y) for 2018 across the Rweru-Mugesera-Akagera Wetlands wildlife landscape, based on the distribution of geo-referenced photographs uploaded to Flickr

Table 65. Typology of tourists to Rwanda, Tanzania, and Burundi in 2018

PURPOSE OF VISIT	BURUNDI (%)	RWANDA (%)	TANZANIA (%)
Holiday	7	7	64
VFR	30	30	16
Business	29	29	9
Other	33	33	11

Note that this data was not available for Burundi so it was assumed that these estimates would be the same as neighboring Rwanda.

Based on the spatial distribution of tourism activity, the contribution of the Rweru-Mugesera-Akagera Wetlands to tourism value was estimated to be US$5.3 million in 2018: US$4.5 million in Rwanda, US$0.7 million in Tanzania, and US$0.08 million in Burundi. In Rwanda, this represents 3.6 percent of total attraction-based spending in the country, and in Tanzania and Burundi it represents less than 1 percent of the total national attraction-based spending (Table 66). The tourism value of Akagera National Park was estimated to be US$2.6 million per year (US$26/ha/y), accounting for 50 percent of the total tourism value across the Rweru-Mugesera-Akagera Wetlands. The tourism value of the Lacs du Nord protected area in Burundi was estimated to be US$0.03 million per year (US$1.40/ha/y), accounting for just 0.5 percent of the total tourism value across this study region. The spatial distribution of the value is shown in Table 66. Nature-based tourism also generates an estimated $7 million in net benefits (consumer surplus) to overseas visitors.

Table 66. Estimated total attraction-based tourism value for Rwanda, Tanzania, and Burundi in 2018 and estimated nature-based tourism value of the Rweru-Mugesera-Akagera Wetlands

COUNTRY	TOURISM DIRECT CONTRIBUTION TO GDP	LEISURE SPENDING AS A PROPORTION OF TOTAL SPENDING (%)	TOTAL ATTRACTION-BASED TOURISM VALUE PER COUNTRY	TOURISM VALUE OF WETLAND LANDSCAPE	% OF NATIONAL VALUE
Burundi	$49 m	44	$15 m	$0.08 m	0.5
Rwanda	$416 m	48	$124 m	$4.53 m	3.6
Tanzania	$2,762 m	84	$1,712 m	$0.66 m	0.04

All values in 2018 US$ millions

WATER QUALITY AMELIORATION

Much of the catchment of the Rweru-Mugesera-Akagera Wetland system is heavily developed, with a large proportion under cultivation. In addition, Rwanda's capital city of Kigali lies just upstream of the wetland system. Based on the outputs of the InVEST model, the catchment areas of the wetlands generate elevated nutrient loads on the order of 2,700 tons of phosphorus and 7,000 tons of nitrogen, respectively, per year. These nutrient loads are transported downstream, through the wetland system, *en route* to Lake Victoria. Of particular concern is the phosphorus, which contributes to the eutrophication of the lake.

Based on recent studies of other papyrus-dominated wetlands around Lake Victoria, which suggest a removal rate of 77 kg P per ha (Kansiime & Nalubega, 1999), and taking into account the condition of different parts of the study area, the papyrus swamps of the wetland system have the capacity to remove as much as 8,700 tons of phosphorous per year, preventing it from reaching Lake Victoria. Much of this would be due to the settling out of sediments to which phosphorous is attached and transported. If the papyrus were harvested, this would remove a further 4.47 tons of nitrogen and 2.04 tons of phosphorous per ha (assuming it was fully removed; Kansiime *et al.,* 2007).

This suggests that the wetland system plays a very significant role in preventing excess nutrient loads from reaching Lake Victoria. The replacement cost of the service as currently utilized would be on the order of US$726,000, or up to US$1.06 million if the potential were fully utilized.

CARBON STORAGE

Natural ecosystems make a significant contribution to global climate regulation through the sequestration and storage of carbon. About half of all vegetative biomass comprises carbon. In addition to accumulation in woody biomass, carbon accumulates in soils and peat as a result of the collection of leaf litter and partially decayed biomass. Degradation of vegetated habitats releases carbon and contributes to global climate change with impacts on biodiversity, water supply, droughts and floods, agriculture, energy production, and human health, whereas restoration or protection of these habitats mitigates or avoids these damages, respectively. The conservation and restoration of natural systems thus helps to reduce the rate at which greenhouse gases collect in the atmosphere and the consequent impacts of climate change.

Freshwater wetlands are some of the most productive ecosystems on earth and, when compared to forests, grassland, and shrublands, have a higher soil carbon density (Kayranli *et al.*, 2010; Mitsch *et al.*, 2013; Were *et al.*, 2019). While wetlands cover only about 5-8 percent of the terrestrial land surface, they are estimated to store up to 30 percent of the global soil carbon stock (Mitsch *et al.*, 2013). Their ability to sequester and store large amounts of carbon is due to their anoxic wet conditions, saturated soils, and high rates of primary production in wetland plants when compared to terrestrial plants (Were *et al.*, 2019). However, the degradation and encroachment of wetland habitats contributes to the release of carbon and methane into the atmosphere. The protection of wetland habitats is thus considered critically important for mitigating climate change.

Based on global datasets derived from satellite data (see FAO & ITPS, 2018; Spawn & Gibbs, 2020), it was estimated that approximately 92 million tons of carbon are stored within the wetland vegetation and soils of the Rweru-Mugesera-Akagera Wetlands (Table 67, Figure 86). Just under two-thirds of the total carbon stocks are situated within the wetland habitats of Rwanda, and only about 3 percent is within the wetland habitats of Burundi. Figure 86 clearly shows the high storage capacity of wetland habitat when compared to the surrounding landscape. Densities are highest (>1,000 tons per hectare) in parts of the Akagera National Park and south of the park in Tanzania, as well as the floodplain areas of the Akagera River and the wetlands just north of Lake Rweru in Rwanda. The mean carbon stored per hectare ranges from 290 tons per hectare in Burundi to more than 630 tons per hectare in Tanzania, with maximum carbon storage reaching more than 1,500 tons per hectare in parts of Rwanda and Tanzania. Our finding of the carbon stocks per hectare were similar to those of REMA (2019).

Table 67. The total amount of carbon stored within the Rweru-Mugesera-Akagera Wetlands and summary statistics (tons carbon per hectare) per country

COUNTRY	TOTAL STOCK OF CARBON (TONS)	MEAN T/HA	MIN T/HA	MAX T/HA
Burundi	3,038,340	290.42	41.67	1,352.10
Rwanda	56,061,085	488.61	36.16	1,590.14
Tanzania	33,396,295	630.87	30.57	1,554.35

It has been estimated that a ton of carbon released into the atmosphere will cause global damages on the order of US$417 (net present value over 80 years), of which Rwanda's share is US$0.16 per ton, Burundi's share is US$0.04 per ton, and Tanzania's share is US$1.04 per ton (Ricke *et al.*, 2018). The total global damage costs avoided by retaining the total stock of biomass carbon is US$9.135 billion per year (Table 68). The avoided damage cost to Rwanda is estimated to be US$2.2 million per year, Burundi is US$0.02 million, and Tanzania just under US$8.2 million per year.

Table 68. The total global damage costs avoided by retaining the total stock of biomass carbon and the avoided damage cost to each country (US$ million/y)

	BURUNDI	RWANDA	TANZANIA	REST OF THE WORLD
Carbon storage value (damage costs avoided, US$ million/y)	0.01	1.81	6.34	7,328

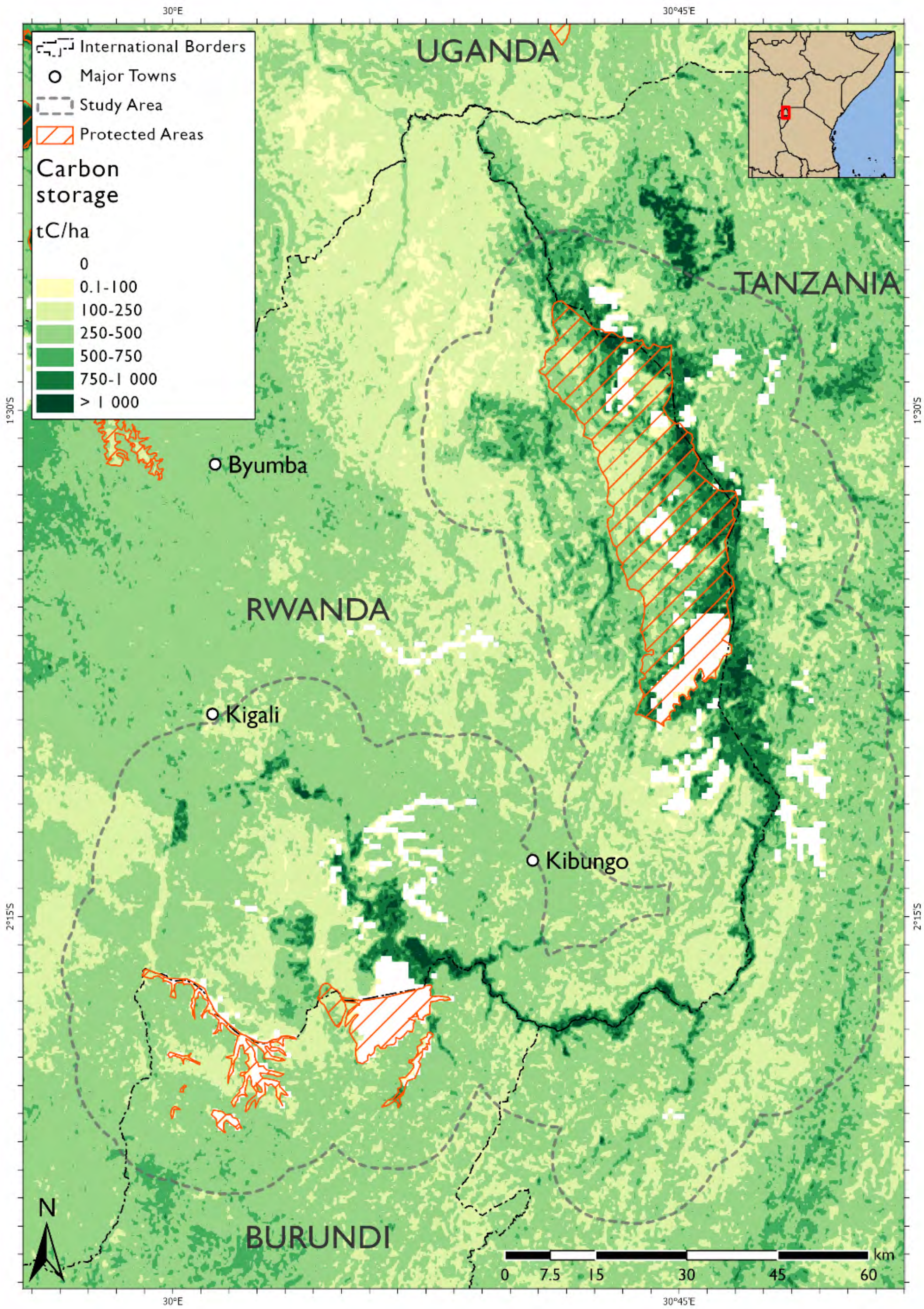


Figure 86. Total carbon storage (tons/ha) across the Rweru-Mugesera-Akagera Wetlands and surrounding areas

HARVESTED RESOURCES

In addition to their conservation value, natural resources play an important role in supporting the livelihoods of people. Wild plant and animal resources are harvested for food, medicine, energy, and raw materials, particularly where there are limited economic opportunities. The capacity of the landscape to supply different types of wild resources is related to vegetation type and condition, availability of water, and other factors. However, a number of other factors determine their use and value, and these vary in space and time. The accessibility of wild resources is determined by regulations such as land tenure and harvesting rights, social norms and informal agreements, geographic features such as topography and rivers, and human-made features such as roads. The demand for wild resources is influenced by the socio-economic circumstances of households and the prices of alternatives.

Wildlife habitats usually require full, no-take protection, not only because of the risk associated with overharvesting that changes the nature and functioning of wildlife habitats, but also because of the disturbance that it can cause, especially affecting the shier and more vulnerable wildlife species. The people that live around these wildlife habitats are largely dependent on wild resources, particularly during times of economic stress. Examples of such stressors include crop disease, drought, or floods, which are likely to only worsen with climate change, and international pandemics such as COVID-19, which has had far-reaching consequences. It is during these times that people fall back on nature to fill livelihood needs. However, this is a potentially vicious cycle of unsustainability as more people rely on nature for food and raw materials and stocks become depleted. The stocks of resources protected within parks and reserves help to maintain the stocks utilized outside of these wildlife habitats. The more resources harvested unsustainably, the fewer there will be available in the future and the less we can rely on nature to fill this need. As resource stocks outside of protected areas become degraded, there will be a higher demand for the resources on the edge of protected areas as well as on the inside.

THE DEMAND FOR NATURAL RESOURCES

The people living around the Rweru-Mugesera-Akagera Wetlands in Rwanda, Tanzania, and Burundi are dependent on agriculture as their main source of income (Figure 87). The main crops grown for sale across the wider region are bananas, beans, and coffee. Livestock sales supplement cash earned from selling crops. Rainfall in this region is relatively reliable, and most of the crops that are produced can sustain households for most of the year (FEWS NET, 2008). The mosaic of wetland lakes and swamps provide important natural resources, with many of the local households dependent on wetland resources in some way to meet their basic subsistence needs. These include the use of palms, reeds, and sedges for mats, baskets, and other handicrafts; grass for thatching and livestock fodder; and the collection of wild plants for foods and medicines. Small-scale fishing with nets is reportedly undertaken by only a small percentage of households in Rwanda and catches are relatively low (NISR, 2010), whereas in Burundi, fishing is extensive. Effective protection of the Akagera National Park has reduced and limited hunting in the Rwandan parts of the study region. In this section, we focus exclusively on wetland resources that are harvested by households.

Figure 87. Agriculture is the main livelihood activity, and bananas are the most important cash crop. Houses are typically made from mud bricks with corrugated iron roofs.

Credit: IITA (top left), Gwendolyn Stansbury (top right); Neil Palmer (bottom right); Gwendolyn Stansbury (bottom left)

The rivers and large swamp areas in this study region provide wetland grasses, reeds (*Phragmites* sp.), and papyrus (*Cyperus papyrus*), which are harvested for various purposes (Table 69, Figure 88). Reeds and papyrus are important resources in many wetland areas in Africa, being commonly used to produce mats and various other household products such as chicken coops, food storage containers, and crafts (Ngombe, n.d.; Turpie, 2000; Abila, 2002; Dixon & Wood, 2003; Mmopelwa, Blignaut & Hassan, 2009; Kakuru, Turyahabwe & Mugisha, 2013). They are also used in house construction, particularly for roofing materials and sometimes doors and windows (Ngombe, n.d.; Thenya & Mwaniki, 2017).

Grasses are commonly harvested for thatching but appear to not be widely used in the study area (Table 69), with less than 1 percent of households having thatched roofs across the Rwandan side of the study region according to that country's household census data (NISR, 2012). The use of thatching grass is slightly higher in Tanzania, where 7 percent of households are recorded to have thatched roofs (NBS Tanzania, 2015). A field study further south of the wetland region in the Nkasi District found that 4.9 percent of households harvested thatching grass (Rwamahe, 2008). In Tanzania's Kirua Swamp area, grasses were reportedly harvested by just over 12 percent of households from the Pangani Floodplain, most of which was used for fencing and thatching (Turpie *et al.*, 2005).

Field data on the use of reeds and sedges in this region is limited. In the Akagera Wetlands, it was reported that fewer than 10 percent of households harvested papyrus, with an average collection of just

30 kg over a four-month period (REMA, 2019). Outside of the study area, a field study from the Yala Swamp in Kenya found that 86 percent of households harvested non-woody wetland resources for building materials, and that papyrus and grass were the main thatching materials used, with 30 papyrus bundles being used to thatch an average-sized house (Abila, 2002). Other field studies from the Yala Swamp put the proportion of households harvesting reeds and sedges at 49 percent (Thenya & Mwaniki, 2017) and 98 percent (Ngombe, n.d.). In another study from further afield, households surrounding the Illubabor Wetlands in western Ethiopia rely heavily on sedges for thatching of houses with 85 percent of households harvesting papyrus from the wetland (Dixon & Wood, 2003). Approximately 9 percent of households surrounding the Kirua Swamp were found to harvest on average 21 bundles of papyrus per year where it was used for roofing material, and just over 7 percent of households harvested on average 77 bundles of reeds per year, which were used for temporary structures and for making doors (Turpie *et al.*, 2005).

Figure 88. The Akagera Wetlands are covered by vast papyrus beds

Credit: John and Melanie Kotsopoulos

Table 69. Proportion of rural households harvesting reeds, sedges, and wetland grasses within each country in the Rweru-Mugesera-Akagera Wetlands and the estimated demand per average household per year

COUNTRY	REEDS, SEDGES, AND WETLAND GRASSES	
	% RURAL HH	KG/HH/Y
Rwanda	65	331
Tanzania	65	331
Burundi	65	331

The collection of wild plants for foods and medicines is a common activity for households in this region (Karame *et al.,* 2017; REMA, 2019; Table 70). However, studies providing quantitative estimates of demand are scarce. Similar to the other study regions, it is assumed that the collection of wild fruits and edible plant parts is mostly carried out by the poorest households for subsistence purposes, especially in years when crops fail. The reliance on agriculture is noticeable across this region. Crops are the main source of food and income; bananas, beans, and cassava are grown for consumption and sale at market (FEWS NET, 2008, 2012). Market access is good across the study region as road networks link the zone to larger trading centers (FEWS NET, 2008). It is usually the case that the number of households harvesting wild plant foods is low in areas where there is good access to markets or shops. Indeed, Turpie *et al.* (2005) found that in the Kirua Swamp area that food plants were collected by 9 percent of households and wild fruits by just 5 percent of households.

Wetland habitats are known to provide a wide variety of plant material used for medicinal purposes. Around the Rweru-Mugesera Wetlands, a total of 16 medicinal plant species were recorded as being harvested by households (Karame *et al.*, 2017); at the Yala Swamp in Kenya a total of 23 plant species were recorded as being harvested by households for medicinal purposes (Ngombe, n.d.). The dependence on traditional medicines is likely to be relatively high in this region where a large proportion of the study region is rural and access to Western medicines is limited. Approximately 30 percent of households surrounding the Kirua Swamp harvested medicinal plants from the wetland and floodplain (Turpie *et al.*, 2005). In the Yala Swamp region, this estimate was as high as 45 percent (Ngombe, n.d.).

Table 70. Proportion of rural households harvesting wild plants foods and medicines within each country in the Rweru-Mugesera-Akagera Wetlands and the estimated demand per average household per year

COUNTRY	WILD PLANT FOODS		MEDICINES	
	% RURAL HH	KG/HH/Y	% RURAL HH	KG/HH/Y
Rwanda	9	7.6	38	4.0
Tanzania	9	7.6	38	4.0
Burundi	9	7.6	38	4.0

Fishing is an important activity in this study region. The Rwandan National Agricultural Survey of 2008 covered all rural areas of the country and provided detailed information pertaining to the number of households engaged in fishing and associated production (Table 71), summarized at the province and district level. Information on fishing in Burundi was extracted from the recent Agricultural Statistics Report (Ministry of Environment Agriculture and Livestock, 2020). Information from Tanzania was limited.

Fishing is practiced using nets in the lakes that are scattered across the study region as well as the smaller rivers (FEWS NET, 2012). The number of fisher households and annual fish production was surprisingly low for the region when compared to other studied wetland areas (Table 71). The Rwandan National Agricultural Survey recorded just more than 1 percent of households to be engaged in fishing activities in the study region (Gatsibo, Kayonza, Kirehe, and Nyagatare Districts). In the villages surrounding the Rweru-Mugesera Wetlands, it has been reported that between 5-10 percent of households engage in fishing activities. This is significantly lower than the 81 percent and 59 percent of households surrounding the Yala Swamp (Ngombe, n.d.; Abila, 2002) and 16 percent of households surrounding the Kirua Swamp (Turpie *et al.*, 2005). In Rwanda, fisher households caught on average 83 kg of fish per year (NISR, 2010); in 2017, it was estimated that a total of 147 tons of fish were caught from the rivers and lakes of the Akagera Wetlands (REMA, 2019). The average annual production from Lake Rweru is estimated to be between 200 and 250 tons (Karame *et al.*, 2017). The total catch from the Lacs du Nord in Burundi was estimated to be 3,600 tons in 2018 (Ministry of Environment Agriculture and Livestock, 2020), an order of magnitude higher than estimates for Rwanda. It has been reported that fish stocks are declining in the region (FEWS NET, 2012). The three most commonly caught fish are imamba *(Protopterus aethiopicus*), African catfish (*Clarias gariepinus*), and tilapia (*Oreochromis niloticus*).

Data on bushmeat consumption for the region is scarce. Since hunting is illegal, most people are reluctant to divulge information on hunting activities, and studies that do collect figures on hunting are usually considered an underestimate of actual involvement (*e.g.,* see Abila, 2002; Turpie *et al.,* 2005). Hunting usually involves spears, trapping, and in some cases dogs, and is usually carried out to supplement food requirements (Abila, 2002). Before Akagera National Park was fenced, there would have been greater movement of large and medium-sized game between the park and surrounding areas. The construction of the fence to protect wildlife has meant that only small mammals and birds such as small antelope, rodents, wild pigs, and a variety of water birds are likely to be hunted for food in this region. Average household participation in hunting activities surrounding the Yala Swamp was estimated to be 30 percent, and Kirua Swamp just 2 percent (Abila, 2002; Turpie *et al.*, 2005).

Table 71. Proportion of rural households harvesting wild animal resources within each country in the Rweru-Mugesera-Akagera Wetlands and the estimated demand per average household per year

COUNTRY	SMALL MAMMALS & BIRDS		FISH	
	% RURAL HH	KG/HH/Y	% RURAL HH	KG/HH/Y
Rwanda	7	2.6	1.5	1.2
Tanzania	7	2.6	1.3	1.1
Burundi	7	2.6	5	11.5

THE SUPPLY, USE, AND VALUE OF HARVESTED WILD RESOURCES

To briefly recap, the resource use results are the combined product of natural resource stocks, the availability of these resources for harvesting (protected area status), and the local demand for the various resources. Stocks of natural resources per unit area varied according to habitat type and condition. However, the supply of natural resources was also moderated by protected area status, as we reduced the proportional availability of natural resources where they occurred within protected areas. The magnitude of this reduction varied according to the level of protection. Finally, the data for available stocks per hectare was combined with estimated household demand per hectare. Demand is a function of both the average quantity of resources used per household, and the number of households in the area (population density). For this study area, only the resources provided by wetland habitats were valued.

A unique feature of the resource use patterns in the Rweru-Mugesera-Akagera Wetlands is the large core area occupied by Akagera National Park, where use of all natural resources was assumed to be zero (Figure 89-Figure 90). This judgment was made due to the fenced nature of the park, which we assumed would preclude all resource use by surrounding communities. Outside of Akagera National Park, patterns of resource use vary by type of wetland habitat. The distribution of wetland habitats is highest in the central part of the wildlife landscape and southwards toward Lake Rweru. Hence, aquatic resources such as reeds and sedges are more prevalent here, particularly along the Akagera River and north of Rweru in Rwanda. The total value of wild harvested resources was estimated to be US$50.2 million across for the whole wetland area: US$12.4 million in Burundi, US$26.1 million in Rwanda, and US$11.7 million in Tanzania.

The extensive use of reeds and sedges is a distinguishing feature of the Rweru-Mugesera-Akagera Wetlands. Unsurprisingly, reeds and sedges have the highest monetary value per hectare of all the harvested resources analyzed, by a considerable margin (Table 72).This is facilitated by the extensive papyrus swamps that extend across the region (Figure 89). The model estimated generally more intensive use of reeds and sedges per hectare in Burundi and Rwanda. It should also be remembered that a sizable proportion of the papyrus swamps on the Rwandan side had a consumptive use value of zero due to the presence of Akagera National Park. Fishing is an important activity in this region, with per hectare values highest in Burundi. Note that the assessment was only of the wetland area, and thus did not include some of the terrestrial resources listed in the other landscapes.

Table 72. Average quantities, monetary values per hectare, and total value (US$ millions) for subsistence harvesting of wetland resources in the Rweru-Mugesera-Akagera Wetlands study region

RESOURCE GROUP	UNIT	RWANDA			TANZANIA			BURUNDI		
		USE (UNIT/HA)	US$/HA	TOTAL US$ MN	USE (UNIT/HA)	US$/HA	TOTAL US$ MN	USE (UNIT/HA)	US$/HA	TOTAL US$ MN
Reeds & sedges	kg	175.92	103.79	24.75	126.84	74.84	10.92	601.90	355.12	11.97
Wild plant foods & medicines	kg	3.25	4.24	1.01	3.23	3.97	0.60	5.13	7.48	0.25
Small mammals and birds	kg	0.65	0.88	0.21	0.71	0.96	0.14	1.22	1.63	0.05
Fish	kg	1.24	0.52	0.12	0.47	0.20	0.02	8.53	3.58	0.12

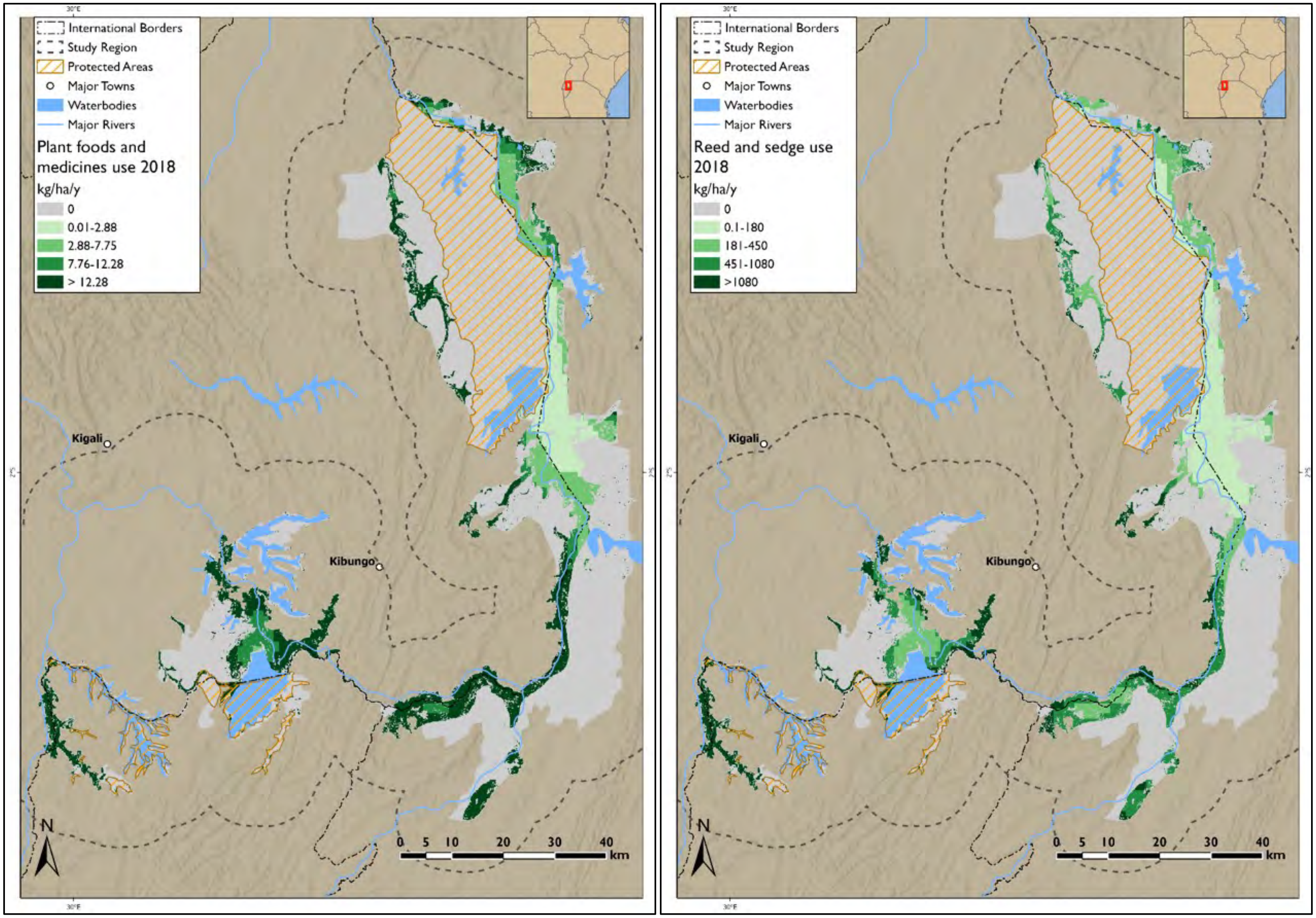


Figure 89. Estimated variation in the subsistence harvesting of wild plant foods and medicines (left) and reeds and sedges (right) across the Rweru-Mugesera-Akagera Wetlands

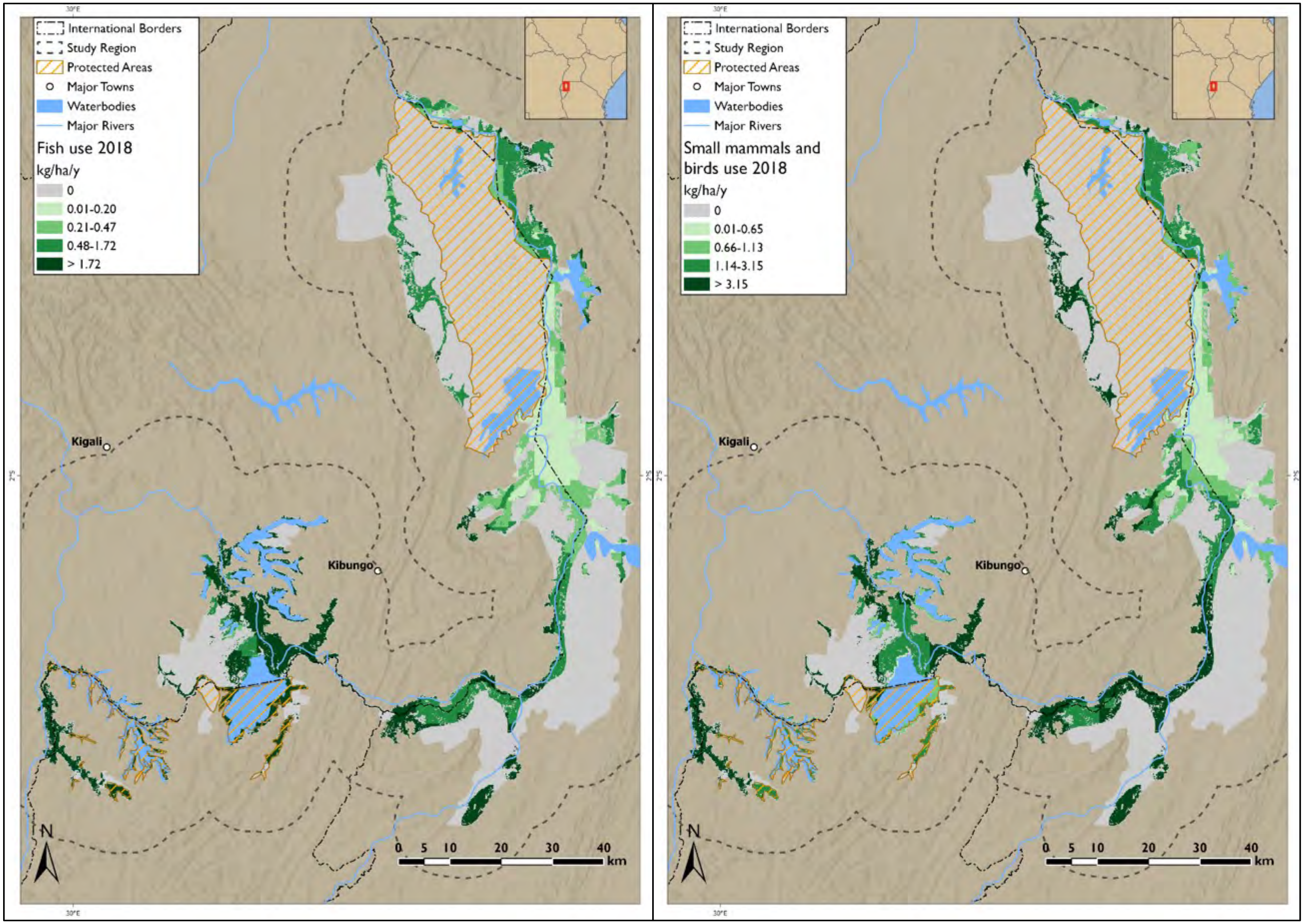


Figure 90. Estimated variation in the subsistence harvesting of fish (left) and small mammals and birds (right) across the Rweru-Mugesera-Akagera Wetlands

SUMMARY

The Rweru-Mugesera-Akagera Wetlands complex is one of the largest wetland areas in the basins surrounding Lake Victoria. Large areas of papyrus swamps cover this area, as well as several open water lakes that are home to a wide array of birds and wildlife. Parts of the wetland system are protected in Burundi and Rwanda, with Akagera National Park being one of the largest protected wetlands in East Africa.

The contribution of the Rweru-Mugesera-Akagera Wetlands to tourism value was estimated to be US$5.3 million in 2018: US$4.5 million in Rwanda, US$0.7 million in Tanzania, and US$0.08 million in Burundi. In Rwanda, this represents just over 1 percent of the total tourism value in the country. The tourism value of Akagera National Park was estimated to be US$2.6 million per year (US$26/ha/y), accounting for 50 percent of the total tourism value across the Rweru-Mugesera-Akagera Wetlands. Nature-based tourism also generates an estimated $7 million in net benefits to international visitors.

The large wetland system, with its extensive areas of papyrus-dominated swamps, is able to remove large quantities of the nutrients that enter it as a result of human activities in its catchment areas. These nutrients would otherwise reach Lake Victoria, adding to the problems of eutrophication there. This service was estimated to be worth on the order of US$0.7 million per year. In addition, the high biomass of the wetland system stores an estimated 92 million tons of carbon, which is estimated to avoid local climate change damage costs around US$8 million per year and global damages of US$7 billion per year.

Wetland resources, harvested for materials and food, play an important role in supporting the livelihoods of people across this region. The value of all wild resources harvested totaled US$50.2 million for the wetlands. Including a conservative estimate of the existence value of biodiversity, the wildlife landscape is estimated to be worth at least $300/ha/year on average to East Africa and almost $35,000/h/year globally. The total values for this landscape within each country represent less than 1 percent of GDP.

Table 73. Summary of the benefits derived from ecosystem services of the Rweru-Mugesera-Akagera Wetlands

	BURUNDI	RWANDA	TANZANIA	REGION	REST OF WORLD	TOTAL
Nature-based tourism	0.08	4.50	0.70	5.28	7	12
Biodiversity existence	-	0.003	0.02	0.02	89	89
Water quality amelioration	0.23	0.23	0.23	0.70	-	1
Carbon storage	0.01	1.81	6.34	8.15	7,328	7,337
Harvested resources	12.40	26.10	11.70	50.20	-	50
Total value $ millions per year	**12.7**	**32.6**	**19.0**	**64.4**	**7,424**	**7,488**
Total value $ per ha per year	**524.7**	**257.7**	**298.2**	**299.9**	**34,598**	**34,898**

IMPLICATIONS FOR THE FUTURE UNDER A BAU SCENARIO

The Rweru-Mugesera-Akagera Wetlands landscape has already experienced extensive transformation across much of its area. Rwanda's Akagera National Park is the only remaining stronghold for wildlife in the complex today. Outside the park, little natural non-aquatic habitat remains today in the Burundian and Rwandan portions of the landscape, while more intact natural cover remains in parts of the Tanzanian portion. Nevertheless, extensive lake and swamp habitats still survive, particularly in the Akagera region. Overall, the wetland complex experiences substantial pressure from the dense human populations that characterize much of the region. Furthermore, existing pressures are set to be exacerbated by climate change in the future. Thus, in the following sections, we start by describing some of the expected impacts of a range of existing pressures based on past trends. We then describe some of the expected impacts of future climate change derived from modelling studies. Finally, we draw together the discussion on existing pressures and future climate change impacts to predict the future of wildlife, habitats, and ecosystem services provided by the wetland complex under a business-as-usual scenario.

HABITAT CONVERSION TO CULTIVATION AND SETTLEMENT

Extensive conversion of habitats has occurred across the Rweru-Mugesera-Akagera Wetlands. Fertile soils and enhanced water availability throughout the year make wetlands attractive sites for cultivation (Kassenga, 1997; Dixon & Wood, 2003; Khan *et al.*, 2019). As the region is more drought-prone than wetter areas to the east and west, the motivation to cultivate in and around swamps and lakes is particularly strong here (Khan *et al.*, 2019). Ongoing population growth and increased scarcity of land has also pushed people into more marginal areas in search of new space for agricultural expansion, including wetlands (Dixon & Wood, 2003). Traditionally, small areas on the edges of wetlands would be cultivated. More recently, a general shift from upland to wetland agriculture has occurred, as continued growth in population and food demand have resulted in degradation of intensively cultivated upslope areas (Nabahungu & Visser, 2013; Leemhuis *et al.*, 2016). Hence, the search for new agricultural frontiers has led to more intensive land use in wetlands, including the drainage of entire wetlands (Kassenga, 1997; Dixon & Wood, 2003; Sakané *et al.*, 2011; Nabahungu & Visser, 2013; Leemhuis *et al.*, 2016; Rukundo *et al.*, 2018). Particularly in the Rwandan portion of the landscape, wetlands have also become the sites of large-scale agro-industrial developments like sugar cane plantations, resulting in substantial habitat loss (Nsengimana, Weihler & Kaplin, 2017). Nevertheless, the large size of several lakes and swamps in the wetland complex appears to have allowed them to remain relatively intact in the face of heavy encroachment pressures (Ndayisaba *et al.*, 2017). Buffer zones around some lakes and wetlands, in which agriculture is not permitted, have been created in the Rwandan portion of the landscape (Karame *et al.*, 2017). While this could help prevent further loss of riparian habitat, Karame *et al.* (2017) also found that most local people did not understand and/or resented the loss of land to the buffer zones, which may undermine the success of this intervention. Settlement and urbanization have also resulted in substantial loss of habitat, particularly in the northwest part of the landscape, which has experienced urban sprawl from Kigali and extensive habitat transformation. Overall, it has been estimated that 30 percent of swampland (90,000 ha) has been lost to cultivation in the Rwandan portion of the Akagera Basin (Republic of Rwanda, 2010). This signifies a substantial loss of habitat for large wildlife associated with wetland habitats, such as hippopotamus and sitatunga (Fischer *et al.*, 2011). Key informants from the Mugesera-Rweru portion of the wetlands confirmed that habitat loss is ongoing and contributing to the disappearance of large mammals and birdlife. Papyrus habitats outside of protected areas in the landscape thus appear to be at serious risk of conversion.

The loss of wetland habitat has been accompanied by extensive loss of surrounding forest and woodland habitats, particularly in the western part of the landscape (Wasige *et al.*, 2012a). Compared to the rest of the complex, habitats in the Akagera region were relatively untouched until the Rwandan civil war. This region experienced substantial habitat loss from the 1990s, when large portions of Akagera National Park and all of the former Mutara Game Reserve were set aside to accommodate thousands of refugees returning to Rwanda after the civil war (Kanyamibwa, 1998). This resulted in substantial habitat transformation and degazettement of the affected areas, accounting for 60 percent of the protected area coverage at the time (Apio *et al.*, 2015). However, the subsequent re-establishment of conservation law enforcement in Akagera National Park has prevented further habitat degradation and loss in the remaining protected area. Sizeable remnants of non-aquatic natural habitat are thus limited to the remaining portion of Akagera National Park and surrounding areas, as well as parts of the Tanzanian portion of the landscape. This has meant a near total loss of habitat and landscape connectivity for savanna wildlife in the rest of the complex. Given the dense and growing human populations and intensity of cultivation that has resulted, it is unlikely these changes will be reversed, increasing the importance of securing remaining wildlife habitat in the region. Indeed, Global Forest Change data indicates deforestation is ongoing and appears to be increasing in speed in the broader landscape surrounding the wetlands, though much year-to-year variation is evident. 2017 was a particularly bad year for deforestation, with about 2,300 ha of forest lost (Figure 91).

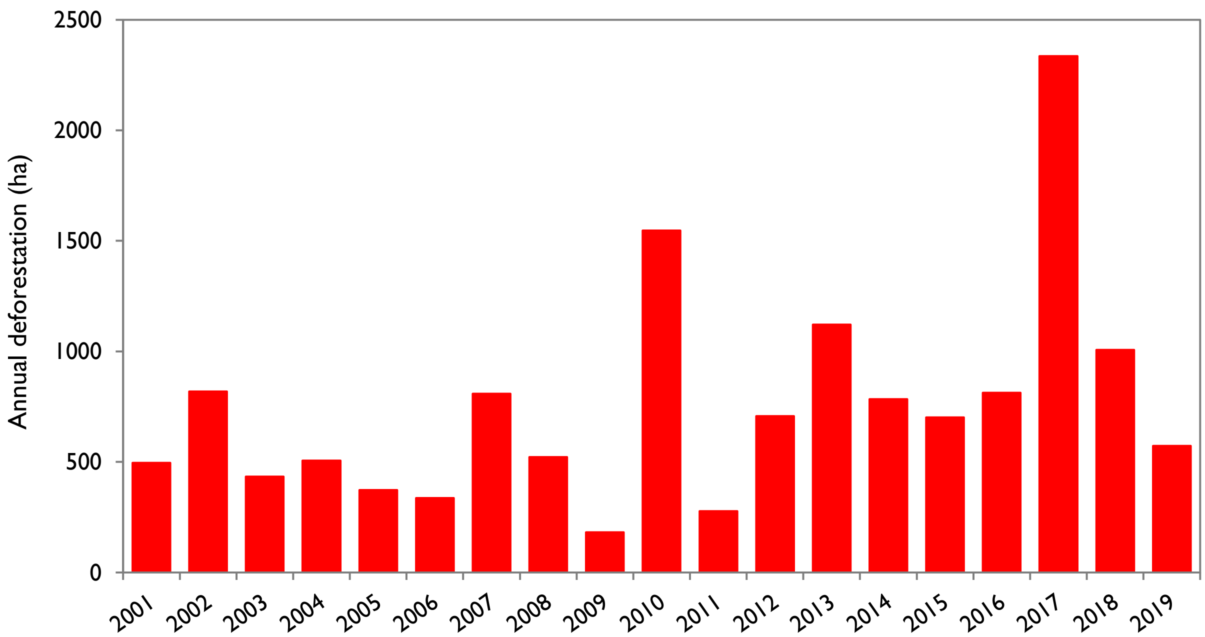


Figure 91. Annual forest loss in the Rweru-Mugesera-Akagera Wetlands Complex from 2001 to 2019

Source Hanen/UMD/Google/USGS/NASA

The ESA CCI land cover data at 300 m resolution suggest that the area under crops grew until the mid-2000s, then again during 2010 to 2015, but has decreased since then (Table 4174). The Copernicus 100 m landcover data series, which goes back to 2015, suggests that there has been an increase in cropland in the area within 10 km of the wetlands from 2015 to 2018 of 15,580 hectares per year. This highlights the potential inaccuracy of land cover data products and the need for ground-truthing. Given the information in the literature, the latter trend is more likely. The built-up area has increased at a fairly rapid rate, which is a concern that most of this landscape is ostensibly under protection.

Table 74. Extent and annual rates of change of land cover classes in the Rweru-Mugesera-Akagera Wetlands 10 km buffer area from 1992 to 2004 and from 2004 to 2018

LAND COVER CHANGE	1992 TO 1998	1998 TO 2004	2004 TO 2010	2010 TO 2015	2015 TO 2018
Average annual change in area under crops (ha/year)	1,559	203	-1,195	-238	-2,721
Average annual change in built-up area (ha/year)	14	1,066	372	362	282

Source: Based on ESA CCI Land Cover 300m resolution (European Space Agency, 2018)

OVER-HARVESTING OF NATURAL RESOURCES

Households living around the wetland complex are often highly dependent on resources harvested from the wetlands. Rapid population growth in the region has resulted in excessive harvesting pressures, leading to degradation of wetland habitats (Karame *et al.*, 2017). For example, over-harvesting and burning have caused a reduction in papyrus habitats in various parts of the landscape (Fischer *et al.*, 2011), important habitat for unique species localized to these areas. The wetlands also experience high pressure from livestock. Remaining non-cultivated areas provide an attractive forage source, particularly during the dry season, as well as a source of drinking water (Kassenga, 1997; Sakané *et al.*, 2011). However, excessive livestock pressure degrades wetland vegetation cover and promotes erosion, increasing sedimentation of aquatic habitats. Livestock had a particularly drastic impact on the Akagera portion of the landscape in the 1990s. Over this time, Akagera National Park and the degazetted Mutara Game Reserve were host to about 600,000-700,000 cattle belonging to refugees returning home after the civil war (Kanyamibwa, 1998). This resulted in intense competition for limited resources between remaining wildlife and the burgeoning livestock populations (Bariyanga *et al.*, 2016). Nonetheless, notable populations of some species still exist in less-transformed areas surrounding the park. However, the future of these wildlife populations is uncertain with intensifying livestock pressure outside the park (Bariyanga *et al.*, 2016).

POLLUTION OF WATER RESOURCES

In addition to causing direct loss of habitat, the expansion of cultivation in the region degrades wetland ecosystems through increased export of sediments and nutrients. Furthermore, farmers are resorting to using larger amounts of fertilizer and pesticides in an effort to meet growing food demand, particularly in Rwanda (Khan *et al.*, 2019). As a result, nutrient pollution and eutrophication from agricultural runoff are worsening threats to the system (Wasige *et al.*, 2012a). Already, nutrient loads exceeding the safe level for aquatic life have been reported from various sampling sites along the Akagera River (Wali *et al.*, 2011). In response to these threats, key informants call for measures to reduce the application of agricultural chemicals in the landscape and to support communities in adopting more sustainable and environmentally friendly farming practices. Agriculture also results in a high demand for water from the wetlands, further compromising the hydrological functioning of the wetland ecosystems (Nabahungu, 2012; Khan *et al.*, 2019).

Pollution and water demands from agriculture are likely to increase as populations continue to expand, particularly with the intensive cultivation schemes being carried out in Rwanda as a solution to high food demand (Nsengimana *et al.*, 2017). High rates of urbanization, especially in the vicinity of Kigali, also add substantially to the nutrient and sediment loads entering the wetland system. This includes excessive siltation from bare soils associated with construction areas, and toxic chemicals from untreated sewage and industrial runoff (Nabahungu, 2012). It is thought that the combined effects of water abstraction and sedimentation are one likely driver of the reduction in lake levels noted in parts of the wetlands complex (Ndayisaba *et al.*, 2017). The effects of these pollution sources also extends beyond the Rweru-Mugesera-Akagera complex, with the Akagera Basin thought to be the largest source of sediment and nutrient loads entering Lake Victoria (Wasige *et al.*, 2012b).

Key informants also report livestock numbers in the area continue to increase. These can be a further source of habitat degradation and pollution as overgrazing will increase erosion and export of sediment to rivers and wetlands in the areas. Watering of livestock at rivers and wetlands can also cause denudation and destabilization of riparian areas, significantly affecting sediment loads. Increasing numbers of livestock and thus manure loads will also contribute to nutrient pollution challenges.

INVASIVE SPECIES

Invasive species are another cause of habitat degradation in the wetland complex. Water hyacinth (*Eichhornia crassipes)* has spread extensively through the region, and is thought to have contributed to reduced water levels in lakes (Ndayisaba *et al.*, 2017; Mukarugwiro *et al.*, 2019). Based on remote sensing data, Mukarugwiro *et al.* (2019) estimated that water hyacinth covered about 100,000 ha of the Rweru-Mugesera-Akagera complex. This amounts to about half of the area covered by indigenous papyrus vegetation, and a third of the remaining open water area. Furthermore, Mukarugwiro *et al.* (2019) note that lakes within Akagera National Park have particularly bad invasions of water hyacinth, especially Lakes Hago and Rwanyakizinga, which have 40 percent and 36 percent water hyacinth coverage, respectively. This is well above the acceptable 10 percent threshold, below which water hyacinth invasions might not negatively affect local biodiversity and ecosystem health (Jones, 2009).

Dense water hyacinth cover reduces light penetration, affecting the ability of phytoplankton and submerged plants to photosynthesize. Similar to eutrophication, this can lead to a decline in oxygen levels with catastrophic effects on fish and other aquatic life (Howard & Harley, 1997). According to key informants, some areas have been so severely invaded that the flow of the Akagera River has been altered. This has resulted in increased transport of sediment to Lake Rweru, affecting water quality and potentially resulting in drying of the lake if nothing is done.

With control efforts in the region generally failing to keep pace with the expansion of water hyacinth (Mukarugwiro *et al.*, 2019), it is likely that the invasion and resulting impacts on aquatic ecosystems will worsen in the future. This was confirmed by key informants, who reported that control efforts are being hampered by a lack of appropriate equipment and inadequate commitments from local communities to assist with managing the issue.

Other invasive plant species reported by key informants include the giant sensitive tree (*Mimosa pigra*). This is a prickly shrub or small tree from tropical America that invades disturbed wetland areas and prevents the regeneration of indigenous vegetation. Key informants also identified invasive fish species such as the marbled lung fish (*Protopterus aethiopicus*) and sharptooth catfish (*Clarias gariepinus*) as a threat. These predate on smaller indigenous fish species in the wetlands, particularly tilapia.

HUNTING PRESSURES

Extensive habitat conversion has virtually eliminated most large wildlife across much of the wetlands complex, leaving little possibility for hunting. The Akagera region of the landscape was relatively less affected by human activities until the 1980s, when wildlife populations were decimated during the Rwandan civil war. By the early 1990s, it was estimated that about 90 percent of the large mammals in Akagera National Park had been lost during the conflict (Kanyamibwa, 1998). Further hunting of wildlife occurred with the return of refugees to the area after the end of the war, many of whom used automatic weapons to hunt in the park. This culminated in the extinction of lion and black rhino from the landscape, though these species have since been reintroduced (Bariyanga *et al.*, 2016). Hunting pressure on the remaining portion of Akagera National Park today has been much reduced, following the establishment of effective conservation law enforcement and fencing of the park (Apio *et al.*, 2015). Key informants also reported that community-based natural resources management initiatives and revenue-sharing arrangements have contributed to the drastic reduction in poaching. Some poaching still occurs along wetland margins, particularly close to the Rwanda/Tanzania border (Macpherson, 2013). This is largely thought to target hippopotamus and fish, and does not appear to be having a substantial negative effect on wildlife populations in the park at present.

Remnant populations of wildlife in Rwanda and Burundi outside of Akagera National Park are more seriously threatened by hunting, making the future of these populations precarious (Bariyanga *et al.*, 2016). All things being equal, demand for bushmeat in these areas will increase as human populations continue to grow, as was noted by key informants. Wildlife populations in national parks in the Tanzanian portion of the landscape are also threatened by bushmeat hunting. A key informant reported that around 8,000 kg of poached bushmeat are recovered each year from Ibanda-Kyerwa and neighboring Rumanyika-Karagwe National Parks. The most commonly targeted species for poaching here include zebra, wildebeest, various antelope, and hippopotamus. According to Tanzanian informants, most people in the area prefer wild meat, indicating that the problem is not simply driven by poverty alone. Suggestions for reducing bushmeat hunting pressures in the landscape included improved nature-based tourism revenue-sharing arrangements, better enforcement of conservation laws, and awareness raising among communities. A Tanzanian key informant also suggested that TANAPA should periodically provide local communities with meat from the parks sold at a low cost.

HUMAN WILDLIFE CONFLICT

Some human-wildlife conflict occurs in the wetland, but the problem does not appear to be as serious as in the other transboundary landscapes. Prior to the fencing of Akagera National Park, human-wildlife conflict was a serious problem in areas that were degazetted in the 1990s. Since these areas were still used by wildlife, significant conflict occurred between wild animals and returning refugees settling in these degazetted areas, including crop raiding and attacks on humans (Bariyanga *et al.*, 2016). However, following fencing of the park and attempts to drive wildlife back into the fenced area, human-wildlife conflict has decreased in the Akagera region (Bariyanga *et al.*, 2016). Key informants also report that park management has dug trenches to separate wild habitats from settlements and fields, which has helped reduce HWC. Nevertheless, key informants reported that there are still hyena and leopard found outside the park, which attack livestock, along with crop-raiding baboons. Park management continues to work collaboratively with local communities to reduce HWC around Akagera. For example, a community key informant reported that communities have been trained how to handle and safely capture hyena found outside the park, and receive a reward of around US$50 if they capture and hand over live hyenas to the park authorities. Key informants around Ibanda-Kyerwa also reported problems of crop raiding by elephant and livestock predation by hyaena. There have also been incidences of people killed by wildlife here. If unchecked, these problems will likely contribute to decreased tolerance of wildlife and retaliatory killings. In the rest of the landscape, hippopotamus can be a source of HWC where they occur, as they often destroy crops found along the edges of lake habitats (Karame *et al.*, 2017).

OVERALL IMPACT ON WILDLIFE POPULATIONS

The above pressures have resulted in substantial loss of wildlife across the wetland complex, with large populations now limited to Akagera National Park. Extensive habitat conversion eliminated large wildlife from most of the landscape some time ago. More recently, wildlife populations in the Akagera region were decimated in the 1990s, as has been discussed above. Improved management has led to an impressive recovery of many wildlife species in the remaining portion of the park, aided by the reintroduction of species like lion and black rhino. Outside of Akagera National Park, only a few species of large wildlife remain, including hippopotamus, crocodile, and certain ungulates like sitatunga and bushbuck, while large predators are absent (Fischer *et al.*, 2011; Karame *et al.*, 2017). Furthermore, populations of some of these species appear to be declining further due to habitat loss and hunting. For example, Karame *et al.* (2017) report that sitatunga sightings have become rare in the Rweru-Mugesera Wetlands, despite local residents reporting that the species was seen frequently 10 years before their study was conducted. These trends were confirmed by community key informants from the Rweru-Mugesera portion of the wetlands, who widely reported serious declines or disappearance of large wildlife species.

PROJECTED CHANGES IN TEMPERATURE AND RAINFALL

The Rweru-Mugesera-Akagera Wetlands landscape experiences the greatest seasonality in terms of rainfall of the study area with a clear dry season from June to August and a wet season from October to April. It is predicted that ca. 2050, the mean rainfall will decline, with the greatest proportional decreases in August (-7.9 percent) and September (-6.4 percent). The wet season is generally likely to experience increases with a 15 percent increase predicted for December and 11.6 percent for January. Overall, the mean annual precipitation is predicted to increase by only 9 mm (Figure 92). Mean annual temperature across the landscape is expected to increase by 2.7°C on average, with June through October predicted to increase by at least 2.8°C (Figure 93).

Geographically, the whole area is expected to become wetter, but with the southeastern parts becoming relatively wetter than the rest (Figure 94). The whole area will also become hotter relatively uniformly.

While the annual changes in precipitation are likely to be variable across the landscape, Rwanda's Akagera National Park is predicted to have a relatively low mean annual change in precipitation, with just a 0.7 percent increase. Mean annual temperature, however, may rise by as much as 2.6°C by 2050 (Table 75).

PROJECTED CLIMATE CHANGE IMPACTS ON HABITATS AND WILDLIFE

Climate change is a major threat to biodiversity, affecting both individual species and overall ecosystem functioning (Scheffers *et al.*, 2016). To survive a shift in suitable climate, species may need to either adapt to their changed environment or relocate to more suitable areas (Moritz & Agudo, 2013). However, opportunities to move may be restricted by anthropogenic or natural barriers such as cultivated land, mountain ranges, or water bodies. This challenge is particularly severe in the Rweru-Mugesera-Akagera complex, where extensive natural habitat is limited to the Akagera region and landscape connectivity has been eroded by extensive conversion of habitats to agriculture.

Using the outputs of existing SDMs, the expected combined species richness of mammals, birds, reptiles, and amphibians is shown in Figure 95. The maps indicate expected species richness under current conditions and under the projections of three different climate models for 2070 (model ac, bc, and cc), showing the range in results depending on which future climate model one uses. Note we use the term "expected," because potential species distributions have been interrupted by anthropogenic land use and other pressures. Hence, real species richness is likely to be substantially lower than expected species richness across the parts of the landscape that have been transformed by cultivation. According to the models, current expected species richness was predicted to be highest in the northern parts of the landscape, and the extreme west. Species richness is predicted to decline substantially throughout the complex under all three future climate change scenarios, indicating a substantial loss of biodiversity. This pattern is also reflected when species richness is broken down into the broad taxonomic groupings (birds, mammals, etc.) of animals (see Appendix 5). A number of key informants, particularly those living in the Burundian portion of the wetlands, confirmed that lower rainfall and droughts are already a serious issue, and attributed declines and disappearance of wildlife to this.

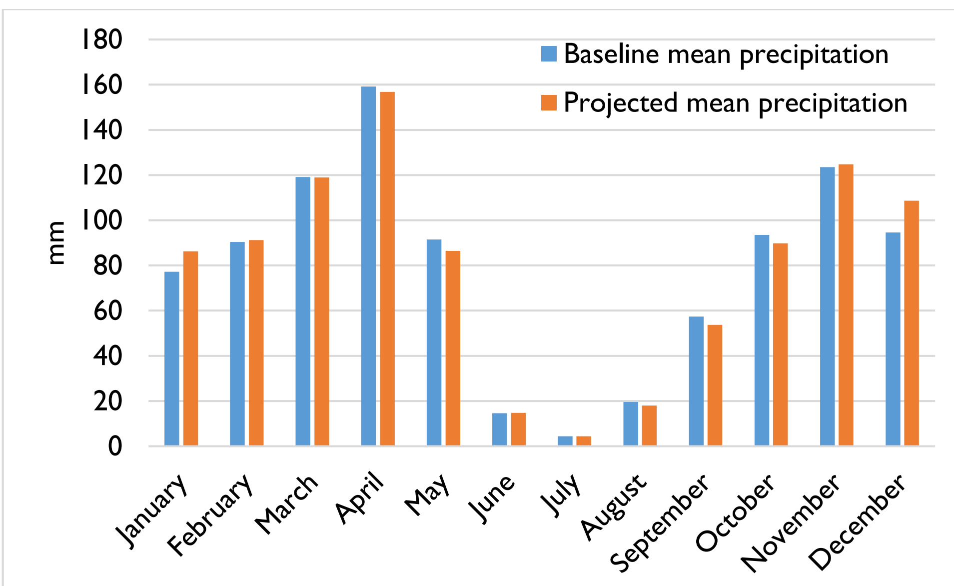


Figure 92. A comparison between historic and projected mean monthly precipitation (mm)

Source: Based on data from WorldClim Version2 and CMIP5

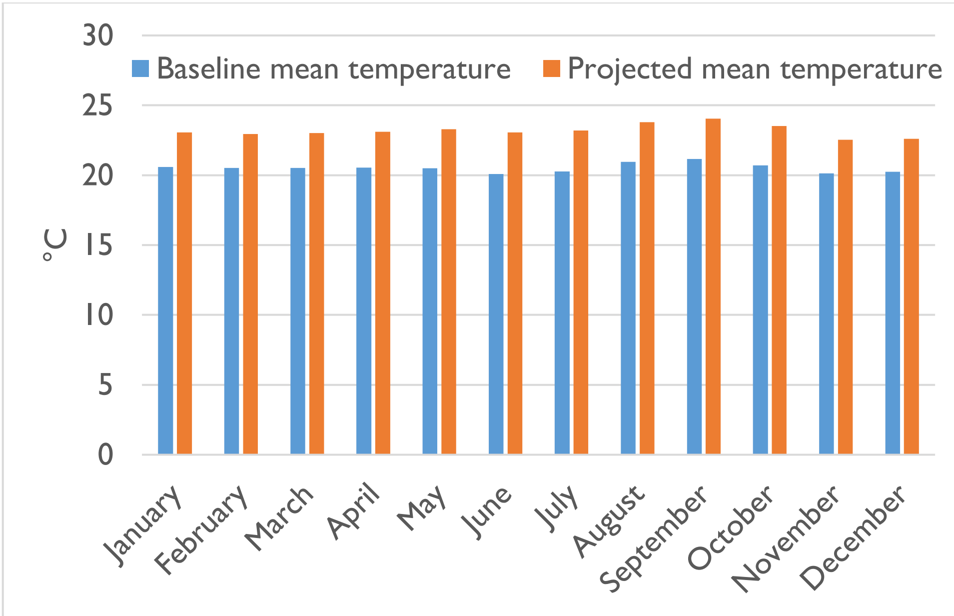


Figure 93. A comparison between historic and projected mean monthly temperature (°C) for the Ruweru-Mugesera-Akagera wetlands

Source: Based on data from WorldClim Version2 and CMIP5.

Table 75. Historic, projected, and percentage changes for mean annual temperature (°C) and total annual precipitation (mm) for key protected areas in the study area

	MEAN TEMPERATURE (°C)			MEAN PRECIPITATION (MM)		
PROTECTED AREA	HISTORIC ANNUAL AVG.	PROJECTED ANNUAL AVG.	CHANGE	HISTORIC ANNUAL TOTAL	PROJECTED ANNUAL TOTAL	% CHANGE
Akagera	20.8	23.4	2.6	962	969	0.7

* Other protected areas in the study area were not large enough to rasterize and generate statistics for in GIS.

Source: Based on data from WorldClim Version2 and CMIP5. Protected areas are listed in descending order of area.

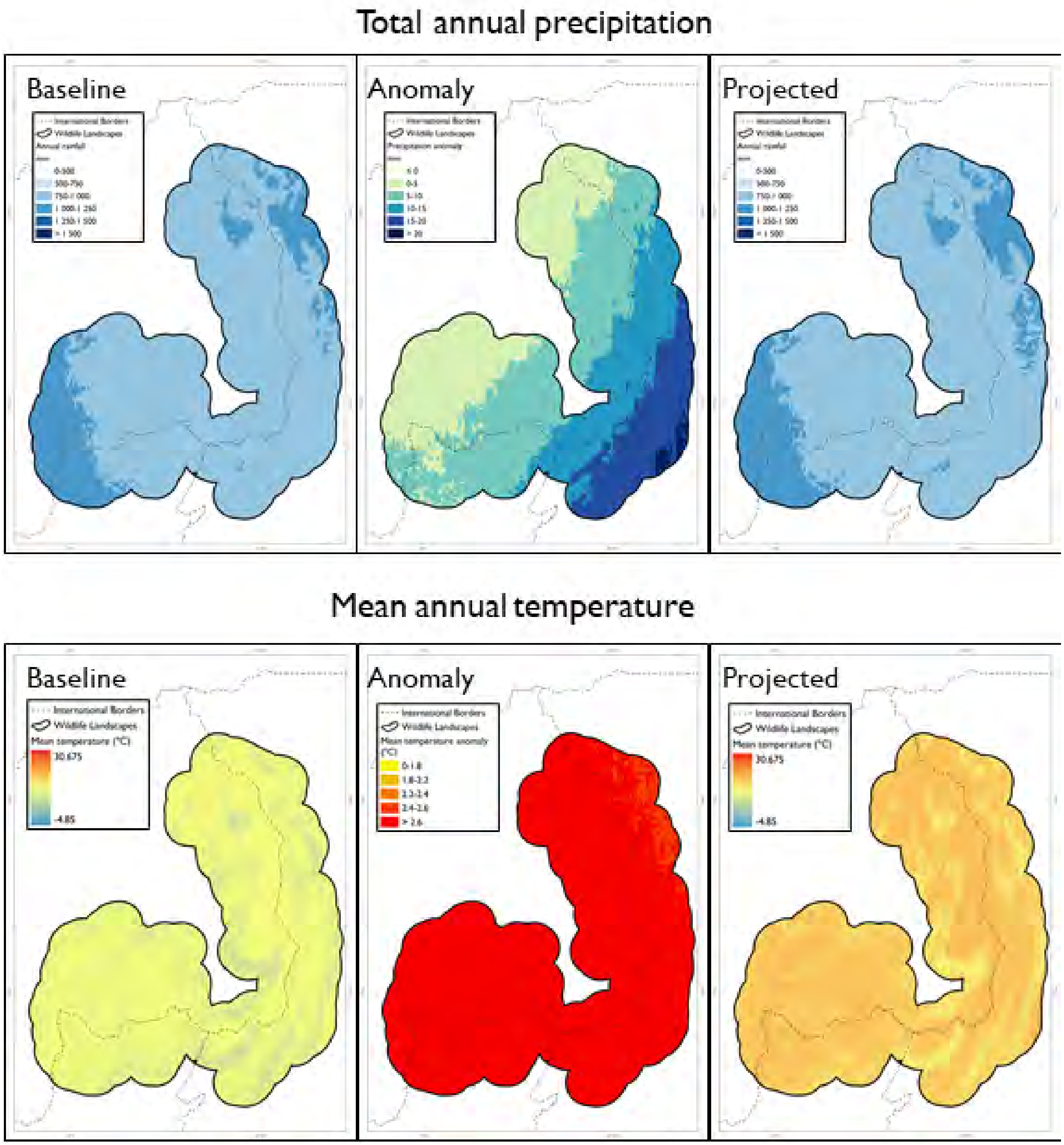


Figure 94. Baseline/historic (1960 – 1990) and projected (2040 – 2060) total annual precipitation (mm) and mean annual temperature (°C) across the Rweru-Mugesera-Akagera Wetlands landscape

Source: Based on data from WorldClim Version2 and CMIP5

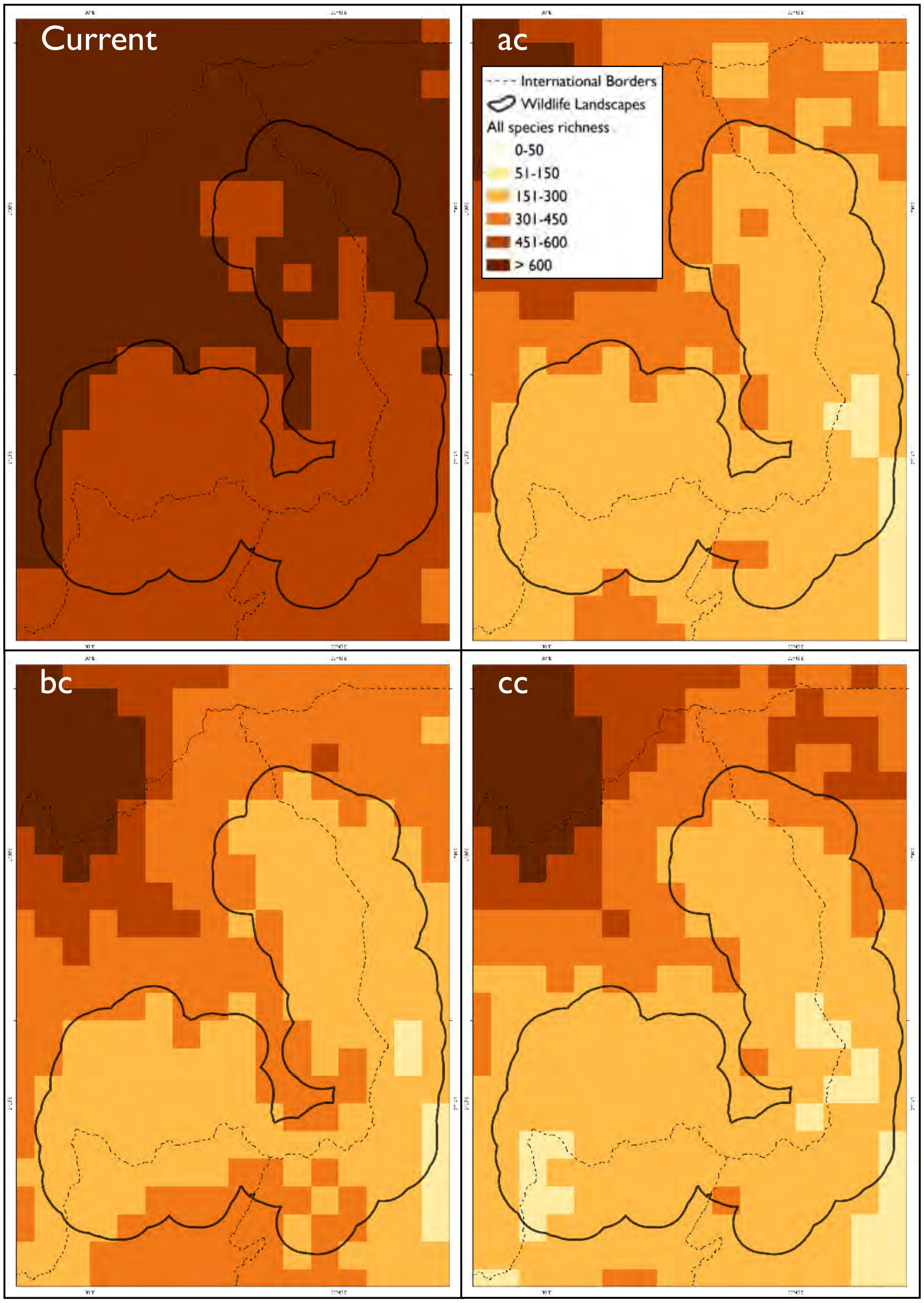


Figure 95. Current geographic variation in expected species richness (amphibians, birds, mammals, and reptiles) for the Rweru-Mugesera-Akagera Wetlands landscape, followed by the projected expected species richness pattern under each of the three future climate scenarios used

Source: Based on modelled species distributions from Conservation International

PROJECTED CLIMATE CHANGE IMPACTS ON CROP SUITABILITY

According to the FAO's EcoCrop analytical tool (FAO, 2010), the suitable area and suitability scores for most crops will increase slightly in the future compared to present conditions. Sorghum was the only notable example of a species where suitability was predicted to increase markedly in the future. Overall, these findings suggest climate will remain conducive to continuing cultivation in the region in the future. The suitability predictions for individual crop species are described further in Table 76, based on the maps shown in Figure 96. Suitability is described in terms of the suitable area for a given crop (*i.e.*, the region with a suitability score of greater than 0), as well as the relative suitability score, which ranges from 0 (unsuitable) to 1 (optimum conditions).

Table 76. Summary of the expected changes in the suitable area and suitability scores for crops in the Rweru-Mugesera-Akagera complex and immediate surrounds, based on the maps shown in Figure 96

CROP	CURRENT SITUATION	IMPACT
Beans	Uniformly high suitability across the landscape (hence map not shown).	Suitability remains uniformly high across the landscape.
Cassava	Entire landscape suitable, while areas of highest suitability are found in the northeast and southwest of the landscape.	Entire landscape remains suitable, with little change in suitability scores. Small area around the Burundi/ Rwanda/Tanzania border confluence increases in suitability.
Plantain	Suitable area limited to the northeast and southwest of the landscape. Suitability mostly low across the suitable area, peaking in the extreme southwest.	Suitable area expands slightly in the northeast and southwest of the landscape, while a small part in the central part of the landscape becomes suitable. Increases in suitability also predicted across much of the suitable area.
Sorghum	Entire landscape suitable, with uniform suitability scores across the landscape.	Entire landscape remains suitable, while suitability increases throughout the landscape. High suitability across much of the northern and western parts of the landscape.
Maize	Virtually all of the landscape suitable, but high suitability limited to the southwest of the complex and isolated patches in the east and northeast.	Virtually all of the landscape remains suitable, while suitability increases slightly over much of the suitable area.
Millet	Entire landscape suitable, with high suitability across the majority of the landscape.	Entire landscape remains suitable, with high suitability spreading further in the landscape.

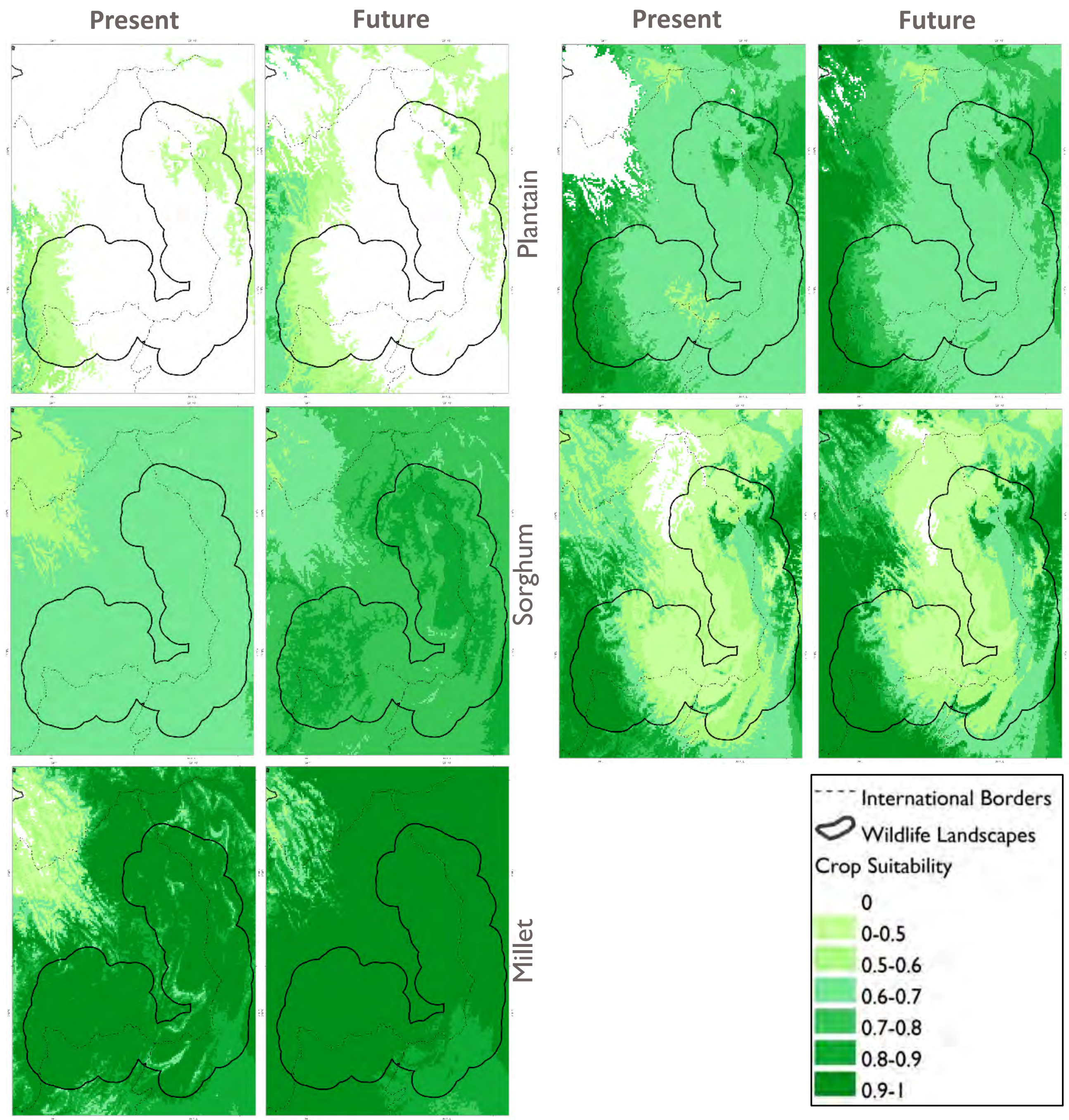


Figure 96. Estimated present and future suitability for some of the key crops grown in and around the Rweru-Mugesera-Akagera Wetlands.

Source: Model outputs generated using the FAO EcoCrop database and model and climate projections for 2040-60

POTENTIAL IMPLICATIONS OF A BUSINESS-AS-USUAL SCENARIO

This section provides an integrated, qualitative assessment of the impacts of a business-as-usual scenario on wildlife, ecosystem services, and human wellbeing over the period from the baseline (2018) to 2030. The combination of 1) increasing population and demand for land and resources and 2) the impacts of climate change on habitats, species, and agriculture need to be considered. There is a great deal of uncertainty in this. Notwithstanding these caveats, the following impacts could be expected.

Further conversion of wetland habitat could occur. Population growth and scarcity of land in the region mean further conversion of wetland habitats to agriculture can be expected to meet food demands. Sprawling urban settlements will also increasingly encroach on the wetlands. Over-harvesting of wetland resources and burning of papyrus will add to habitat loss. Based on population trends, it was estimated that demand for papyrus from communities surrounding the wetlands would increase by 84 percent by 2050 in a BAU scenario, which could have a substantial impact on papyrus stocks, particularly where population densities and demand for papyrus are already high. Due to the role papyrus plays in removing sediment, nutrients, and other pollutants entering waterbodies (Kiwango & Wolanski, 2008), over-harvesting and conversion of papyrus to agriculture could reduce water quality in the lakes and rivers of the wetland system. This would have a negative impact on people and livestock who depend on the wetlands as a water source. Loss of papyrus could also negatively affect aquatic biodiversity like fish, which use papyrus as a nursery habitat for their larvae (Mnaya & Wolanski, 2002). This would in turn compromise the livelihoods of local people dependent on fish as a source of food and/or income.

Loss of terrestrial habitats surrounding the wetlands is likely to continue and may accelerate. Global Forest Change 2000-2019 data suggests woody cover loss has been increasing in speed in recent years and provides an estimation of possible future habitat loss. As with the Albertine Rift Forests landscape, the current increasing trajectory suggests woody cover loss could continue to accelerate into the future as population and demand for fuelwood increases, and more woody habitat is converted to agriculture and settlement. By 2050, it was estimated that a **further 63,000 ha of woody cover could be lost under BAU, representing 17 percent of the area currently covered by trees.** Future loss of remaining woody habitat surrounding the wetlands will likely lead to increased pollution and siltation of the wetlands, with consequences for both biodiversity and human livelihoods, as described further below.

Nutrient and industrial pollution could worsen. Key informants already identify over-reliance on chemical fertilizers as a major driver of wetland degradation. Nutrient pollution from fertilizers will likely increase in the future as more land is converted to agriculture and fertilizer usage increases in an effort to improve land productivity. Sewage and industrial runoff are also expected to increase as urban and industrial areas spread further. These would have a negative impact on wetland ecosystems through algal blooms and eutrophication, compromising biodiversity and livelihoods due to die-offs of fish while reducing the quality of drinking water for livestock and people (Wali *et al.*, 2011). As discussed above, these impacts of pollution on biodiversity and livelihoods could be worsened if the capacity of the wetlands to remove pollutants is reduced by the overharvesting and clearance of papyrus.

Water hyacinth could continue to spread, negatively affecting ecosystems and aquatic life. Lack of an effective control strategy means spread of water hyacinth will continue, leading to declines in the health of aquatic plants, fish, and other aquatic life. This will compromise fishing livelihoods by reducing stocks and making navigation of the wetlands by boat increasingly difficult. Water hyacinth may also contribute to declining lake levels.

The wetlands could experience increased siltation. Further expansion and intensification of agriculture will lead to greater sediment runoff into the complex, as will construction activities as settlements spread through the region. Siltation could be exacerbated by the loss of papyrus and other buffering vegetation from cultivation and/or over-harvesting, as this would reduce the capacity of the wetland vegetation to capture sediment entering the system. Increased siltation of the wetlands would compromise aquatic health, as well as affecting water levels and flows in the complex. Reduced water supply due to siltation has already been noted as a problem in nearby wetland areas, such as the Cyabayaga Wetland in northeast Rwanda (Nabahungu & Visser, 2013).

Wetland integrity could be compromised by water abstraction and other hydrological modifications. Rising agricultural water demands will lead to greater abstraction of water from the wetlands, as could rising domestic uses from expanding populations and settlements. In addition, the hydrological functioning of the system may be perturbed by future hydropower developments, which will have knock on impacts on all of its biodiversity.

Disappearance of large wildlife could occur. Further habitat loss and hunting pressures will likely result in the total disappearance of remaining large wildlife outside of Akagera and Ibanda-Kyerwa National Parks, particularly terrestrial species. Reed- and papyrus-dependent species may also become increasingly scarce due to conversion and over-harvesting of papyrus, as has been reported for sitatunga (Karame *et al.*, 2017). Large aquatic wildlife like hippopotamuses might nevertheless persist outside the park (provided sufficient food is available; Karame *et al.*, 2017), as the bigger lakes and wetlands are resilient to conversion, unlike surrounding terrestrial habitats. Nevertheless, hippopotamus in Rweru-Mugesera are also said to be in considerable decline due to habitat loss (Fischer *et al.*, 2011), suggesting that they could also be vulnerable to disappearance under the current trajectory. Further habitat loss would also threaten IUCN-listed species still found outside of Akagera National Park, such as the papyrus gonolek and papyrus yellow-warbler birds. Pressures on wildlife could be accentuated by climate change, with some key informants already blaming drought for the decline or disappearance of wildlife from parts of the wetland complex.

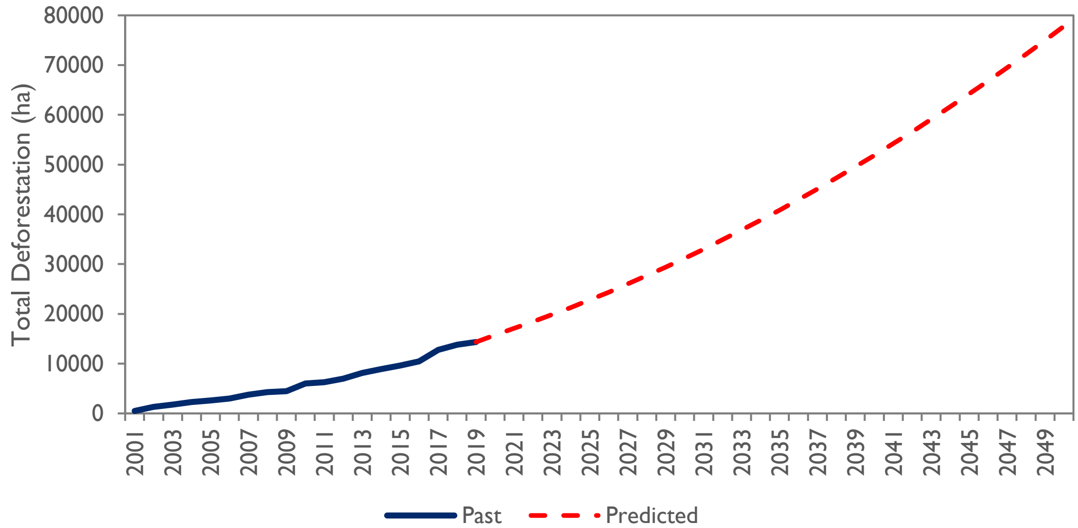


Figure 97. Cumulative deforestation since the year 2000 in the broader Rweru-Mugesera-Akgera wetlands. The solid blue line shows past deforestation derived from Global Forest Change data, while the red dotted line shows predicted future deforestation in a BAU scenario based on trends in deforestation between 2001 and 2019.

The potential overall effects of the above pressures on wildlife and wildlife habitats on ecosystem services under a BAU scenario can be summarized as follows.

The ability of the wetlands to remove pollutants is expected to decrease. Conversion of wetlands to agriculture and over-harvesting and burning of papyrus reduce the ability of the wetland complex to filter out nutrients. This will be exacerbated by an increase in pollution from intensified cultivation and fertilizer use, as well as urban and industrial expansion. These will worsen wetland health, reducing the capacity of the complex to filter out these very nutrients in a vicious cycle. With nutrient levels in the wetlands already exceeding safe levels, further pollution will likely lead to a tipping point where wetland function declines significantly.

Availability of water for agriculture and domestic use is expected to decrease. Greater abstraction of water for growing agricultural and domestic demand, climate change, and further spread of water hyacinth could all drive a decline in wetland water levels. This would reduce water availability in the wetlands.

Water quality in the system is expected to also decrease. Further increases in nutrient loads from agricultural runoff, sewage, and industrial pollution and increased sedimentation will further compromise water quality in the wetlands. Water may become increasingly unsuitable for livestock and domestic consumption.

Nature-based tourism revenue has been substantially reduced by COVID-19, with a varied likelihood of recovery. As in other regions, the COVID-19 pandemic has had a substantial impact on tourism revenues due to restrictions on international travel. Once the effects of the pandemic ease, only in the Rwandan portion of the wetlands complex is tourism predicted to eventually recover beyond pre-COVID levels by 2050. The regional value of tourism is predicted to increase US$1.7 million by 2050

(32.5 percent increase) under BAU, largely due to the expectation that Akagera National Park will continue to be effectively managed in the future. However, tourism growth is predicted to reach a ceiling by around 2040, due to the limited size of the park and loss of wildlife attractions elsewhere in the wetlands. In contrast, tourism value is predicted to decline by US$57,000 (9 percent of current value) in the Tanzanian portion of the wetlands by 2050, and by US$6,000 (8 percent of current value) in the Burundian portion. This is because tourism is not well developed in these parts of the wetlands, and ongoing population growth and habitat loss mean the attractiveness of these areas for wildlife tourism will not increase under a BAU scenario.

There is scope to try diversify the tourism portfolio of the wetlands beyond the main attraction of Akagera National Park, despite the scarcity of large wildlife outside the park. For example, water-based recreational activities such as fishing and boating could be feasible across much of the study area. This potentially includes parts of the wetland complex in close proximity to Kigali, which is advantageous for visitors who might not have the time to travel to Akagera National Park. Opportunities for viewing waterbirds also exist outside the national park, with species like the charismatic grey-crowned crane said to persist in other parts of the wetlands (Fischer *et al.*, 2011), while remaining papyrus swamps outside protected areas also harbor unique bird species. However, increasing the attractiveness of the rest of the wetland complex to tourists would also require substantial effort in developing accommodation facilities and ensuring that road infrastructure allows for easy access to potential tourism sites. Key informants involved in bird guiding in the area also report that destruction of papyrus swamps is reducing the attractiveness of the Mugesera-Rweru complex for birdwatchers.

Wetland degradation is expected to increase the severity of local and global climate change. Loss of wetland habitat to cultivation and settlement will release carbon stored in vegetation, as well as in the carbon-rich wetland soils. Loss of papyrus swamps in particular will result in a significant reduction in carbon storage, due to the high biomass stocks and rapid growth of the plant, meaning it has high carbon storage and sequestration potential. It was predicted that **carbon storage in the wetlands complex could decline by 1.5 percent (5.9 MtC) by 2050** in a BAU scenario, representing a **loss in value of US$110,000** relative to the current landscape.

The stocks of fish, papyrus, and other resources are expected to decline. Worsening pollution, siltation, sedimentation, and water hyacinth invasion will have a negative impact on fish and other aquatic life, compromising livelihoods of people dependent on these natural resources. Papyrus stocks will also decline due to conversion to agriculture, burning, and over-harvesting. It is estimated that demand for papyrus from communities surrounding the wetlands will increase by 84 percent by 2050 in a BAU scenario, which could have a substantial impact on papyrus stocks, particularly where population densities and demand for papyrus are already high. Increase in demand for fish is predicted to be even greater, with a 113 percent projected increase by 2050.

Table 77. Estimated changes in the value of ecosystem services and water treatment costs by 2050 caused by land use changes under a BAU scenario for the Rweru-Mugesera-Akagera Wetlands. For services with a global value, both total value to the world and value to the East African region only are shown (latter value in parentheses).

ECOSYSTEM SERVICE	CURRENT VALUE (US$)	2050 VALUE (BAU) (US$)	% CHANGE
Nature-based tourism	12.0m (5.3m)	15.9m (7.0m)	+32.5
Carbon storage	7.3b (8.2m)	7.2b (8.1m)	-1.5

CONCLUSIONS, POLICY IMPLICATIONS, AND NEXT STEPS

This study provides a first regional scale assessment of a relatively comprehensive suite of ecosystem services in four priority transboundary wildlife landscapes of the EAC states. This is therefore an important initial contribution to the understanding of the economic benefits provided by the region's natural capital. Using conservative assumptions, the study estimates that within these relatively undeveloped landscapes that still offer significant and viable habitat for wildlife populations, ecosystems generate services of about $300, $500, $700, and $1,500/ha/year on average for the wetland, savanna, plains, and forest landscapes, respectively. Benefits to the different countries also vary, with the national portions of the different landscapes bringing benefits ranging from $260/ha/year for wetlands in Rwanda to $2,700/ha/year for forests in Burundi. The benefits at global scale are orders of magnitude greater than this, with the values ranging from $32,000 to $56,000/ha/year on average for the four landscapes. This difference is largely because of the significant benefit of carbon retention in avoiding increases in future climate change damages around the world. These are ballpark estimates, based on best available information and large-scale, and thus relatively coarse, modelling and assumptions. Nevertheless, they provide a first indication of the potentially very high value of these areas that are already well noted for their conservation importance. Indeed, the total combined value of the wildlife landscapes within each country represent a significant contribution to GDP. In Burundi, the total combined estimated value of wildlife landscapes equates to 5 percent of GDP, in Kenya 3 percent, Rwanda 4 percent, Uganda 9 percent, Tanzania 7 percent, and South Sudan 9 percent.

The wildlife landscapes selected for this study are of international renown as wildlife tourism destinations, and it is largely assumed that their primary value is tourism. However, this assumption puts the landscapes in jeopardy from a policy perspective, since the tourism economy is vulnerable to shocks such the COVID-19 pandemic, regional political instability, or global economic recession. Indeed, tourism values are high, particularly in the Great East African Plains landscape, where tourism is estimated to generate direct benefits on the order of $2.7 billion per year. However, even from a regional perspective, this value is well exceeded by the value of other, less obvious, regulating ecosystem services, particularly erosion control, flow regulation, and carbon sequestration. This is an important finding of the study. Harvested resources also form a large proportion of the local value, particularly for the forest and wetland landscape, but these values are simultaneously a threat if use is unsustainable.

The global benefits of the study areas are significant and have important policy implications. They not only include climate benefits, but also very large values held by global society for the conservation of wild habitats and species, and significant benefits derived by tourists visiting the areas. These positive externalities of the region could be internalized. In fact, this is already occurring to some extent. Global society's willingness to pay for conservation measures to avoid carbon emissions as well as for the conservation of wildlife is partly reflected in donor payments that support a range of conservation activities in the EAC countries. Tourists' consumer surplus, which is their willingness to pay over and above the actual costs incurred, could be further captured through optimal, differentiated pricing systems. High-price, low-volume tourism policies also have the added advantage of being a low-impact form of tourism.

The local benefits are also significant and suggest that these landscapes should be conserved not only for the international benefits they bring, but as an important source of ecosystem services for the region. While the values vary across the landscape depending on geographic and socio-economic context, the average values noted above suggest that the national benefits of conservation action will likely exceed local scale opportunity costs in terms of forfeited small-scale activities. This makes it feasible to introduce stronger policies and actions to ensure the continued protection, integrity, and connectivity of the habitats that remain.

The threats are significant, however. Across the region, population is still growing rapidly, and this poses one of the greatest threats to the future value of these landscapes, along with climate change. Growing numbers of rural poor and increasing wealth as well as population growth in urban areas both potentially put significant pressure on the landscapes, as both rural and urban dwellers expand the land and resource requirements for food and energy production. Population and climate as the primary drivers of the threats against the wildlife landscapes are extremely difficult to change, but countries will potentially need to put policies in place to do so. While a range of measures is already being put in place to increase the sequestration of carbon to mitigate climate change, population pressure can only be tackled indirectly, especially as it remains a controversial subject. At a local level, however, policies should at least be directed toward locating development benefits *away* from important wildlife landscapes, and not at their edges, as had been a tendency in the past. The latter may only exacerbate the pressures on these areas, since people are drawn to opportunities.

Sometimes threats to the unique and precious features of landscapes are caused by improper management decisions, not because of irresponsibility, but rather due to lack of crucial information such as economic value of habitats. To meet demands for burgeoning populations around the landscapes, human activities will continue to cause significant changes to land cover (Martinez-Harms *et al.*, 2015). This study identified important areas in the landscapes for the supply and delivery of ecosystem services. It would therefore be possible to provide governance-based incentives, *e.g.*, designing a land-use development plan that balances multiple private and public values in the landscapes (Goldstein *et al.*, 2012). This needs to be cognizant that competing strategies may affect different ecosystem services differently, hence the importance of valuation.

It is implicit in this report that economic value of wildlife habitats matters even if there is no direct contribution to the GDP, *i.e.*, non-cash values are important to the local and regional economies. For example, under current institutional arrangements, no money changes hands when landscapes sequester carbon or are important in water quality amelioration. These services benefit the people of East Africa and could be replaced by built infrastructure at considerable costs. This situation is a classic example of a positive environmental externality where the landscapes provide a host of regulatory services that support livelihoods and businesses. In this respect, 1) private parties may thus under-invest in environmental protection because they do not realize the benefits of that investment, and 2) governments may sanction a development project in anticipation of additional tax revenues, *i.e.*, direct contribution to GDP, while habitats in natural state produce only benefits outside the market economy, given that no one has to pay to receive them.

Notably, several economic tools have been developed in response to such externalities and may be enjoined in policy decisions (*e.g.*, payment for ecosystem services, REDD+, biodiversity-relevant taxes, green mutual funds, etc.). The high cost of environmental protection has traditionally been borne by governments and NGOs. The private sector can also contribute to ecosystem conservation and restoration through social investment and philanthropy, and innovatively, by steering the day-to-day investment decisions of actors and financial institutions toward halting biodiversity loss, restoring and conserving natural resources, and promoting the sustainable use of natural resources. This can be achieved through a number of emerging financing tools such as taxation incentives, biodiversity offsets, debt-for-nature swaps, and green and blue bonds (IUCN ESARO, 2020).

Given the high stakes, very careful consideration of conservation policies and the measures to achieve them is required. The next steps of the study will therefore involve the investigation of feasible policy interventions that will be effective in retaining the biodiversity and economic value of these wildlife landscapes.

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APPENDIX 1: SELECTION OF STUDY REGIONS

The process for assessing and prioritizing the transboundary landscapes for inclusion in this study was two-tiered: first, at the inception workshop (in Nairobi on February 11-12, 2020) with valuable input from partner state delegates, and second, by the technical team based on political, economic, and environmental considerations.

The partner states noted the following considerations during the workshop:

- Ecological importance to each EAC country;
- Possible economic offshoots from improving the habitat linkages;
- Relevance of study to upcoming decisions related to landscape management;
- The need to take into account 17 transboundary conservation areas listed and prioritized according to importance to each country; and
- Matching prioritization by multiple countries across the political boundary.

The technical team made the following considerations after the workshop:

- Need for habitat linkages and connectivity across political boundaries, given the level of landscape fragmentations;
- Equitable distribution of focused study areas covered among the six EAC countries;
- Different biomes represented (*e.g.*, rainforests, grasslands, etc.); and
- Scale of analysis required (*e.g.*, type and amount of data available).

From this exercise, seven transboundary protected areas were identified and put forward for economic valuation. These were prioritized by each partner state according to their value or significance with respect to conservation and economic development needs. Each of these different areas and land types contributes differing values (goods and services) to these transboundary areas and their associated communities and nations.

The maintenance of wildlife corridors and broader ecological connectivity as part of the natural landscape within the EAC is critical in ensuring the persistence of a given suite of wildlife species and habitats in the region. A large component of the tourism sector, and the economic development that it drives, is dependent on this. It was therefore imperative to look at protected areas not in isolation, but as part of the larger wildlife habitat landscape, where they play a critical role in terms of natural landscape connectivity. Since transboundary protected areas in the EAC are comprised of different fragmented wildlife habitat units, it is plausible to consider historic natural landscapes for economic valuation. Undertaking an economic valuation of the natural capital of the historic landscapes in which native habitat remnants persist enables us to better inform and direct policies and management strategies in these broader regions toward more sustainable outcomes. These include swathes of historic forests, grasslands, etc. to increase conservation and socioeconomic potential for rare, endangered, or diverse wildlife species. Notably, the seven priority transboundary protected areas identified during the workshop occur in different landscape types with differing land-cover/land-use types and biophysical conditions. In consideration of the above, the technical team proposed the use of a landscape approach that improves habitat connectivity. Thus, several transboundary protected areas were combined to provide appropriate scale for economic valuation of wildlife and their habitats.

In selecting the four transboundary landscapes (Western Forests, Eastern Plains, Northern Savannas, and Wetlands) for economic valuation, the following were considered:

- Political considerations:
  - Equity: the proposed landscapes encompass at least two key protected area systems in each country; and
  - Inclusivity: all partner states are included. For each country, at least one of their priority protected area systems is included.
- Economic considerations:
  - Taking a broader landscape-scale approach facilitates better understanding of the economic value of a larger, broader region and the country components of this value. Analysis at this scale provides values that have a better regional resonance.
- Ecological considerations:
  - The spatial resolution of much of the ecological, topographic, and climate data are better attuned to working at a broader scale, compared with working at the scale of individual protected areas;
  - Extended areas are better able to incorporate known national and global conservation priorities, *e.g.*, global biodiversity hotspots and ecoregions;
  - Larger areas are better for examining trends in land-use change and underlying drivers (*e.g.*, the conversion of land to cultivation, urbanization, etc.), as well as assessing policies enabling the maintenance of landscape-level connectivity; and
  - The four priority landscapes represent four distinct and unique biomes: grasslands, savanna, forests, and wetlands. All of these are of importance to the East African Community collectively.

APPENDIX 2: GROUPING OF HABITAT TYPES

A detailed analysis was undertaken at a regional (East Africa) scale of spatial data on vegetation types, ecoregions, land cover, and biomass, in conjunction with a review of relevant ecological literature. To get an as-accurate-as-possible representation of the habitat across East Africa, Copernicus 100 m land cover data for 2018, the potential natural vegetation map for eastern and southern Africa, and change in the NDVI between 2001 and 2018 from Trends.Earth were combined to generate a single map of habitat types to be used throughout the baseline assessment. The final classification comprised 72 habitat types across all regions, which includes a degraded and undegraded form of each natural habitat type where relevant (Table 78). There are 16 forest habitat types, 14 woodland habitat types, 12 grassland/wooded grassland habitat types, 19 bushland/shrubland habitat types, 5 aquatic habitat types, 4 desert/bare habitat types, and 2 anthropogenic types (cultivation, built-up). These 72 habitat types were then grouped more broadly for mapping purposes (Table 78).

Table 78. The 72 habitat types and their broader groupings for the East Africa region mapped using a combination of land cover, vegetation, and NDVI data

HABITAT TYPE	GROUP
Afromontane Forest	Afromontane Forest
Degraded Afromontane Forest	Afromontane Forest
Afromontane Forest with Bamboo	Afromontane Forest
Degraded Afromontane Forest with Bamboo	Afromontane Forest
Bamboo	Bamboo
Degraded Bamboo	Bamboo
Lake Victoria Forest	Afromontane-Lowland Forest
Degraded Lake Victoria Forest	Afromontane-Lowland Forest
Swamp Forest	Mixed / Scrub Forest
Degraded Swamp Forest	Mixed / Scrub Forest
Scrub Forest	Mixed / Scrub Forest
Degraded Scrub Forest	Mixed / Scrub Forest
Coastal Mosaic	Mixed / Scrub Forest
Degraded Coastal Mosaic	Mixed / Scrub Forest
Zanzibar Rainforest	Lowland Forest
Degraded Zanzibar Rainforest	Lowland Forest
Woodland/Degraded Afromontane Forest	Afromontane Forest
***Acacia-Commiphora* Woodland**	Woodland

HABITAT TYPE	GROUP
Degraded *Acacia-Commiphora* Woodland	Woodland
Butyrospermum Woodland	Woodland
Dry Miombo	Woodland
Degraded Dry Miombo	Woodland
Wet Miombo	Woodland
Degraded Wet Miombo	Woodland
Combretum Woodland	Woodland
Degraded Combretum Woodland	Woodland
Woodland	Woodland
Degraded Woodland	Woodland
Closed Riverine Woodland	Woodland
Open Riverine Woodland	Woodland
***Acacia-Commiphora* Wooded Grassland**	Wooded Grassland
Degraded *Acacia-Commiphora* Wooded Grassland	Wooded Grassland
Butyrospermum Wooded Grassland	Wooded Grassland
Degraded Butyrospermum Wooded Grassland	Wooded Grassland
Combretum Wooded Grassland	Wooded Grassland
Degraded Combretum Wooded Grassland	Wooded Grassland
Palm Wooded Grassland	Wooded Grassland
Degraded Palm Wooded Grassland	Wooded Grassland
Wooded Grassland	Wooded Grassland
Degraded Wooded Grassland	Wooded Grassland
***Acacia-Commiphora* Deciduous Bushland**	Dry Bushland
Degraded *Acacia-Commiphora* Deciduous Bushland	Dry Bushland
Dense *Acacia-Commiphora* Deciduous Bushland	Dry Bushland
Degraded Dense *Acacia-Commiphora* Deciduous Bushland	Dry Bushland
Dense Evergreen Bushland	Moist Bushland
Degraded Dense Evergreen Bushland	Moist Bushland
Open Evergreen Bushland	Moist Bushland

HABITAT TYPE	GROUP
Degraded Open Evergreen Bushland	Moist Bushland
Semi-Desert Shrubland	Semi-Desert Shrubland
Degraded Semi-Desert Shrubland	Semi-Desert Shrubland
Desert Shrubland	Semi-Desert Shrubland
Degraded Desert Shrubland	Semi-Desert Shrubland
Itigi Thicket	Dry Bushland
Degraded Itigi Thicket	Dry Bushland
Herbaceous Vegetation	Herbaceous Vegetation
Degraded Herbaceous Vegetation	Herbaceous Vegetation
Halophytic Vegetation	Halophytic Vegetation
Degraded Halophytic Vegetation	Halophytic Vegetation
Ericaceous Vegetation	Ericaceous-Heath Vegetation
Afroalpine Vegetation	Ericaceous-Heath Vegetation
Freshwater Lake	Freshwater Lake
Salt Lake	Salt Lake
Wetland/Swamp	Wetland/Swamp
Degraded Wetland/Swamp	Wetland/Swamp
Mangroves	Mangroves
Degraded Mangroves	Mangroves
Bare/Sparse Vegetation	Bare/Sparse Vegetation
Snow/Ice	Snow/Ice
Desert	Desert
Open sea	Open sea
Built-up/Urban	Built-up/Urban
Agriculture	Agriculture

APPENDIX 3: VALUATION OF HYDROLOGICALLY LINKED SERVICES

INTRODUCTION

The hydrological services of wildlife landscapes were estimated by modeling the rates of sediment delivery over the landscape and comparing this to the outputs obtained from a hypothetical landscape devoid of vegetative cover using the InVEST 3.8.7. As inputs, the Copernicus Global Land Operations 2018 land cover map and the HydroSHEDS hydrologically conditioned global digital elevation model (DEM) (Lehner *et al.*, 2008), which has a resolution of 90 m, were used. The watershed and sub-watershed boundaries were derived from the global HydroBASINS dataset of watershed and sub-watershed boundaries (Lehner & Grill, 2013). In delineating regions for modeling, we ensured that all watersheds encompassed by the wildlife landscapes were included, as well as adjacent watersheds that might benefit from the hydrological services provided by the wildlife landscapes.

Although a 30 m global DEM data is available, an advantage of the HydroSHEDS DEM is that it has been hydrologically conditioned through a sequence of procedures including sink filling, stream burning, molding of valley courses, and "seeding" to ensure natural sinks, such as endorheic (closed) basins, are retained (Lehner, Verdin & Jarvis, 2006; Tavares da Costa, Mazzoli & Bagli, 2019). This was particularly advantageous in the Great Eastern Plains study region, where use of other DEMs resulted in the treatment of endorheic basins as hydrological sinks by flow calculation algorithms, resulting in inaccurate results. For the purposes of hydrological modeling, study regions were delineated by watershed and sub-watershed boundaries, which were derived from the global HydroBASINS dataset of watershed and sub-watershed boundaries (Lehner & Grill, 2013). In delineating regions for modeling, we ensured that all watersheds encompassed by the wildlife landscapes were included, as well as adjacent watersheds that might benefit from the hydrological services provided by the wildlife landscapes. Descriptions of the individual models, and the additional datasets used in each case, follow below.

FLOW REGULATION

We used the InVEST 3.8.7 seasonal water yield model to estimate the contribution of wildlife habitats to maintaining baseflow through their influence on the infiltration of rainfall into groundwater flows that ultimately reach surface springs and stream flows. The model estimates the runoff associated with rainfall events (called quickflow), water losses to evapotranspiration, and local recharge that contributes to groundwater storage and baseflow, which helps to sustain river flows during the dry season.

Quickflow is calculated using a curve number (CN)-based approach. Soil and land cover properties determine what proportion of rainfall runs off the land (contributing to quickflow) and what proportion infiltrates into the soil (contributing to local recharge). The curve number approach is a simple way of capturing these soil and land cover properties, with higher CN values indicative of greater runoff potential, while low CN values indicate greater likelihood of infiltration. The soil property component of the curve number calculation involves assigning soils to one of four hydrologic groups according to runoff potential. For this, we downloaded the HYSOGs250 m global raster of hydrologic soil groups (Ross *et al.*, 2018). A curve number must then be assigned to each land cover class and hydrologic soil group combination. Such estimates are scarce for the region, so our study drew on curve numbers from a few East African studies (Baker & Miller, 2013; Beatty *et al.*, 2018; Bagstad *et al.*, 2020), as well as from the United States where the CN-based approach has been most widely used (Natural Resources Conservation Service - USDA, 2004).

In addition to soil and land cover data, the model requires various forms of precipitation data for quickflow calculation. Total monthly precipitation for the region was obtained from World Clim at 1 km resolution (Fick & Hijmans, 2017). In addition to the amount of rainfall per month, the model requires the number of wet days per month. Data on precipitation days per month was obtained from the Climate Research Unit gridded climatology of monthly means from 1961-1990 (New *et al.*, 2002). Although relatively old, this was the most recent dataset of average wet days over an extended time period that we could find for the region. To account for variation in monthly wet days within our study regions, we followed (Bagstad *et al.*, 2020) in using WWF ecoregions (Olson *et al.*, 2001) as a climate zone proxy. This involved calculating the average wet days per month for each ecoregion in our study areas. The contribution of each pixel to quickflow can then be calculated for each pixel based on curve numbers and the amount and frequency of precipitation. For pixels in streams, the model set quickflow as the value of precipitation on that pixel, based on the assumption that only runoff, and no infiltration, occurs in streams. Monthly quickflow values were then summed to provide an estimate of annual quickflow.

Once quickflow has been calculated, the model can move on to estimation of local recharge. Precipitation that does not run off as quickflow or get lost through evapotranspiration can infiltrate the soil to produce local recharge. Hence, in addition to quickflow, information on reference evapotranspiration, and the water requirements of vegetation on each pixel are needed for estimation of local recharge. Monthly reference evapotranspiration measures the energy (expressed as a depth of water *e.g.,* mm) supplied by the sun (and occasionally wind) to vaporize water. Monthly reference evapotranspiration data at 1 km resolution were downloaded from the CGIAR-CSI database. The water requirements of different vegetation/land cover types are measured by the plant evapotranspiration coefficient (Kc). The crop factor represents the ratio of evapotranspiration for a given land cover class to that of a reference crop (alfalfa). Once again, local estimates of crop factors are scarce, necessitating reliance on a few studies from the region, along with comparable estimates from beyond East Africa (Descheemaeker *et al.*, 2011; Beatty *et al.*, 2018; Bagstad *et al.*, 2020). Actual evapotranspiration can then be calculated from the combination of reference evapotranspiration and plant evapotranspiration coefficients, along with the fraction of annual upslope recharge that is available in each month (α). The α parameter is a function of rainfall seasonality. Following the InVEST user guide, this was set as the antecedent monthly precipitation value, as a proportion of total annual precipitation: $P_{month-1} / P_{annual}$. Finally, a value for local recharge for each pixel is obtained by subtracting quickflow and actual evapotranspiration from precipitation.

Once local recharge has been calculated, the model can work out the contribution of each pixel to baseflow, *i.e.,* water that reaches a stream. Pixels with a negative value for local recharge (evapotranspiration exceeds available water) do not contribute to baseflow and are thus assigned a value of 0. For pixels that do contribute to local recharge, baseflow is a function of the amount of local recharge from a pixel that actually reaches a stream. In addition to each pixel's contribution to baseflow, the model outputs a map of total baseflow flowing through each pixel, contributed by all upslope pixels.

Following Bagstad *et al.* (2020), we compared actual evapotranspiration predicted by our model to annual actual evapotranspiration from SSEBop Evapotranspiration Products (USGS, 2021) This allowed us to evaluate our crop coefficient estimates of evapotranspiration from different land cover classes in our study regions. Where predicted evapotranspiration was lower than evapotranspiration from the empirical data, we increased crop coefficient values for subsequent model runs, and vice versa where model-predicted evapotranspiration was higher than empirical evapotranspiration data.

Flow regulation by ecosystems decreases the seasonal variation in flows by slowing down water flows through the landscape and contributes to river base flows during the dry season. This reduces the size of reservoirs needed to meet water demands, as well as affecting the availability of water to people who draw their water directly from streams. Ideally, this service should be valued by analyzing long time series of flow estimates at strategic points in the landscape. The models capable of doing this, such as SWAT, are dynamic models that are comparatively complex to set up, especially at large spatial scales (see for example Turpie *et al*., 2020). In this study, valuation of the flow regulation service was based on the InVEST output of estimated per-pixel contribution to baseflow through groundwater recharge, expressed in m^3 per hectare and summarised in terms of Mm^3 storage per sub-catchment. This was taken to be a reasonable proxy for the saving in storage infrastructure that would be needed in the absence of the service.

EROSION CONTROL

We used the InVEST sediment delivery ratio (SDR) mode, which estimated sediment retention and export through combining estimated soil loss with a connectivity index. The SDR model estimated sediment retention and export through combining estimated soil loss with a connectivity index. The revised universal soil loss equation (RUSLE) was used to calculate soil loss, computed as follows:

$$USLE = R \times K \times LS \times C \times P$$

where USLE is the potential average annual soil loss, R is the rainfall erosivity factor (MJ mm/ha/h/yr), K is the soil erodibility factor (t ha hr/MJ/ha/mm), LS is a factor of slope length and steepness, C is the land cover management factor, and P is the supporting practice factor.

For rainfall erosivity (R), we used the Global Rainfall Erosivity Database (GloREDa) (Panagos *et al*., 2017), which provides a global map of rainfall erosivity at 250 m resolution. Soil erodibility (K) is a function of various intrinsic topsoil (0-30 cm) properties, including texture, organic matter content, structure, and permeability. These data were obtained from the International Soil Reference and Information Centre (ISRIC) SoilGrids database at 250 m resolution (de Sousa *et al*., 2020). Using these properties, K was calculated using the nomograph proposed by Wischmeier & Smith (1978):

$$K = \frac{2.1 \times 10^{-4}(12 - OM)M^{1.14} + 3.25(s-2) + 2.5(p-3)}{759}$$

where M is a parameter linked to particle size, OM is organic matter content (%), s is a soil structure class and p is a soil permeability class. M is calculated from the proportional contribution of different soil particle sizes as follows: M = (%silt + %very fine sand) / (100 - %clay). As the ISRIC data do not have a very fine sand fraction layer, we estimated very fine sand to be 20 percent of the sand fraction, following Panagos *et al.*, (2014) and Fenta *et al.*, (2020). OM is organic matter content (%), which was obtained from the soil organic carbon (%) layer through multiplication by a conversion factor of 1.72. S is the soil structure factor, and was assigned on the basis of soil texture classes following the scheme used by

Bagarello *et al.*, (2009). P is the soil permeability factor, which was also assigned according to soil texture classes following the classification scheme used by Panagos *et al.* (2014).

The land cover management (C) component of the RUSLE equation accounts for how different land cover types affect soil erosion relative to bare fallow areas (Wischmeier & Smith, 1978). A cover management value was thus assigned to each of the land cover classes in our land cover dataset, ranging from low values for dense natural vegetation (*e.g.*, closed forest) indicating high protection from erosion, to high values for cropland indicating lower protection from erosion. These values were obtained through consultation of a range of studies from the East African region and beyond (Angima *et al.*, 2003; Leh *et al.*, 2013; Hamel *et al.*, 2017; Bagstad *et al.*, 2020; Fenta *et al.*, 2020).

The support practice (P) factor in the RUSLE equation is primarily relevant to agriculture lands. It indicates the ratio of soil loss after implementation of structural soil conservation measures to soil loss from straight-row cultivation running up and down a slope (Wischmeier & Smith, 1978). At the regional-scale, conservation structures cannot be accurately mapped at this stage, making estimates of P-factor problematic in large-scale erosion assessments (Fenta *et al.*, 2020). In the absence of adequate information, some East African studies simply use a P-factor value of 1 for agricultural land (*e.g.*, Beatty *et al.*, 2018; Fenta *et al.*, 2020), arguing that erosion control measures are too sparse to make a notable difference to erosion at the regional scale. However, Fenta *et al.* (2020) note that contour tilling is a widely practiced erosion control measure across East Africa. Hence, we chose to use a moderate P-value factor for agricultural land across the region, derived from consultation of other East African studies (Karamage *et al.*, 2017; Bagstad *et al.*, 2020; Fenta *et al.*, 2020).

The LS component of the RUSLE equation was calculated by InVEST, as a function of slope length and gradient.

The RUSLE component of the SDR model thus provided an estimate of soil loss for each pixel. Sediment export and retention could then be modelled by combining this measure of soil loss with the sediment delivery ratio. The SDR is derived from a connectivity index, which measures the hydrological linkage between sources of sediments and sinks (streams), and is a function of both the area upstream of a given pixel and of the flow path between the pixel and the nearest stream. Thus, each pixel's connectivity index value varies according to vegetation cover (C-factor of the USLE), slope, and the size of the upslope contributing area. Finally, sediment export from a given pixel to streams was calculated as the product of soil loss (from the RUSLE component) and the sediment delivery ratio. Pixel level values for sediment export can then be aggregated to the watershed or sub-watershed scale, allowing for comparison with empirical data of sediment loads in catchments where this is available.

Finally, the model calculated sediment retention as a measure of avoided soil loss for each pixel. This was produced through subtracting sediment export under current land use from the sediment loss that would occur if all land cover were converted to bare ground *i.e.,* maximum potential soil loss (RKLS). In other words, sediment retained = RKLS – USLE. RKLS for each pixel was calculated by removing the land cover management (C) and support practice (P) factors from the RUSLE.

Valuation of the service was based on the soil loss avoided for each one-hectare pixel, in m^3 per year. The avoided sedimentation of downstream rivers, reservoirs, lakes, estuaries, and coral reefs was assumed to be fully demanded, and was valued using a replacement cost, being the construction and maintenance of sediment check-dams where the construction of sediment check dams were priced at US$1.24 per m^3 (Mekonnen *et al.* 2015).

WATER QUALITY AMELIORATION

Nutrient-enriched runoff from agricultural and urban land can have a negative impact on the water quality of downstream aquatic ecosystems. The excess nutrients introduced to these systems can change their trophic status in a process known as eutrophication. This is usually accompanied by increased abundance of algae and plant growth, which changes the nature and composition of these systems and affecting the benefits that can be derived from them. At extremes, it can lead to toxic algal blooms, loss of dissolved oxygen, and fish kills. Still water bodies, such as reservoirs and lakes, are particularly susceptible to this type of degradation. Where water is collected or extracted for drinking water supply, the elevated levels of algae, as well as nutrients and suspended sediments, increases the costs of water treatment. Natural vegetation can help mitigate these problems. Some of the nutrients in nutrient-enriched runoff can be removed when it passes through natural vegetation and wetlands in the landscape, ameliorating the pollution problem before it reaches downstream ecosystems and locations where water is abstracted for use. This is the *active* aspect of the service, in that ecosystems remove pollution through ecological process such as vegetative growth. The capacity to perform this active service will be linked to the characteristics and condition of the ecosystem, and the use of the service will depend on the amount of anthropogenic activity upstream of the ecosystem. In addition, retaining ecosystems in their natural state, as opposed to replacing them with alternative uses such as agriculture or human settlements, usually maintains a higher quality of water leaving the area than if the transformation took place. This is the *passive* aspect of the service. The demand for the active and passive services comes from the users of water downstream of these ecosystems. Together, the active and passive services are valued as the costs avoided as a result of retaining the ecosystem in its natural condition.

The water quality amelioration service was estimated using the InVEST 3.8.7 nutrient delivery ratio (NDR) model. This uses a nutrient mass balance approach to quantify nutrient export to downstream aquatic systems. To do so, it combines measures of nutrient input across the landscape, with land-cover-specific retention and connectivity properties of pixels belonging to the same downstream flow path. The same base layers for land cover, elevation, and watersheds were used as described above for the sediment retention model. However, the NDR model also requires a number of specific inputs, as described below.

To map nutrient inputs, a nutrient load value was assigned to each pixel in the landscape. This required rates of nitrogen and phosphorous outputs in kg/ha/year to be assigned to each land cover class, including both anthropogenic and natural classes. However, such estimates are scarce for the region. Nutrient loads for agricultural lands can be derived from estimates of organic and inorganic fertilizer application. In addition to shortages of data, high variation in fertilizer use both across and within our study areas added a further challenge. To account for these differences, agricultural land cover was reclassified by country, allowing us to set country-specific nutrient loads for agricultural land according to the available literature (Freeman & Omiti, 2003; Ariga *et al.*, 2006; Namaazi, 2008; Leh *et al.*, 2013; Bagstad *et al.*, 2020). In contrast, nutrient loads for natural land cover classes were kept constant per land cover class across the region, in the absence of sufficient local data to justify variation by area (Leh *et al.*, 2013; Bagstad *et al.*, 2020). Each pixel's land-cover specific nutrient load value was then modified to account for local runoff potential. This required a runoff proxy, for which we used mean annual precipitation data from 1970-2000 at 1 km resolution from World Clim (Fick & Hijmans, 2017).

The next step involved calculation of nutrient delivery, which is how much of the annual nutrient load reaches watercourses by emulating the movement of nutrients across the landscape. This involved calculation of the NDR, which is a function of a) the ability of downstream pixels to retain nutrients and b) an index of hydrological connectivity. Estimation of nutrient retention (the *active* service) required a nutrient retention coefficient to be assigned to each land cover class, which varied between 0 (no retention) and 1 (all nutrients retained). Nutrient retention is generally higher for natural vegetation types than cultivated land, while urban areas have very low values for nutrient retention. Once again, we had to draw on a limited number of studies for estimates of retention efficiency of different land cover classes in the region and comparable areas (Leh *et al.*, 2013; Redhead *et al.*, 2018; Bagstad *et al.*, 2020). The model also required an estimate of critical flow length (the distance at which the maximum nutrient retention efficiency is reached) for each land cover class. This was conservatively set to the resolution of the DEM (90 m) for all land cover classes. In general, the literature suggests that the value is smaller than 90m.

Finally, nutrient export from each pixel was calculated as the product of nutrient loads and the nutrient delivery ratio. The final model output was the mass of nutrients from each pixel that eventually reaches a stream. This was aggregated to the watershed and sub-watershed level to give estimates of nutrient export at the desired scale. In order to estimate the service, the model was run for the baseline (2018) land cover as well as for a hypothetical land cover in which the wildlife habitats were replaced by small-scale agriculture as the next most likely land use. The difference in nutrient outputs was taken to be the result of the passive and active services combined.

The InVEST model was limited in terms of estimating the role of wetlands in the landscape, as it does not model any instream nutrient retention processes. In other words, InVEST does not model any further retention once nutrients reach the Akagera River or other stream pixels. However, InVEST was still of use for estimating the nutrient loads entering the wetland from contributing catchment areas. Hence, for the Rweru-Mugesera-Akagera Wetlands, the nutrient loads entering the lake under baseline conditions were estimated using the InVEST model, while the uptake of the nutrients was estimated based on information from the literature. Kansiime & Nalubega (1999) showed that uptake rates of total nitrogen by the functional papyrus areas of the Nakivubo Wetland, Uganda, were on the order of 475 kg N/ha/year, and total phosphorus reduction (through sedimentation and uptake) was on the order of 77 kg P/ha/year. *Miscanthidium violaceum* mats had lower rates of reduction of both phosphorus and nitrogen nutrients than *C. papyrus*. In the case of the Rweru-Mugesera-Akagera Wetlands, we applied the above P uptake rates to the area of healthy swamp vegetation and assumed a rate of one-third of this for the degraded vegetated swamp areas.

The primary demand for the service was assumed to be for catchments that drained into reservoirs or lakes that are used for water supply. In these cases, we valued the services from those catchments in terms of the water treatment cost savings.

In addition, we recognized that the eutrophication of natural lakes, wetlands, and estuaries would also have an impact on the supply of ecosystem services from those systems. These actual avoided impacts could not be estimated within the scope of this study. Therefore, for the catchments performing a service for important downstream aquatic ecosystems, we assumed that the service was fully demanded.

We used the cost of constructing artificial wetlands to filter water before entering these ecosystems as the replacement cost of the service.

Treatment wetlands are purpose-built systems that are designed to maximise the rate of nutrient uptake through adequate throughflow and aeration of water, the cutting and disposal of plant material to stimulate growth, and the dredging and disposal of contaminated sediment (Armitage et al., 2013). Construction involves acquiring land, excavation, planting soil, pipelines, vegetation, and on-site work (Kadlec & Wallace, 2009, Gunes *et al*., 2011). We used an estimated total phosphorus removal rate by treatment wetlands of 191 ± 9 kg/ha/year (Kyambadde *et al.,* 2005) to calculate the equivalent size of artificial wetlands that would be needed to replace the estimated service for the above-mentioned catchments and the Rweru-Mugesera-Akagera Wetlands.

There are a range of cost estimates for the construction and maintenance of treatment wetlands in the literature (Table 79), but most are for small wetlands and few examine the effects of scale. We based our estimates on Lloyd *et al.* (2002), who derived a model to estimate construction and maintenance costs as follows:

Construction cost (2002 Aus $) = 343,913 x Ln * (surface treatment area in ha) + 738,607

Maintenance costs (2002 Aus $) = 9842.20*(surface treatment area in ha)^0.4303

Table 79. Estimated construction and design costs for constructed treatment wetlands typically treating municipal wastewater

SOURCE	GROUP	COST
Leinster 2004	Small-scale wetland	Aus $90 - $100 per m^2
Leinster 2004	Large-scale wetland (reticulated lake)	Aus $65 per m^2
Hunter 2003	Large wetland	Aus $500,000 per ha
Weber 2002	Standard constructed wetland	Aus $3,400 – $17,900 per ha
Walsh 2001	Greenfields wetland	Aus $120,000 per ha
Lane 2004	Standard constructed wetland	Aus $75 per m^2
Stewart 2005	Horizontal Flow (HF) wetland	US $86 per m^2 ($74 -97)
Dzikiewicz 1996	HF wetland	€31 per m^2 (€10 – 83)
Rousseau *et al.* 2004	HF wetland	€257 per m^2 (€237 – 277)
IRIDRA 2002	HF wetland	€125 per m^2 (€38 – 247)
Masi *et al.* 2006	HF wetland	€115 per m^2 (€101 – 129)
Steiner & Combs 1993	HF wetland	€74 per m^2 (€27 - 144)
Billore *et al.* 1999	HF wetland	€29 per m^2

SOURCE	GROUP	COST
Platzer *et al.* 2002	HF wetland	\$61 per m^2 (\$22 - 229)
U.S. EPA 2000	HF wetland	\$67 per m^2 (\$32 - 125)
Dallas *et al.* 2004	HF wetland	\$33 per m^2
De Morais *et al.* 2003	HF wetland	€96 per m^2
Shrestha *et al.* 2001	HF wetland	\$31 – 72 per m^2

Source: extracted from reviews by Taylor 2005, Silva & Bragga 2006, Gunes *et al*. 2011, La Notte *et al.* 2012

APPENDIX 4: VALUATION OF HARVESTED WILD RESOURCES

OVERVIEW

For the purposes of this study and based on the nature of the data, the resources were grouped as shown in Table 80. A four-step approach was used to quantify and value subsistence harvesting of wild resources:

- Estimate supply (*e.g.*, kg/ha/y) based on specific land cover/vegetation characteristics;
- Estimate household demand (*e.g.*, kg/household/y) based on household characteristics;
- Estimate and map actual harvesting (*i.e.*, use) using spatial modeling of household use and data on production; and
- Value the actual harvests based on market prices and sustainability assessment.

Each of these steps is described in detail below.

Table 80. Wild biomass groupings based on the CICES framework

	PURPOSE	GROUP
Wild plant resources	Nutrition and health	Wild plant foods and medicines
	Energy	Wood fuel
	Raw materials	Grass
		Reeds and sedges
		Palm leaves
		Poles and withies
		Timber
		Bamboo
Wild animal resources	Nutrition	Terrestrial birds and animals
		Honey
		Fish

DATA SOURCES

Data were collated on the demand for different resources by households, the stocks and yields of these resources in the different habitat types of each study area, and the spatial distribution and characteristics of households in each study area.

Very little of the harvesting of wild natural resources is monitored in any of the six EAC countries. Therefore, this estimation was based on ecological and socio-economic studies that have taken place across the East Africa region and in other areas with similar socio-economic and ecological characteristics. Available information on system yields, quantities harvested, harvesting costs, and market prices for different resource types were obtained from the literature, using information from the study area as far as possible (Table 81). Where data were limited, information from comparable socio-ecological systems in adjacent countries was used. The quality of each study was also taken into consideration in deciding whether findings should be used in devising assumptions.

Table 81. Information sources consulted for estimating availability and use of natural resources in each of the four study regions

NATURAL RESOURCE	REFERENCES	DATA
Albertine Rift Forests Wildlife Landscape		
Firewood, charcoal, poles, timber	Howard 1991; Ndayambaje 2002; Hatfield & Malleret-King 2007; Rwamahe 2008; Hartter 2010; NISR 2012; Drigo *et al.* 2013; Nahayo, Ekise & Niyigena 2013; European Commission 2014; Nyamuyenzi 2015; Harrison *et al.* 2015; Mwageni, Shemdoe & Kiunsi 2015; Bitariho, Sheil & Eilu 2016; Ntiranyibagira *et al.* 2017; UBOS 2018; Gianvenuti & Vyamana 2018	Household participation, average household harvesting rates, prices
	Rutherford, 1978; Chidumayo, 1988; Grundy *et al.*, 1993; Drichi, 2002; Malimbwi *et al.*, 2005; Maliondo *et al.*, 2005; Barnes *et al.*, 2010; Kebede *et al.*, 2013; Nyirambangutse *et al.*, 2017; Sisay *et al.*, 2017	Above-ground biomass, basal area, woody volume m^3/ha, mean annual increment for different vegetation types and land uses, proportion of woody biomass available for different uses
Wild plant foods and medicines	Kanabahita 2001; Rwamahe 2008; Hartter 2010; Nahayo *et al.* 2013; Ndayambaje 2013; Harrison *et al.* 2015	Household participation, average household harvesting rates, prices
	Campbell, 1987; Campbell, Luckert & Scoones, 1997	Annual productivity of wild plant foods
Wild animal resources	Ndayambaje, 2002; Hatfield & Malleret-King, 2007; Rwamahe, 2008; Harrison *et al.*, 2015	Household participation, average household harvesting rates, prices of honey
	Hill, Osborn & Plumptre, 2002; Ndayambaje, 2002; Rwamahe, 2008; Harrison, 2013; Twinamatsiko *et al.*, 2014	Household participation, average household harvesting rates, prices of bushmeat
	Winston, 1992; Kajobe & Roubik, 2006; Jaffé *et al.*, 2010; Garcia *et al.*, 2013	Bee colony density, honey production
	Prins & Reitsma, 1989; Wirminghaus & Perrin, 1993; White, 1994; Plumptre & Harris, 1995; Monadjem, 1997; Caro, 2008	Biomass density of species hunted for bushmeat
	Marks, 1973; Blumenschine & Caro, 1986; Holmern *et al.*, 2006	Estimates of edible meat across species
Bamboo, grasses, reeds	Cunningham, 1996; Bitariho & Mosango, 2005; Bitariho & Emmanuel, 2019	Household participation, number of handicraft products per household
	Bitariho & Mosango, 2005	Number of bamboo stems needed, longevity of various bamboo products to estimate annual stem demand

NATURAL RESOURCE	REFERENCES	DATA
	Scott, 1998; Bitariho & Mosango, 2005; Sheil *et al.*, 2012; Bitariho & Ssali, 2013	Stem density estimates for bamboo
	Rwamahe 2008; NISR 2012; UBOS 2018	Household participation, average household harvesting rates of thatching grass
	Bourlière & Hadley, 1970; Chidumayo, 1997; Turpie *et al.*, 1999; Shirima *et al.*, 2015	Annual productivity of thatching grass
	Thompson, Shewry & Woolhouse, 1979; Cunningham, 1985; Jones & Muthuri, 1985; Chale, 1987; Jones MB & Muthuri FM, 1997; Mnaya *et al.*, 2007; Jones, Kansiime & Saunders, 2018	Annual productivity of reeds and sedges
Great East African Plains Wildlife Landscape		
Firewood, charcoal, poles, timber	Hosier 1984; Jensen 1984; Mung'ala & Openshaw 1984; Chamshama, S.A.O. Kerkhof & Singunda 1989; Emerton 1996; Biran, Abbot & Mace 2004; Wiskerke *et al.* 2010; Giliba *et al.* 2010; Schmitt 2010; Drigo *et al.* 2015; Bär & Ehrensperger 2018	Household participation, average household harvesting rates, prices
	Rutherford, 1978; Mung'ala & Openshaw, 1984; Chidumayo, 1988, 1993; Malimbwi *et al.*, 2005; Maliondo *et al.*, 2005; Barnes *et al.*, 2010; Sisay *et al.*, 2017	Above-ground biomass, basal area, woody volume m^3/ha, mean annual increment for different vegetation types and land uses, proportion of woody biomass available for different uses
Wild plant foods and medicines	Emerton 1996; Odera 1997; Walter 2001; Oiye *et al.* 2009; Kenya Water Towers Agency 2018	Household participation, average household harvesting rates, prices
	Campbell, 1987; Campbell *et al.*, 1997; Assefa & Abebe, 2011	Annual productivity of wild plant foods
Wild animal resources	Emerton 1996; Giliba *et al.* 2010; Schmitt 2010; Kenya Water Towers Agency 2018	Household participation, average household harvesting rates, prices of honey
	TRAFFIC 1997; Loibooki *et al.* 2002; Ndibalema & Songorwa 2008; Knapp *et al.* 2010; Schmitt 2010; Rentsch & Damon 2013; Kenya Water Towers Agency 2018	Household participation, average household harvesting rates, prices of bushmeat
	Schneider & Blyther 1988; Kajobe & Roubik 2006; Jaffé *et al.* 2010; Garcia *et al.* 2013	Bee colony density, honey production
	Mizutani, 1999; Caro, 2008; Kenya Wildlife Service and Tanzania Wildlife Research Institute, 2010; Moses *et al.*, 2015; Ogutu *et al.*, 2016; Moehlman *et al.*, 2020; Western & Mose, 2020	Biomass density of species hunted for bushmeat
	Marks, 1973; Coe, Cumming & Phillipson, 1976; Blumenschine & Caro, 1986; Holmern *et al.*, 2006	Masses for conversion of wildlife numbers to biomass, estimates of edible meat across species
Grasses, reeds and sedges	Turpie, Ngaga & Karanja 2005; KNBS 2019; NBS Tanzania 2015	Household participation, average household harvesting rates, prices of thatching grass, reeds and sedges, and palm leaves
	Hall, Meredith & Altona, 1955; Meredith, Scott & Rose, 1955; Donaldson, 1967; Mills, 1968; Bourlière & Hadley, 1970; Donaldson & Kelk, 1970; Fourie & Roberts, 1976; Rutherford, 1978; Chidumayo, 1997; Turpie *et al.*, 1999; Mworia *et al.*, 2008;	Annual productivity of thatching grass

NATURAL RESOURCE	REFERENCES	DATA
	Verdoodt *et al.*, 2009; Selemani *et al.*, 2013; Shirima *et al.*, 2015	
Northern Savannas Wildlife Landscape		
Firewood, charcoal, poles, timber	Ellis *et al.* 1984; AWF 2014; Egeru, Kateregga & Majaliwa 2014a; UBOS 2018; Barbelet 2012; Uganda Bureau of Statistics 2014; KNBS 2019	Household participation, average household harvesting rates, prices
	Rutherford, 1978; Mung'ala & Openshaw, 1984; Chidumayo, 1988, 1993; Malimbwi *et al.*, 2005; Maliondo *et al.*, 2005; Barnes *et al.*, 2010; Sisay *et al.*, 2017; Santoro *et al.*, 2018	Above-ground biomass, basal area, woody volume m^3/ha, mean annual increment for different vegetation types and land uses, proportion of woody biomass available for different uses
Wild plant foods and medicines	Scott 1998; Nkuutu *et al.* 2000; Kanabahita 2001; Kalema 2010; Lovett 2013; AWF 2014; Arensen 2015; Langoya 2017	Household participation, average household harvesting rates, prices
	Campbell, 1987; Campbell *et al.*, 1997; Assefa & Abebe, 2011	Annual productivity of wild plant foods
Wild animal resources	Scott, 1998	Household participation, average household harvesting rates, prices of honey
	Scott 1998; Olupot, McNeilage & Plumptre 2009; Langoya 2017; Jubara 2019	Household participation, average household harvesting rates and prices of bushmeat
	Schneider & Blyther 1988; McNally & Schneider 1996; Jaffé *et al.* 2010; Garcia *et al.* 2013	Bee colony density, honey production
	Caro, 2008	Biomass density of species hunted for bushmeat
Grasses, reeds and sedges, palm leaves, bamboo	Uganda Bureau of Statistics 2014; KNBS 2019	Households using grasses and palm leaves in construction of houses
	Scott, 1998	Household participation, prices, production rates for bamboo
	Hall *et al.*, 1955; Meredith *et al.*, 1955; Mills, 1968; Bourlière & Hadley, 1970; Fourie & Roberts, 1976; Rutherford, 1978; Chidumayo, 1997; Turpie *et al.*, 1999; Mworia *et al.*, 2008; Verdoodt *et al.*, 2009; Selemani *et al.*, 2013; Shirima *et al.*, 2015	Annual productivity of thatching grass
Akagera-Mugesera-Rweru Wetlands		
Firewood, charcoal, poles, timber	Kaale *et al.* 2000; Turpie 2000a; Rwamahe 2008; NISR 2012; Drigo *et al.* 2013; NBS Tanzania 2015a; Gianvenuti & Vyamana 2018	Household participation, average household harvesting rates, prices
	Rutherford, 1978; Chidumayo, 1988; Barnes *et al.*, 2010; Santoro *et al.*, 2018	Above-ground biomass, basal area, woody volume m^3/ha, mean annual increment for different vegetation types and land uses
Wild plant foods and medicines	Turpie, Ngaga & Karanja 2005; Wanjohi *et al.* 2011	Household participation, average household harvesting rates, prices
	Campbell, 1987; Campbell *et al.*, 1997; Assefa & Abebe, 2011	Annual productivity of wild plant foods

NATURAL RESOURCE	REFERENCES	DATA
Wild animal resources	NISR 2010	Household participation, average household harvesting rates, prices for wild honey and fish
	Abila 2002; Turpie *et al.* 2005	Household participation, average household harvesting rates, prices for hunting of small mammals and birds
	Wirminghaus & Perrin, 1993; Monadjem, 1997; Monadjem & Perrin, 2003; Caro, 2008	Biomass density of species hunted for bushmeat
Grasses, reeds, sedges	Dixon & Wood 2003; Turpie *et al.* 2005; NISR 2010, 2012; Wanjohi *et al.* 2011; NBS Tanzania 2015b; Thenya & Mwaniki 2017	Household participation, average household harvesting rates, prices for reeds and sedges and thatching grasses
	Mills, 1968; Bourlière & Hadley, 1970; Rutherford, 1978; Chidumayo, 1997; Turpie *et al.*, 1999; Estes *et al.*, 2012; Shirima *et al.*, 2015	Annual productivity of thatching grass
	Thompson *et al.*, 1979; Cunningham, 1985; Jones & Muthuri, 1985; Chale, 1987; Jones MB & Muthuri FM, 1997; Mnaya *et al.*, 2007; Jones *et al.*, 2018	Annual productivity of reeds and sedges

Each country has different naming typology for administrative boundary levels (Table 82). For this analysis, the data were analyzed at the second administrative level in the Great East African Plains region, Albertine Rift Forests region, and Northern Savannas region and were analyzed at the third administrative level in the Western Wetlands region. Census data for the six countries contains household information at varying levels of disaggregation and detail. For all countries except Burundi and South Sudan, the census data was available at the second administrative level. For Burundi and South Sudan, we relied on census data at the provincial or state level.

Table 82. The administrative boundary levels for each country

LEVEL 0	LEVEL 1	LEVEL 2	LEVEL 3
Burundi	Province	Commune	Colline
Kenya	County	Sub-county	Ward
Rwanda	Province	District	Sector
South Sudan	State	District	
Tanzania	Region	Sub-region	Ward
Uganda	District	County	Sub-county

Source: GADM database of global administrative areas

POPULATION

To ensure consistency across the study region, spatial population data (*i.e.,* gridded population datasets) from WorldPop (www.worldpop.org - School of Geography and Environmental Science *et al.* 2018) for 2018 was used in this assessment. The data are based on sub-national census figures for each country and are adjusted to match corresponding official U.N. population estimates. The data are mapped as the number of people per pixel at a resolution of 100 m. The 2018 Land Cover was then used to isolate the urban population from the rural population. Average household size, extracted from country census reports, was used to estimate the number of rural households per administrative area.

ESTIMATION AND AVAILABILITY OF STOCKS

Spatial variation in resource stocks and yields per unit area were estimated based on information from the literature for each habitat type. Just over 100 studies were consulted for estimating the quantity of stocks per unit area for each resource in each habitat type. Data from within the study region was used as far as possible. Where data was not available from within the study region, studies from the same habitat type but from outside the region were used. Woody stocks (in m^3 per hectare) were estimated from the GlobBiomass global woody biomass map (Santoro *et al.*, 2018). For each habitat, zonal statistics were used in ArcGis to get an average biomass value from the GlobBiomass above-ground biomass dataset. This is defined as the oven-dry weight of the woody parts (stem, bark, branches, and twigs) of all living trees excluding stumps and roots. Where appropriate, the biomass layer was also used for interpolation where resource stock estimates were lacking for a habitat type, based on the assumption that stocks of certain resources are proportional to woody biomass.

All the harvestable resources were considered fully available and accessible within areas under communal land tenure. In reality, this could be limited by local traditional leaders, but there is little information on this. The assumed availability was reduced to 20 percent of standing stocks in national parks except for Akagera National Park in Rwanda, which is fenced, where the assumed availability was set to zero. In all other protected area types (*e.g.*, forest reserve, game-controlled area, community conservancy, etc.) the availability was reduced to 50 percent of standing stocks. The retention of some availability in these areas was to allow for illegal or limited sanctioned harvesting. While most national parks have a no-take policy for resources, many of them have experienced some level of unsanctioned resource extraction. Over time, other types of protected areas with varying degrees of protection have introduced arrangements and rules to allow controlled access to certain resources.

ESTIMATION OF HOUSEHOLD DEMAND

The quantities of resources harvested by subsistence and small-scale users from terrestrial and freshwater habitats was calculated based on the estimated household demand and available stocks in the landscape for each of the four study regions. Quantities demanded were estimated at the second-level administrative boundary based on population data (see www.worldpop.org), census data available for each country, data from The World Bank's Living Standards Measurement Study, and household survey data collated from a range of socio-economic studies that have been carried out in the rural areas of Burundi, Kenya, Rwanda, South Sudan, Tanzania, and Uganda, and elsewhere in East Africa where data was lacking. Relevant census data available included number of rural households, average household size, households using poles for construction of houses, number of households using firewood for heating and cooking, households using grass for thatching of roofs, and households using timber for construction of houses. Census data on resources used in the construction of houses was limited for South Sudan and Burundi, reported at the state/provincial level, or not reported at all. Where this was the case, estimates from neighboring Uganda in the case of South Sudan or Rwanda for Burundi and information from the literature were used instead. Some census reports (e.g., KNBS, 2019) provided estimates on household activities such as fishing, and agricultural census reports (e.g., NISR, 2010) also provided data on fishing and honey harvesting activities in Rwanda. The USAID FEWS NET reports provided useful socio-economic descriptions of the study regions and in some cases provided information on household reliance on wild resources. To align estimates of quantities harvested and ensure consistency, a set of assumptions were made for each resource (Table 83).

Table 83. Assumptions made for various resources in the estimation of household demand

RESOURCE	ASSUMPTIONS
Thatching grass	• A large bundle of grass has an estimated weight of 6 kg (based on the average taken from range of studies: Shackleton 1990; Otsub et al. 2004; Mmopelwa, Blignaut & Hassan 2009 • Thatching material needs to be replaced every six years.
Reeds and sedges	• A large bundle of reeds has an estimated weight of 11 kg (based on average taken from range of studies: Adekola *et al.*, 2008; Mmopelwa *et al.*, 2009; Köbbing, Thevs & Zerbe, 2013; Turpie *et al.*, 2014.
Palm leaves	• A large bundle of palm leaves has an estimated weight of 6 kg (assumed to be the same as a large bundle of grass). • Palm leaves are used for thatching, often in combination with grasses. • Palms are found in very specific habitats, often in isolated pockets, and are not abundant across the landscape. • Thatching material needs to be replaced every six years.
Woody resources	• Average wood density of 855 kg/m^3 used to convert kg of woody resources into m^3. Based on the average wood density of tropical hardwood African trees (FAO). • Converting charcoal to roundwood equivalent – 5.9 m^3 of wood produces one ton of charcoal. Based on the average tropical hardwoods of 170 kg as the weight of charcoal per one m^3 of wood (FAO). • An average pole used in construction has a volume of 0.013m^3. • The harvesting of poles and timber relates only to subsistence harvesting for the construction of walls and other structures. • Charcoal production relates only to subsistence use and does not include the commercial harvest for distribution to urban centers.
Honey	• A liter of honey weighs 1.425 kg.
Wild plant foods	• In the Northern Savannas region, wild plant foods include shea nuts, wild dates, and gum arabica. While these may have some commercial value, we believe that the commercial shea trade is not well established in Uganda, and the commercial gum markets in Northeast Uganda collapsed a few decades ago. We have therefore grouped them with the other subsistence plant food products as this best reflects the reality of how they are used in the area. We could find no evidence of commercial production.

Based on the data collated from the above sources, household demand was calculated for each resource as the harvest per average household per year (e.g., kg/hh/y). This was then multiplied by the total number of rural households within each administrative area to get total aggregate demand for that area (*e.g.*, kg/y, m^3/y). The total demand in each administrative area was then mapped using the gridded population dataset and adjusted based on the proportion of population per administrative area per hectare (100 m x 100 m grid).

AMOUNT OF WILD RESOURCES HARVESTED

The amount of wild resources harvested for subsistence use was estimated based on the minimum of the estimated demand and the estimated available stocks of resources within a specified distance of the demand source. To estimate and map harvesting at a high resolution, we used a running-mean method as developed by Turpie *et al.* (2020) in KwaZulu-Natal, South Africa. The running-mean method entailed estimation of the value for each grid cell based on multiple spatial computations, in turn based on the spatial relationships between the units of demand (households) and the availability of the relevant resources in the surrounding landscape.

We based the dimensions of our analysis on an estimated average travelling distance to harvest natural resources of about 6 km. The literature on African countries reports a large range in such distances, and often focuses on the time spent harvesting rather than distance travelled. It is also worth noting that total distance travelled is also not necessarily in a straight line away from households, so total distance travelled is likely to be more than twice the potential radius of the area searched. This use of 6 km was based on reviewing the following studies: Banks *et al.,* 1996; Turpie & Egoh, 2003; Madubansi & Shackleton, 2007; Agea *et al.,* 2010; Matsika, Erasmus & Twine, 2013; Wessels *et al.,* 2013; Amoah, Marfo & Ohene, 2015.

The dimensions of the square (10x10 km) relate to the assumption of the expected maximum distance travelled from households to collect resources, since the average distance from center to the perimeter is about 6 km. The running mean was generated by recalculating the values using a total of ten 10x10 km grids, each of which was offset from the previous grid by 1 km to the east and south (*i.e.,* the grid is duplicated but with a specified offset per grid to create a series of sequential grids). In each iteration the relative demand and availability differ. The mean from the series of iterations is the final value per pixel. The use has a cap set at the maximum availability of the resource (*i.e.,* demand cannot outstrip supply, and where it does, then supply must be coming from elsewhere).

The running-mean method leads to resource value estimates being higher in the supply zones closer to the centers of demand and attenuating from there, which provides a relatively realistic pattern of harvesting under some simplifying assumptions and does away with the need for modeling a complex distance-decay function in GIS. However, it is still limited in that it does not take factors such as topography, other physical barriers, or use of road transport into account. In situations where the local demand is higher than can be sustained within usual walking distance, it is to be expected that entrepreneurs with access to transport will bring resources from more distant areas. For this reason, harvesting is unlikely to be capped at levels of availability within the local area, but the total size of the source area used to meet demand will be determined by economics as well as accessibility.

VALUATION

The estimated total amount of resources extracted was valued, irrespective of whether the estimated level of harvesting was sustainable or permitted. The resource rent method was used for valuing the quantities harvested, where value is the total revenue minus intermediate costs, labor costs, depreciation, and return on fixed capital. Total revenue was taken to be the market value of the resources harvested, irrespective of whether they were consumed or sold, using average prices obtained from the literature (Table 84). The costs of harvesting natural resources includes the opportunity cost of labor and input costs, including annualized costs of equipment. Some studies take the approach of using the shadow price of wage labor, which represents the rate at which people would be willing to work, *i.e.*, adjusted for employment conditions. In the remote, rural areas of East Africa, where natural resources are harvested for subsistence purposes, the rate of unemployment is high and there are few alternative income opportunities. Employment in the formal sector is generally very limited. Those individuals that are spending their time harvesting resources are not doing so at the cost of alternative income. This study assumed that all input costs were negligible.

Asset value is calculated based on projected flows of benefits over time, holding external factors such as change in climate, population, income levels, and preferences constant for the sake of simplicity and comparability. However, in the case of wild resources provisioning, the contribution of this service to asset value needs to take sustainability of harvesting into account. To account for this, harvesting was compared to the corresponding sustainable yield for each resource. Where harvesting exceeded the estimated sustainable use, the stocks were eroded at the corresponding rate, affecting future use and values.

Table 84. Values used for harvested wild resources. Value per unit, 2018 US$

RESOURCE	UNIT	GREAT EAST AFRICAN PLAINS	ALBERTINE RIFT FORESTS	NORTHERN SAVANNA	WETLANDS
Wild fruits and vegetables	kg	0.4	0.4	0.4	0.4
Medicinal plants	kg	1.3	3.6	2.7	2.5
Mushrooms	kg	-	1.9	-	-
Reeds and sedges	kg	0.5	0.7	-	0.6
Thatching grasses	kg	0.4	0.4	0.4	0.4
Palm leaves	kg	5.2	5.2	5.2	5.2
Bamboo	culm	-	0.7	0.7	-
Firewood	m^3	18.8	31.3	28.9	26.3
Charcoal	m^3	17.5	20.6	19.1	19.1
Poles and withies	m^3	23.6	24.0	24.0	23.9
Timber	m^3	136.0	130.1	120.3	128.8
Honey	liter	0.7	1.6	1.0	1.1
Bushmeat	kg	0.8	1.2	1.3	1.3
Freshwater fish	kg	0.4	0.5	0.7	0.4

APPENDIX 5: PROJECTED CLIMATE CHANGE IMPACTS ON BIODIVERSITY

Climate change has had and will continue to have a profound impact on global biodiversity. The continent has experienced a 0.7°C mean temperature increase since 1900, with a projected increase ranging from 0.2°C to over 0.5°C per decade (Sintayehu, 2018). While the change in climate is overall that of warmer average temperatures, different regions are predicted to have variable future climate under different scenarios. Most climate models suggest a high likelihood of increased heavy rainfall in many parts of East Africa and generally increased overall rainfall and reduced periods of drought. The region is likely to experience between 0.5 and 5°C mean temperature increases, with the northern and western parts of the region forecast to experience the most intense warming (Daron, 2014; Niang *et al.*, 2014).

Terrestrial ecosystems are strongly influenced by climate and have been shaped over the course of time. Their current structure and composition reflect historical climatic shifts and present-day conditions. These ecosystems are changing more rapidly than before and this, combined with climatic change, is likely to have consequences for various species in the region, leading to possible range shifts into areas that are more climatically suitable. Species distribution models allow for prediction of current and future species distribution based on actual presence records combined with habitat and climate suitability, land cover, and species' traits and phylogeny. Range maps are produced using both statistical and process-based models, with the input of regional biodiversity experts.

Species richness maps and range maps of several charismatic species were derived for current (2018) and future periods (2040 to 2060) under different scenarios (Figure 98). Of course, in reality, the lack of suitable habitat in future range predictions due to reduction in natural land cover means that there is even less available area for species to persist should the models that predict these changes be correct. This highlights the importance of ensuring conservation areas are planned with climatic shifts in mind to ensure resilience of biodiversity at risk. It is important to bear in mind that the models are imperfect, as highlighted by some of the results shown. However, they do allow us to get a good sense of potential shifts in distribution, which should be verified further and inform planning for species and ecosystems.

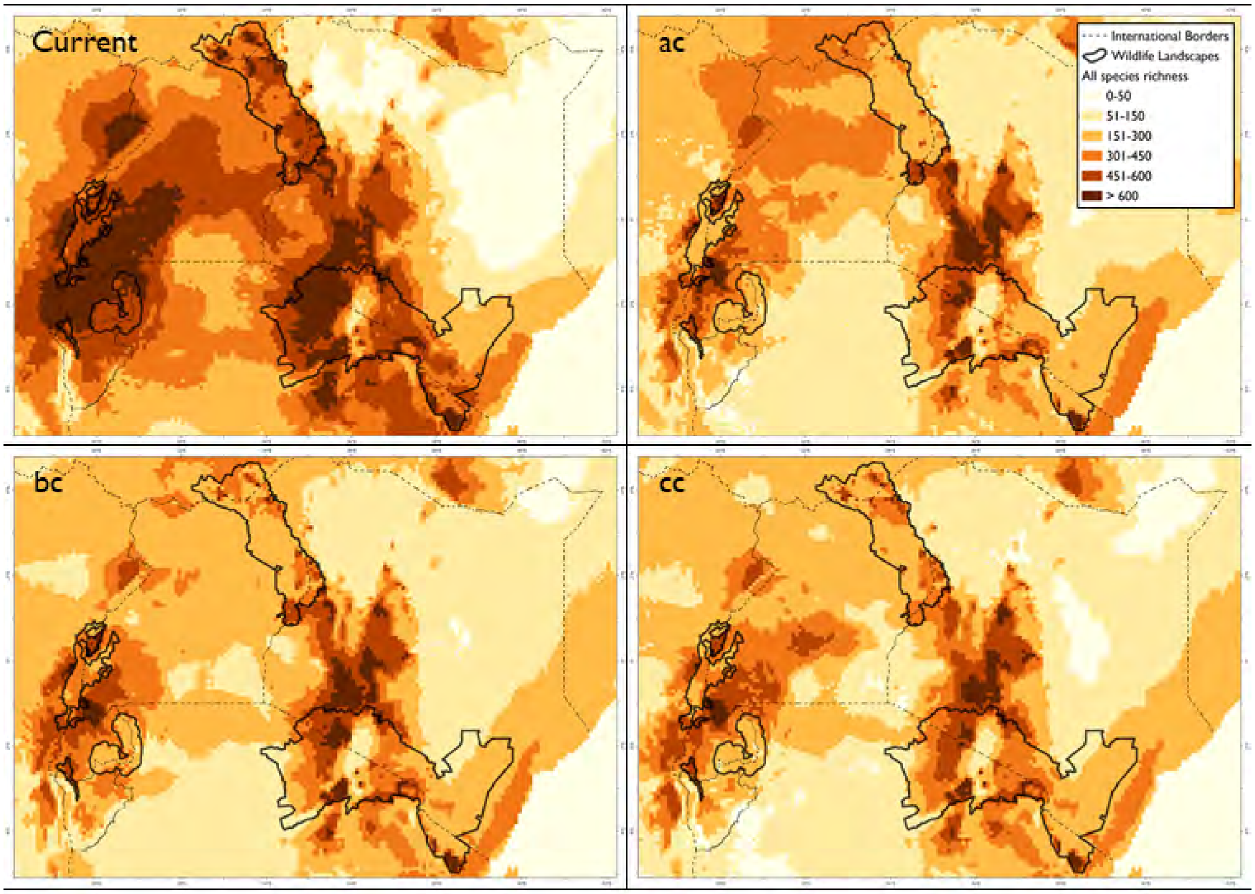

Figure 98. Current habitat suitability of all species (amphibians, birds, mammals and reptiles) richness for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modeled species distributions from Conservation International

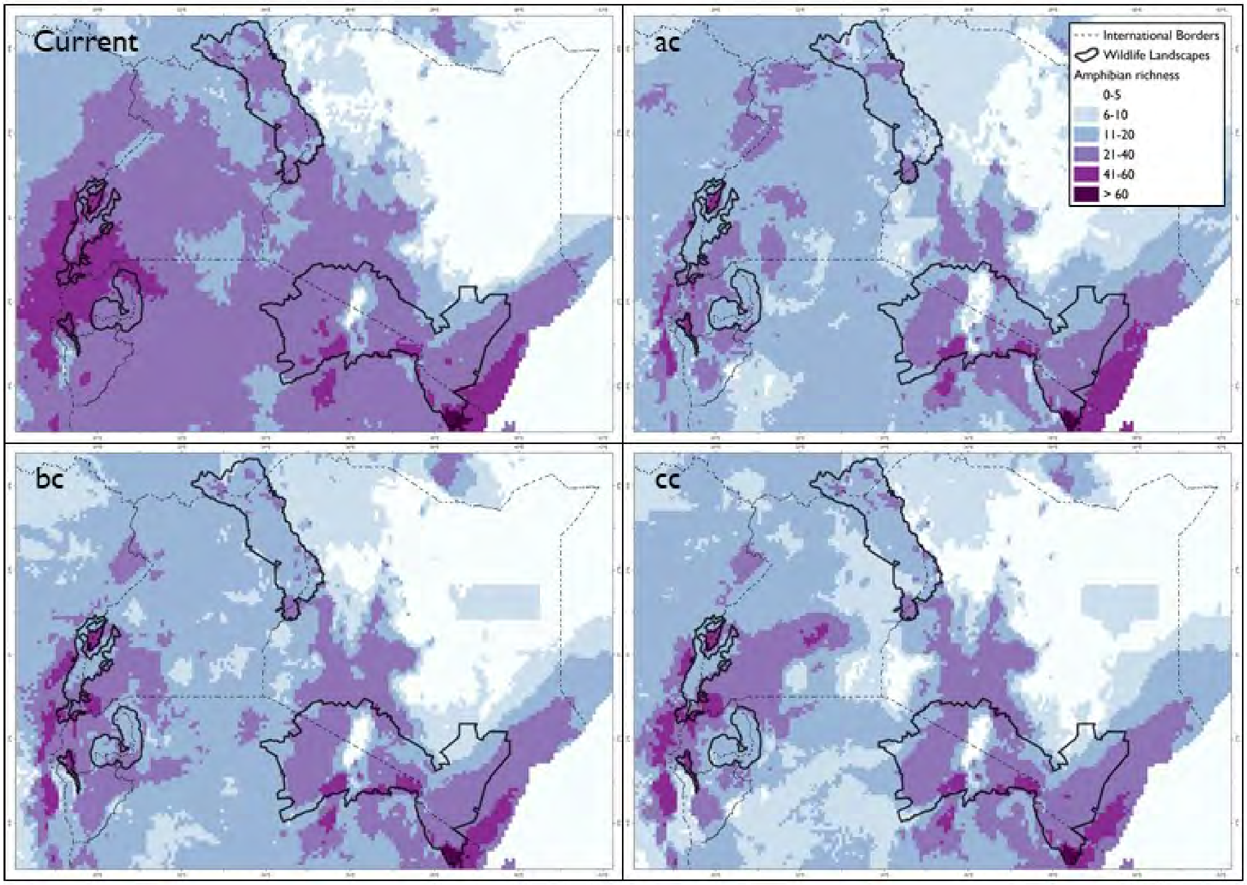

Figure 99. Current habitat suitability of amphibian species for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modeled species distributions from Conservation International

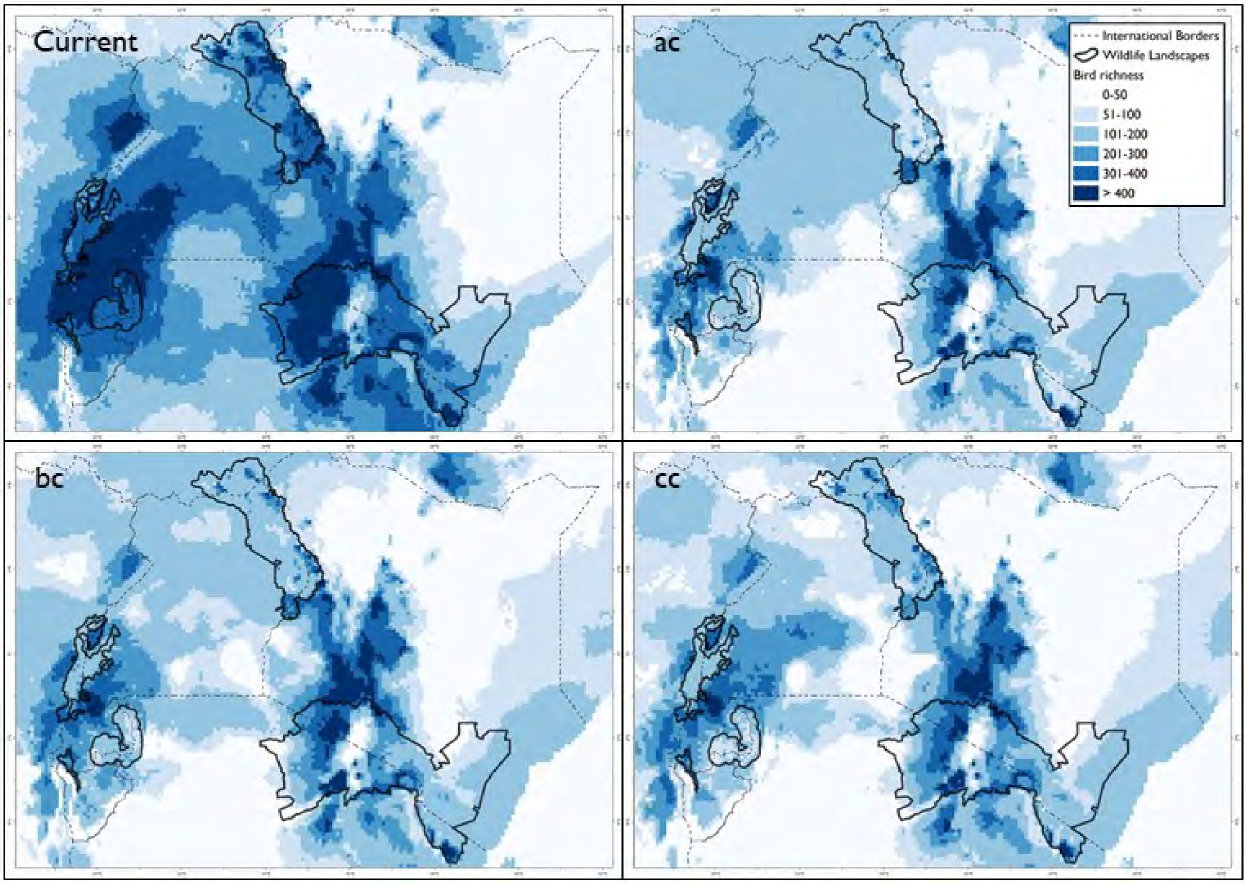

Figure 100. Current habitat suitability of bird species for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modeled species distributions from Conservation International

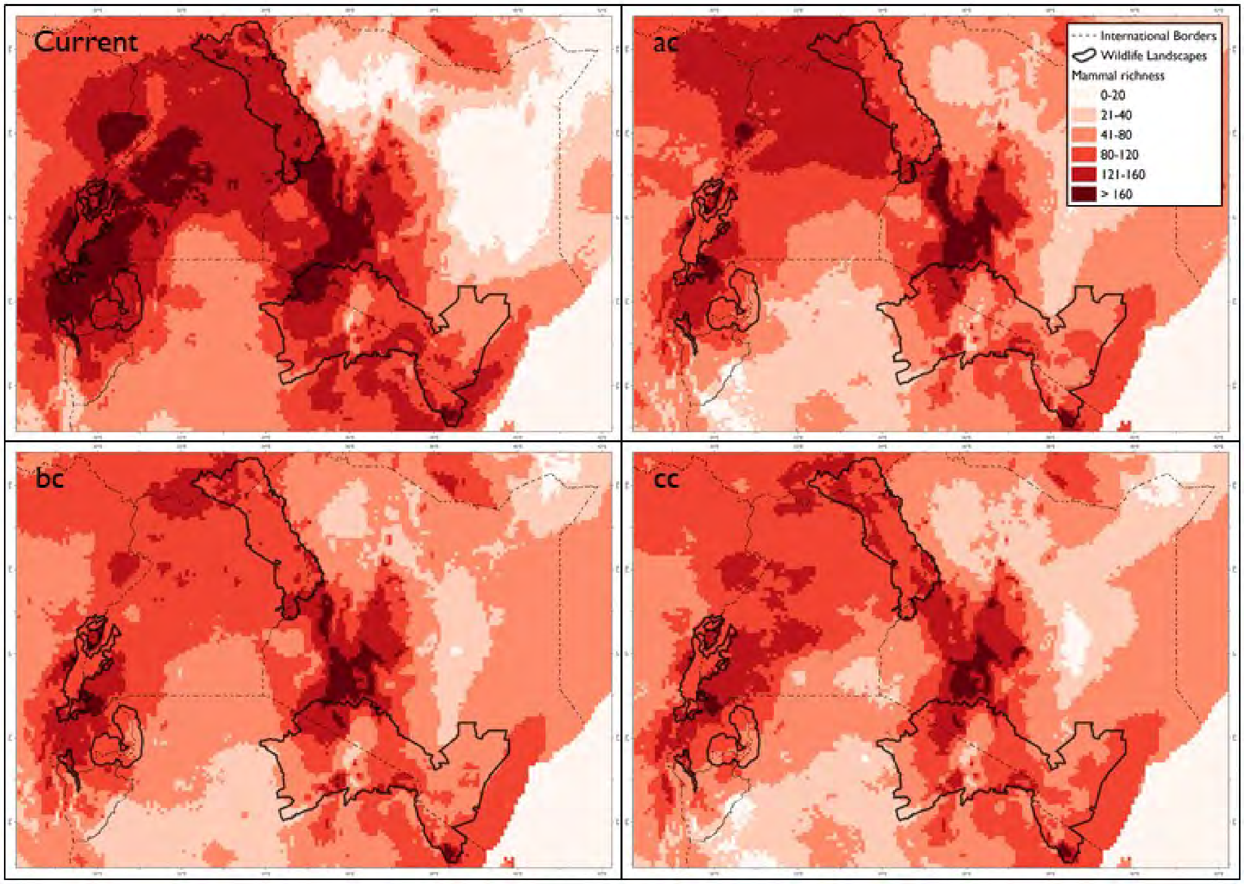

Figure 101. Current habitat suitability of mammal species for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modeled species distributions from Conservation International

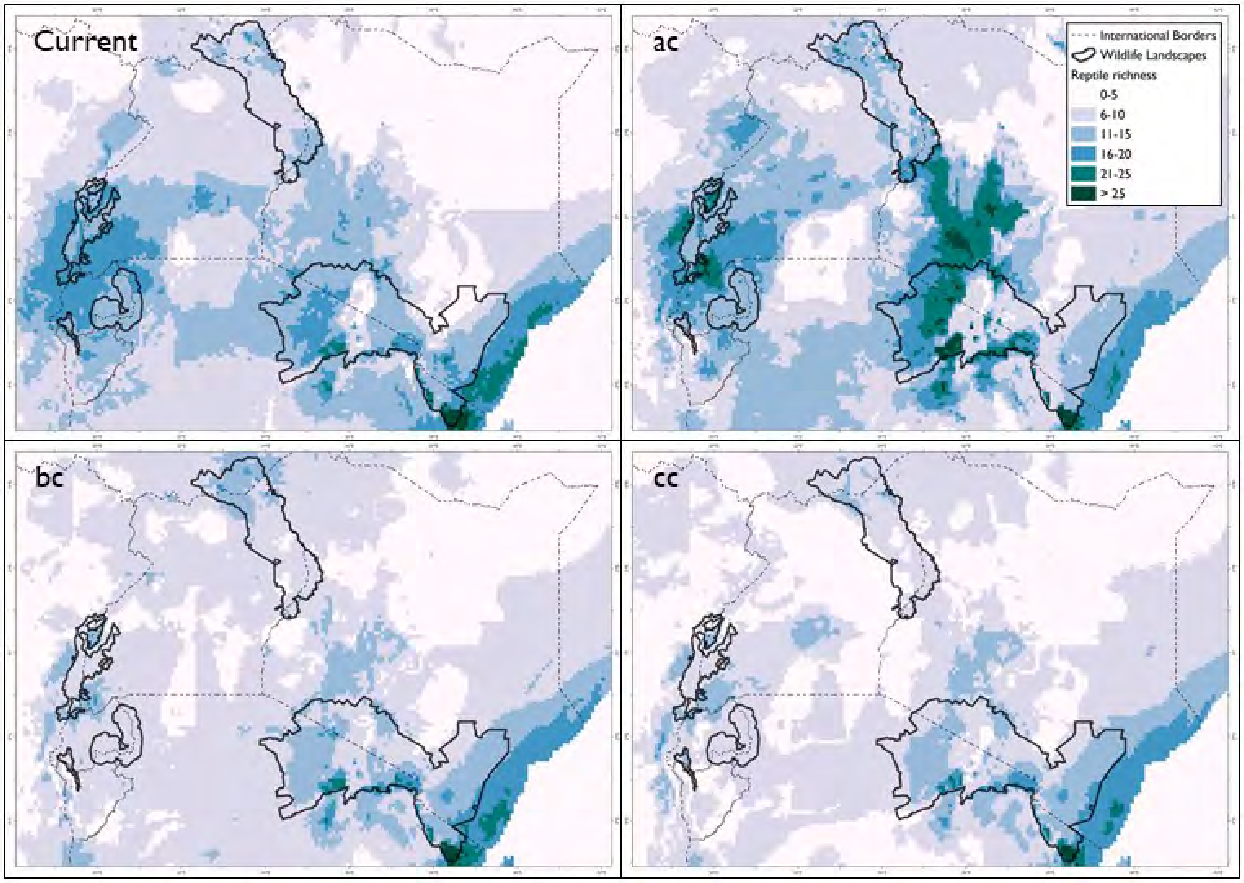

Figure 102. Current habitat suitability of reptile species for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modeled species distributions from Conservation International

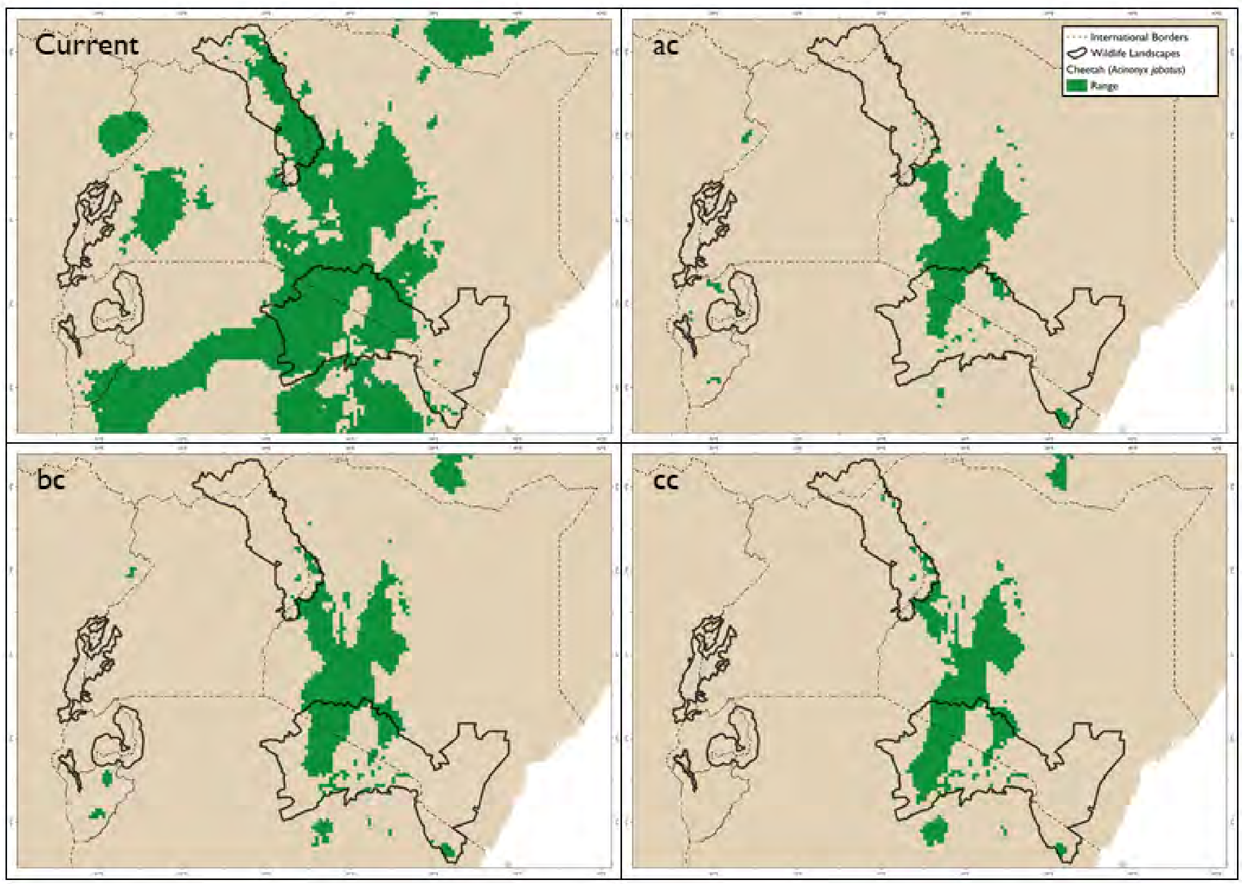

Figure 103. Current habitat suitability of cheetah (Acinonyx jabatus) for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modeled species distributions from Conservation International

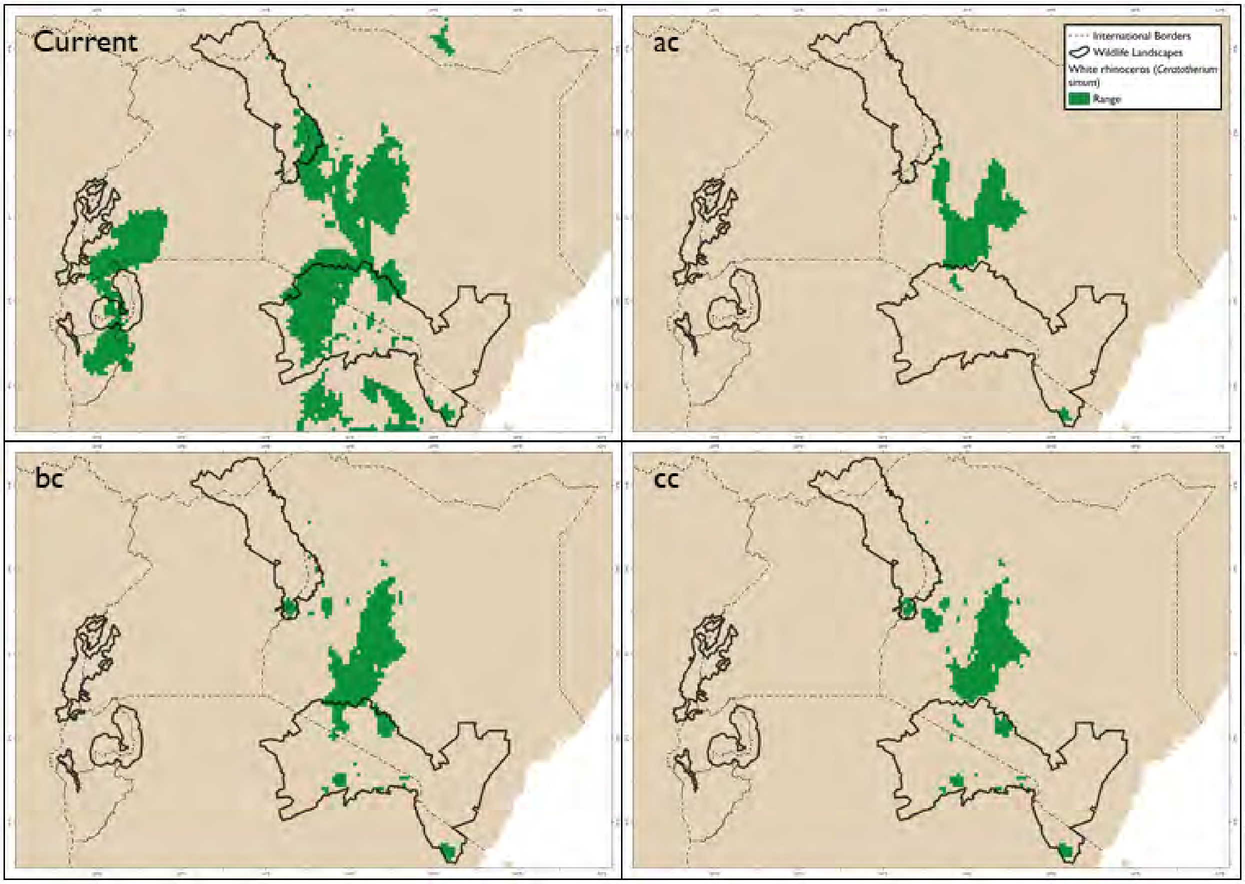

Figure 104. Current habitat suitability of white rhinoceros (Ceratotherium simum) species for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modeled species distributions from Conservation International

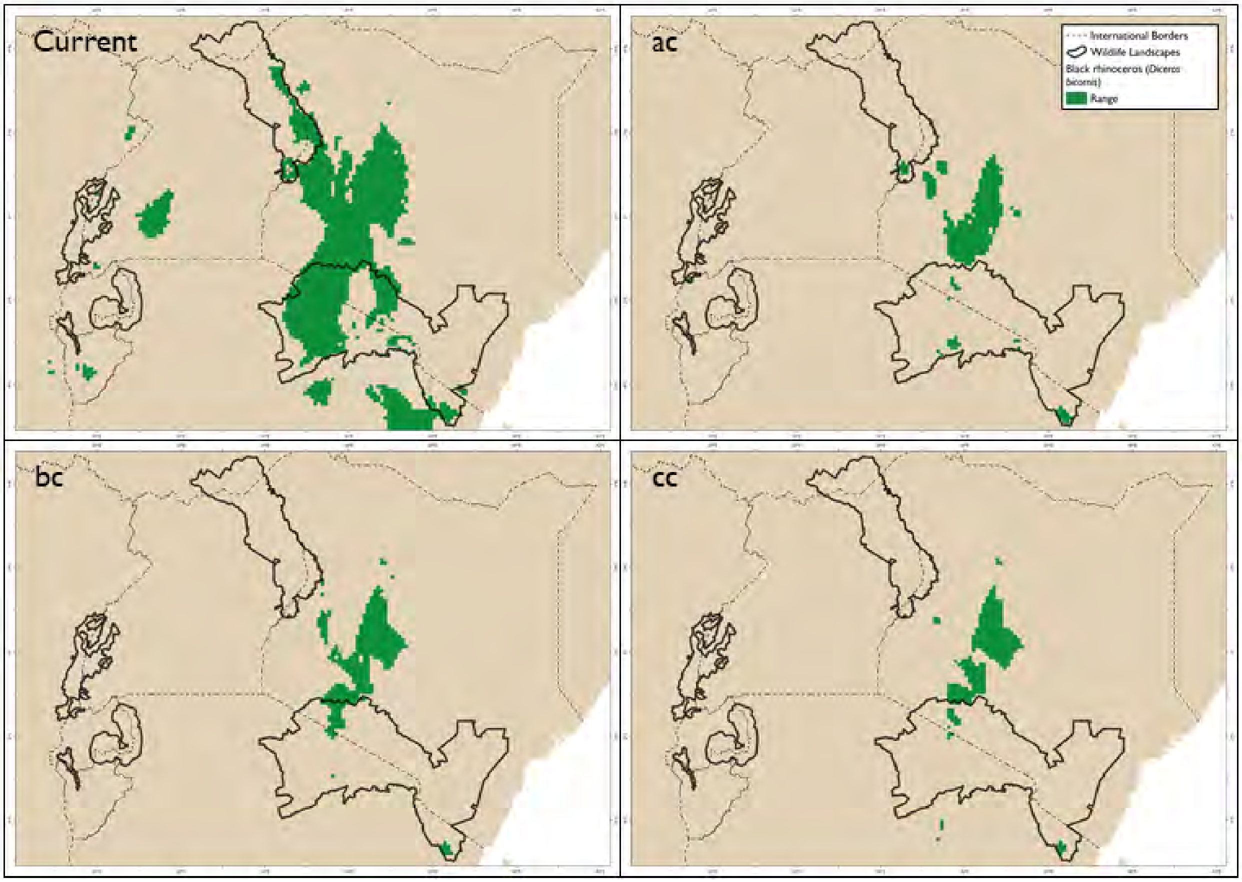

Figure 105. Current habitat suitability of black rhinoceros (Diceros bicornis) for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modelled species distributions from Conservation International

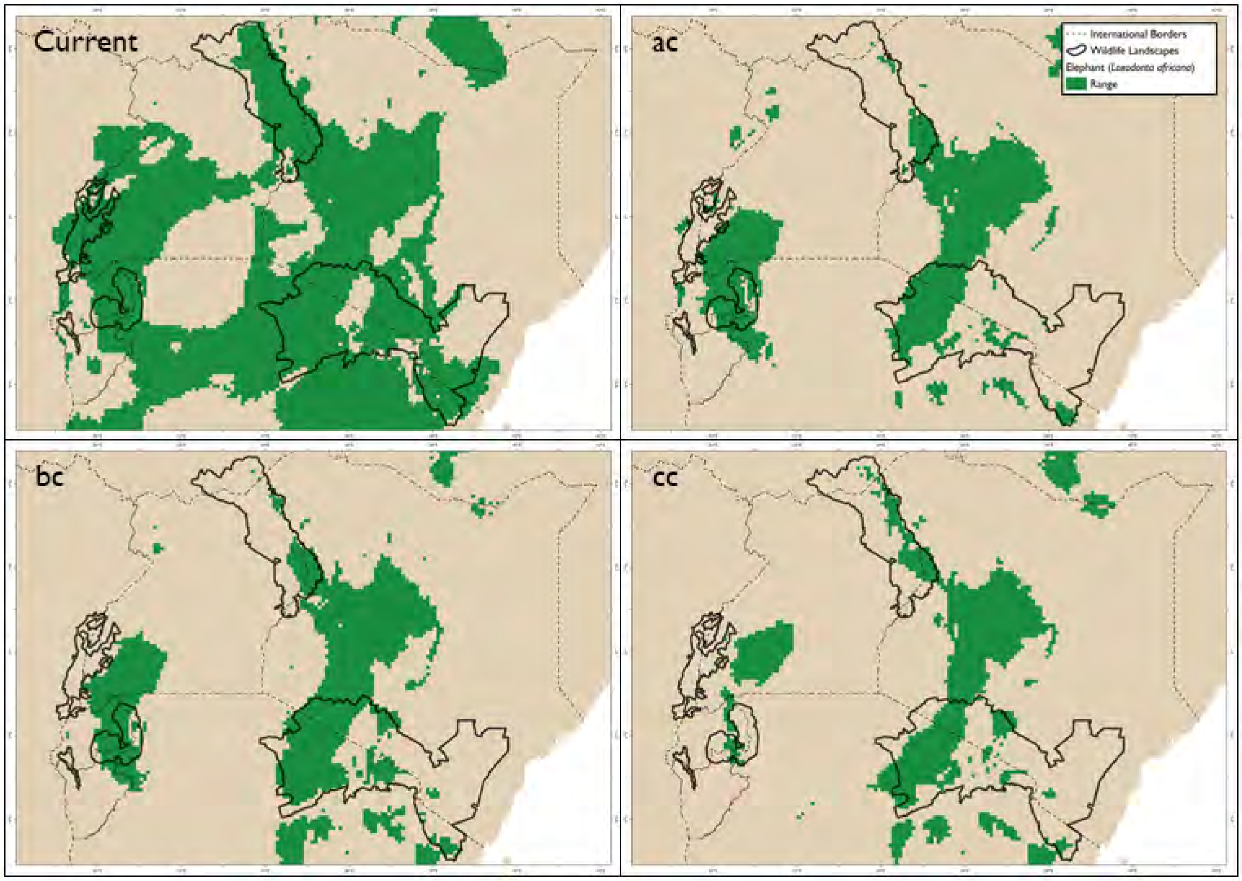

Figure 106. Current habitat suitability of elephant (Loxodonta Africana) for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modeled species distributions from Conservation International

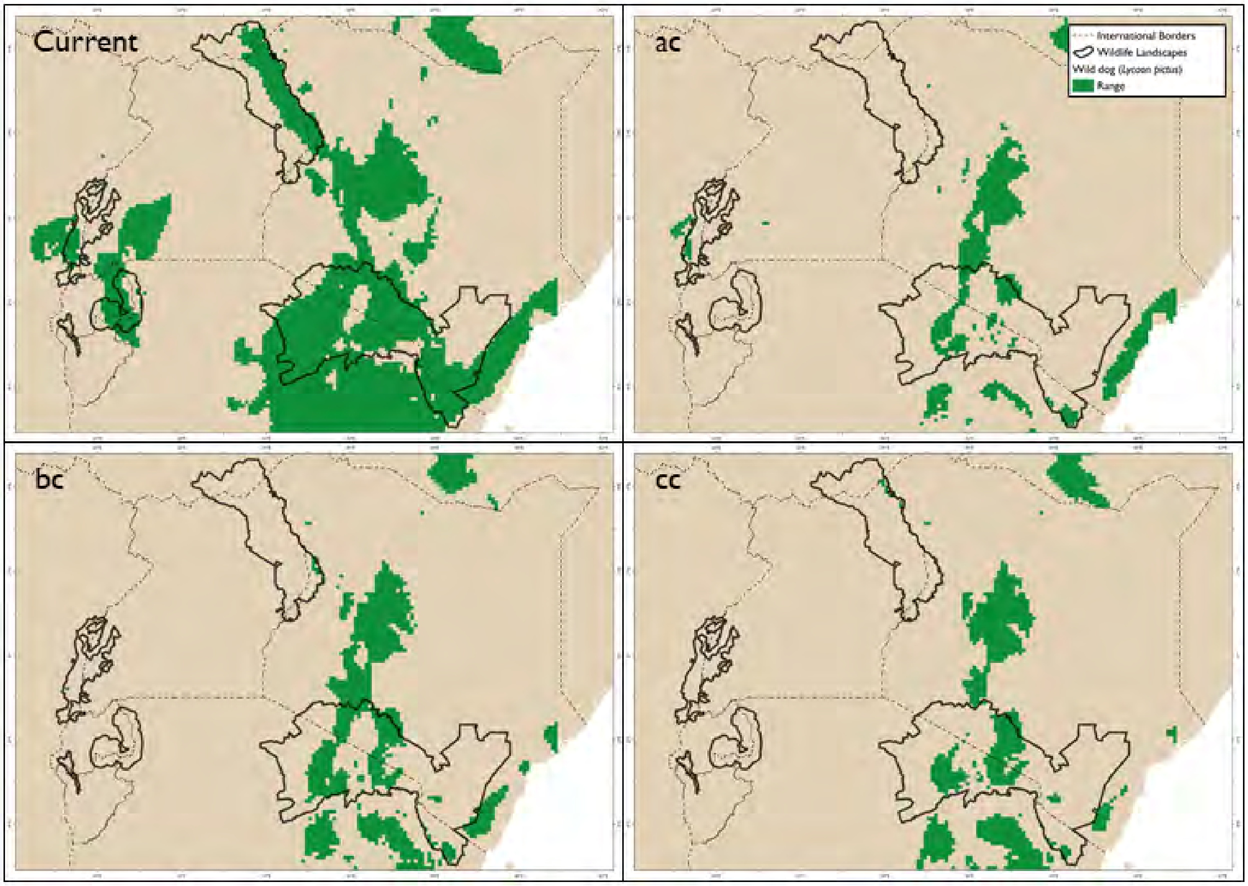

Figure 107. Current habitat suitability of wild dog (Lycaon pictus) for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modeled species distributions from Conservation International

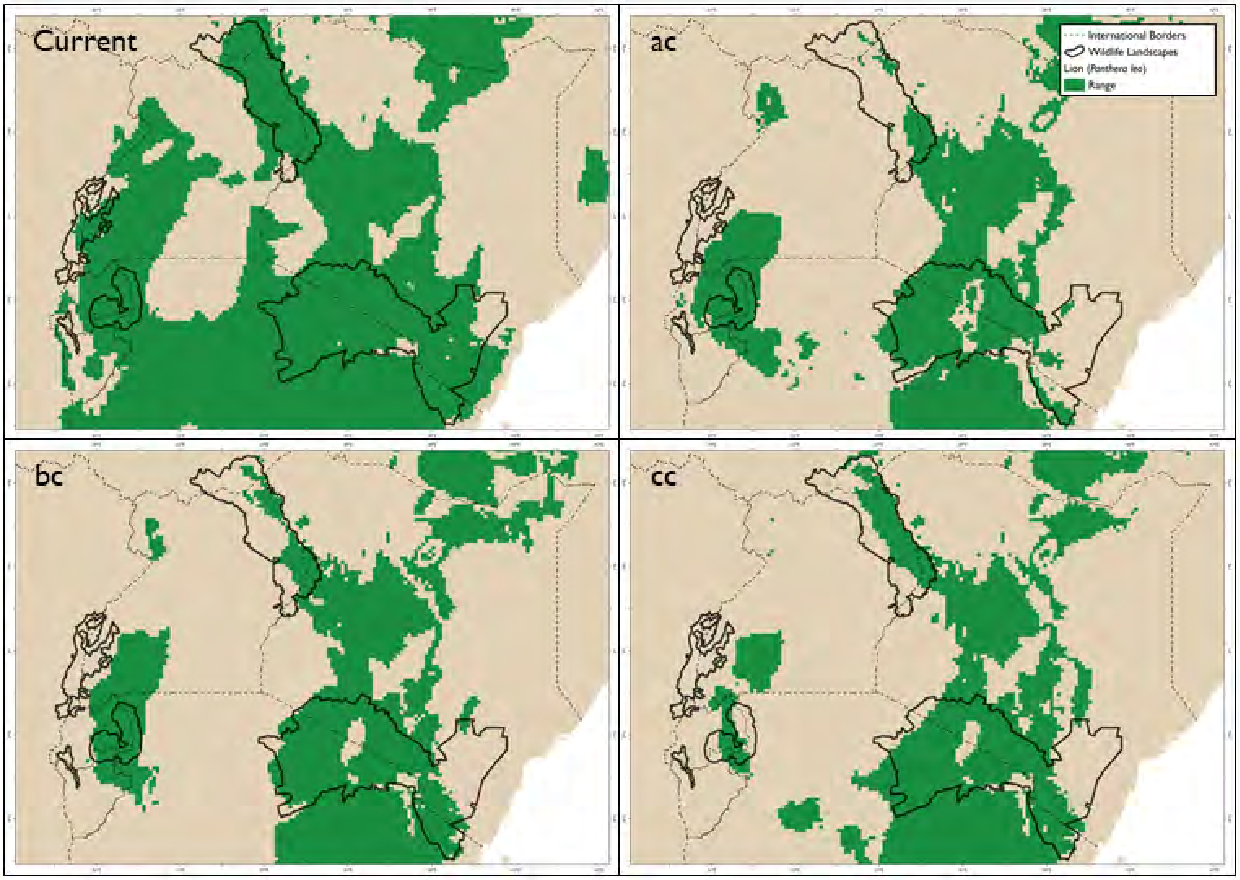

Figure 108. Current habitat suitability of lion (Panthera pardus) for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modeled species distributions from Conservation International

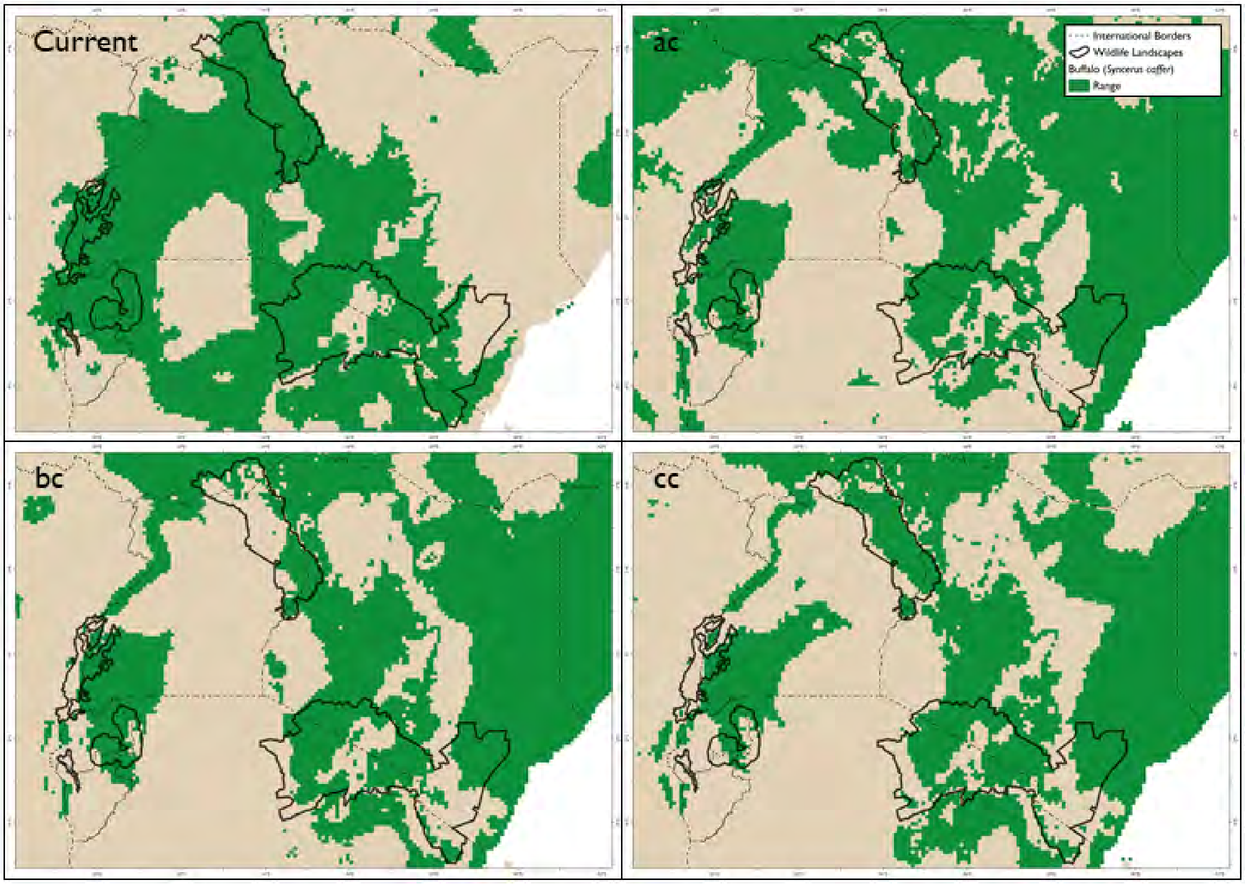

Figure 109. Current habitat suitability of buffalo (Syncerus caffer) for study area (including wildlife landscapes), followed by the projected species richness pattern for each of three climate models used (ac, bc, and cc)

Source: Based on modeled species distributions from Conservation International

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