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INTEGRATED MODELING OF LAND USE AND CLIMATE CHANGE IMPACTS ON MULTISCALE ECOSYSTEMS OF CENTRAL AFRICAN WATERSHEDS

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INTEGRATED MODELING OF LAND USE AND CLIMATE CHANGE IMPACTS ON MULTISCALE ECOSYSTEMS OF CENTRAL AFRICAN WATERSHEDS

A Dissertation Presented

by

SIMON NAMPINDO

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DEDICATION

I dedicate my dissertation to the memory of my mother ROSE-MARY KYESUMBE (RIP).

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ABSTRACT

INTEGRATED MODELING OF LAND USE AND CLIMATE CHANGE IMPACTS ON MULTISCALE ECOSYSTEMS OF CENTRAL AFRICAN WATERSHEDS

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Assessment and management of ecosystem services demands diverse knowledge concerning the processes that impact the flow of those services. Land use land cover change occurring mainly through deforestation, colonization, expansion of agriculture and unregulated extraction of natural resources are the greatest challenges of the 21st century. The Congo basin, located in central Africa known for supporting a relatively intact forest served by the second longest river in Africa faces many environmental and human-driven threats. Extraction of timber, minerals, and now oil and gas, if uncontrolled or properly managed, the biodiversity of the basin will diminish very fast. Wildlife species are at the center of major conflicts in the region resulting in increased poaching, bushmeat hunting, and habitat destruction to provide settlement for refugees fleeing the war tone regions. Furthermore, the increase in human population coupled with poverty is likely to exacerbate resource degradation.

This study undertook to implement an integrated modeling of land use and climate change impacts on multiscale ecosystems of central African watersheds. The study was organized in such a way that implementation of the research was conducted at three varied scales namely the watershed or regional scale, landscape, and household or community scale. At the regional scale, watershed analysis, and hydrological assessment was done using remotely sensed data and modeling of surface runoff and soil loss using Geographical information System, specifically ArcGIS-based software. At the same time, species distribution modeling using generalized linear models that depend on presence only and/or presence and pseudo absence data were used to develop the species distribution maps. The data used to implement this study was requested for and accessed from various institutions that are involved in the research. Land use land cover map produced by USGS, the carbon map developed by NASA's Jet Propulsion Laboratory, environmental data from IPCC's Worldclim, and species inventory data or selected endangered and threatened species was provided mainly by Wildlife Conservation Society, IUCN and Birdlife International among other conservation organizations was used to compile the biodiversity values.

The major findings are that ecosystem services for the Congo basin are spatially varied and there are significant differences in their distribution of ecosystem services at a subwatershed level three and six. Equally, the distribution of endangered and threatened species is more concentrated in the central part of the basin and most of these species occur mainly inside protected areas. Poaching and sustained civil wars were the biggest threat to elephant conservation in the region. Improvement in law enforcement, monitoring, and increasing household incomes for communities living adjacent to protected areas would help to reduce the impact of poaching on elephant population dynamics. More so, if wars and poaching are not controlled, the region may witness an elephant population dominated by juveNile and subadult elephants. Climate change did not show immediate direct impacts on the elephant population, but thermal and latent heat effects could be occurring. An increase in the suitable habitat by 50 % resulted in an upward trend in elephant population for all age classes suggesting that improvement in habitat management, particularly reducing the fires, encroachment, and degradation due to extraction of resources by local communities and corporate would facilitate elephant population increase and stability.

As the national governments pursue REDD+ initiatives, an attempt to pursue this strategy as a single policy approach, would compromise biodiversity, and watershed values. The local communities who depend heavily on these ecosystem services will also become more vulnerable to food insecurity and water scarcity associated with land use change. It is therefore recommended that 1) institutions with the mandate to manage and protect wildlife received greater funding to strengthen law enforcement and monitoring, 2) multiple or coupled policy options be considered in order to maximize ecosystem benefits;3) regional governments should invest in revitalizing the existing meteorological stations and water monitoring centers in order to generate information needed for water resources management 3) poaching needs to be treated as regional, and global security issue and attempts should be made by the international community to reduce the demand for ivory and other wildlife body parts. Lastly, since most of the wild species that exist in the Congo basin are very mobile and move beyond international boundaries, creation of partnerships and research networks need to be promoted. Existing transboundary collaboration initiatives should continue and be supported by the donor community in order to build peace and dram support for wildlife and wild habitat protection.

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CHAPTER 1

INTRODUCTION

1.1 Watershed Ecosystems

There is a growing appreciation that understanding of ecosystems is very critical to the management of water resources and in regulating the composition of the atmosphere that determines Earth's climate (Postel and Richter 2003; Chapin et al. 2008). Conceptual advances in ecosystem ecology have helped to create high recognition of the importance of past events and external forces in shaping the functioning of ecosystems. The main concern is that human exploitation of the Earth's ecosystems has increased tremendously in the last half-century (Steffen et al. 2004) resulting in detrimental effects (Foley et al. 2005; Ellis and Ramankutty 2008). There is also a growing ecological interest to understand how people act as agents of environmental change and impact landscape systems and process such as biogeochemical and hydrological cycles (Postel et al. 1996; Gleick 2003; Oki and Kanae 2006), and how climate change affect the pools and fluxes of materials and energy (McDonnell and Pickett 1990; 1993; Meyer 1997; DeFries et al. 2004; Foley et al. 2005).

Water is essential for the survival of humanity and wildlife. Changes in supplies and deterioration in quality can adversely impact the economic and ecological livelihood of people. This is particularly relevant to countries in Africa, where much of the human population and wildlife rely on local rivers, lakes, ponds, and wetlands for survival and provision of ecosystem products and services. Changes in land use and land cover (LULC) can adversely impact the extent and distribution of both quantity and quality of water resources in watershed systems. These changes are recognized as critical in governing the future availability and quality of freshwater (Chase et al. 2000; Bhuyan et al. 2002; Feddema et al. 2005, Marshall and Randhir 2008a). Land use threats to watersheds can occur directly through human activities like cropping, pasture and rangeland use, logging, urbanization, or through indirect transport of air pollutants or water-borne pollutants such as fertilizers, sediments, pesticides, pathogens, and herbicides, including invasive plants. Major land works such as diversion of surface waters for irrigation, damming of rivers, and vegetation cover destruction result in loss of water available for ecosystem needs.

Diversion of surface waters for irrigation and human uses can result in net loss of water available for ecosystem needs. Anthropogenic activities such as over-grazing, deforestation and other intensive land use activities on fragile soils leads to a net loss of soil water storage, reduced infiltration, and increased storm runoff, which have long-term consequences on the sustainability of agriculture and ecosystem services (Vörösmarty and Sahagian 2000). In addition, disturbance of soils through intensive cultivation, tillage, or other cropping practices can make the soils vulnerable to erosion (Bhuyan et al. 2002). Widespread deforestation can also contribute to a reduction in the recycling rate of water between plant canopies and the atmosphere, and thereby modify the regional climate system (Vörösmarty and Sahagian 2000; Bonan 2008). Deforestation and forest degradation, habitat fragmentation, extractive resource use (FAO 2007), and climate change, however, are the biggest contributors of loss in biodiversity and ecosystem services in watersheds. Mining of renewable (e.g. trees, fisheries, animals) and nonrenewable resources (e.g., minerals, fossil fuels) are some of the most important extractive industries in the region. Mining is typically conducted in and around streams, which causes localized destruction of riverine ecosystems as well as more dispersed impacts such as waterway sedimentation and pollution.

The Albertine Rift valley has some of the highest human population densities on the African Continent (FAO 2005). As a result, the pressure on the remaining natural vegetation and extraction of products in this region is enormous. In addition, declines in soil productivity, and the inability to carry out soil and water conservation practices on farmlands, and consideration of forested areas as idle lands for conversion to agricultural use have continued to diminish the ecosystem goods and services. Also, the increase in human population has resulted in overexploitation of the common pool resources in the region.

1.2 Problem analysis and study development

Land use change can alter ecosystem services and supply of natural resources. In the humid tropics, deforestation is a major threat to "biodiversity hotspots" (Brooks et al., 2005) that have the highest potential to supply ecosystem goods and services. FAO (2005) estimated an annual deforestation rate of nearly five million hectares occur in Africa. Duveiller et al (2008) estimated the deforestation rate in Central Africa at 0.21% per year, while the forest degradation rate was approximately 0.15% per year. Deforestation in the Congo Basin occurs at fine scales and is caused largely by shifting agriculture (Makana and Thomas, 2006). Laporte et al. (2007) estimates that more than 600,000 km^2 (30%) of forest is threatened by logging concessions, which in itself, has contributed to a road density of 0.07 km per km^2. These land use changes have had far-reaching consequences on ecosystems, food production, and local livelihoods in tropical forests. A comprehensive assessment of ecosystem services at a watershed scale is necessary to develop appropriate conservation policies and resource management decisions. Since the Albertine Rift Valley and the Congo River basin span such large areas with overlapping ownership and national boundaries, a systems' approach is required that accounts for multiple ecosystem services (Randhir and Shriver, 2009). Accounting for watershed patterns and processes, and their relative state at spatial scale can be used to evaluate local ecosystem services. Significant quantitative and qualitative information, however, was needed to understand how resources and services are represented across the landscape and to evaluate processes that are driving ecosystem loss. This information will be useful in the design and implementation of conservation strategies relating to PES, and “Reducing Emissions from Deforestation and forest Degradation” (REDD) in a spatial context. Of immediate concern, however, is setting in place mechanisms to secure sustained supply of ecosystem goods and services needed to support national and regional development.

Protecting water resources is a priority issue in natural resource management and is essential to sustain both human and ecological communities. Both water quality and quantity influences the growth and health of human populations as well as the ecological health (Newson, 1992; Randhir and Hawes 2009) of watershed systems. Non-point source (diffuse) pollution from agriculture and urban and domestic waste discharges continue to be the major problems affecting water bodies (Randhir and Hawes 2009). Globally, soil erosion cost has been estimated to be US$26 billion per year, $12 billion of which was in developing countries (UNEP 1986). In Africa, about 5Mg per hectare of productive topsoil is lost to lakes and oceans each year (Angima et al. 2003) resulting in water pollution. In East Africa, land use and nutrient loading into Lake Victoria has resulted in water quality deterioration (Hecky and Bugenyi 1992). The lack of long-term monitoring data and the complexity of the ecosystem changes, however, have made it difficult to quantify the effects of land use changes on water quality in the region (Verschuren et al. 2002). Kirugara and Nevejan (1996) who studied pollution sources in the Kenyan part of Lake Victoria’s catchment reported that phosphorus loadings from inorganic fertilizer use varied between 5,000 tons and 22,000 tons per year whereas animal manure contributed approximately 3,000 tons per year, and domestic waste just 100 tons per year. By contrast, nitrogen loading from inorganic fertilizers varied between 4,000 tons and 22,000 tons per year compared with 40,000 tons per year generated from organic manure alone. Importantly, these results are a conservative estimate of diffuse water pollution based on traditional stock assessments using simple linear models.

In a study of soil loss by Moore (1979) in East Africa , the highest rainfall erosivity (R of 200-500) was observed in Uganda, particularly around Lake Victoria, the highlands of Kenya and parts of the coast while the drier parts of Kenya and Tanzania recorded the lowest erosivity (R =150) levels. A more recent study by Lufafa et al. (2003) in the Lake Victoria catchment recorded the highest soil loss in the back slopes (48 tons/ha/yr) followed by the summits (42 tons/ha/yr) areas. A high soil loss was observed in land under annual crops (93 tons/ha/yr), followed by rangeland (52 tons/ha/yr) and minimal in forest and papyrus swamps. These results have implications for water resources and biodiversity conservation with respect to land use change. Sanchez (2002) estimated that annual nutrient loss equivalent of US$4.6 billion in fertilizer replenishment in 37 African countries was a direct result of land degradation that occurred over past 30 years. Geist and Lambin (2002) noted that tropical deforestation is driven by identifiable regional patterns of causual factor synergies, of which the most prominent ultimate causes are the economic factors, institutions, national policies, technological shifts, cultural, demographic factors, and remote influences driving agricultural expansion, wood extraction, and infrastructure extension at the proximate scale.

At a regional level, analysis of forest change over the past 15 years using satellite images showed that over 1,500 km^2 has been lost to agricultural production in the forested areas of the Albertine Rift (Plumptre et al. 2003; Laporte et al. 2003; FAO 2007) constituting 0.5% of the

Albertine Rift Valley area, and 2.2% of its forested area. Much of this loss has occurred outside protected areas (Plumptre et al. 2008) demonstrating the importance of natural resource governance systems toward the protection of ecosystem resources. The remaining protected areas, however, are experiencing degradation of suitable habitats for biodiversity conservation and increased forest loss to anthropogenic activities, including conflict-related resettlement schemes in most parts of the region. Furthermore, the lack of effective governance coupled with corruption, illegal logging, and the lack of national/regional land use plans in place, continue to drive ecosystem degradation and loss.

Both terrestrial and aquatic ecosystems have been affected by land-related developments. Of the aquatic resources, commercial fisheries have been the worst affected. Increased use of wetlands has lead to loss of perennial streams and springs, leading lowering of ground water levels, and loss of aquatic habitat. The hydrologic situation is likely to get worse with the anticipated climate change, whose effects still remain uncertain (Marshall and Randhir 2008). Cultural eutrophication of lakes remains a major concern of the region (Nixon 1988; Simonit and Perrings 2005). Thus, excessive nutrients affect fish productivity through changes in the amount of available food (Bootsma and Hecky 1993) and due to an increase in the volume of deoxygenated water (Hammer et al 1993). The lake system in the Albertine Rift is the heart of the fishing industry, providing a vital source of income and market protein for many people in the region. The increased demand for fish globally and the relative development in fishing technologies has resulted in overexploitation of the fisheries in the regional. In addition, expansion of fishing villages as a result of human population growth, and improved access to remote areas through road construction, demand for water is on the increase, and water quality deterioration is expected due to poor waste disposal. For example, water-borne diseases such as cholera have been recorded.

Vegetation loss was reported to contribute to the decline in rainfall received in various regions, including the sub-Saharan Africa (Gianni et al. 2003; Malhi and Wright 2005). According to Sheil and Murdiyarso (2009), high rainfall occurs in continental interiors such as the Amazon and Congo River basins only because of near-continuous forest cover from interior to the coast, suggesting that localized forest loss can sometimes change a wet continent to arid conditions. Furthermore, climate model simulations have shown that tropical forests have high rates of evapotranspiration, decreased surface air temperature, and increased precipitation compared to pastureland (Bates et al. 2008; Bonan 2008). East African annual climate cycle is controlled by seasonal migrations of the Inter -Tropical Convergence Zone (ITCZ) (Hills 1979; Hesse et al. 1993; Nicholson 2000; Nicholson and Yin 2001) driven by the El Nino-Southern Oscillation (ENSO) cycles (Indeje et al. 2000). The ITCZ develops southward through the Albertine Rift region during the Short Rains (Oct-Nov-Dec) and returns northward during the Long Rains (Mar-Apr-May). Therefore, rainfall is a climatic variable of significant importance in East African countries, with extremely low and high occurrences resulting in droughts and floods respectively. It is anticipated that increased water shortages due to climate change, especially rainfall are projected to increase the number of people affected by water scarcity (Bates et al. 2008). This has been associated with shortages in food, energy and water, disease outbreak, loss of life and property (Indeje et al. 2000).

1.3 Current approaches

For a long time scientists have worked very hard to try and understand the relationship between natural processes and anthropogenic factors influence nature, climate change, however, is likely to increase the complexity of natural systems. It is uncertain how wild species, hydrological processes, ecosystems are likely to respond to climate change effects. Hydrological systems are likely to be influenced heavily by the loss of icefields, glacier melt, land cover change, and ultimately the sea level rise. The Albertine Rift Valley is threatened by land use change and potential climate change effects. Ecosystem goods and services are likely to be impaired and eventually lost, if no mechanisms are instituted to slow down ecosystem degradation and loss. As such, there is a need for careful assessment of the hydrologic process and impacts of land use and climatic change. This information is critical in identifying areas with high sensitivity to land use and climatic stressors and in building capacity of local decision makers. Recognizing the changes in biotic and abiotic processes along multiple and dynamic perspectives provides a framework for understanding watershed ecosystem reconfiguration.

Adaptive planning, integrated modeling, and collective responsibility by all stakeholders impacted by ecosystem change are needed to address complex systems, and to design appropriate management and policy interventions. Accurate seasonal to inter-annual climate change, impact monitoring, and the design of adaptation strategies could therefore contribute to improved planning and management of climate sensitive activities such as water storage and allocation, irrigation, hydroelectricity power generation , biodiversity conservation, fishing, and recreational tourism. In order to develop effective solutions to watershed and biodiversity protection challenges, a systems-based approach is needed to study the dynamic and cumulative effect of land use change on water quality and aquatic ecosystems. Importantly, there is a need for dynamic and integrated assessment of the ecosystem goods and services at an appropriate scale (e.g. watershed) to provide a sound basis for policy decisions, increase conceptual understanding of the systems, and to evaluate spatial and temporal changes in the ecosystems. A systems analysis using an ecohydrologic approach has been applied to study complex watershed systems (Grant 1998; Randhir 2003) elsewhere, and will be applied to this study to allow for meaningful policy option selection.

Increasing use of geographic information systems (GIS) to process digital spatial data, simulation modeling, and faster computational speeds has enabled integrated modeling of complex systems to inform policy decisions. The use of remote sensing and GIS in modeling natural phenomena has simplified the process, added confidence in the accuracy of modeled parameters, and improved information generation needed for planning and management of natural resources (Nearing et al. 2005; Randhir and Hawes 2009). For example, ArcGIS spatial analyst extension tools, and ArcCN-Runoff tool provides the computational capability to assign Curve Number (CN) depending on the soil characteristics, and land use, and antecedent moisture content or initial abstraction for each storm event within a GIS platform (Bhuyan et al. 2002; Zhan and Huang 2004). Satellite-based remote sensing of ecosystem properties, global networks of atmospheric sampling sites, and the development of global models are important new tools to address global issues (Goetz et al. 2005; Field et al. 2007; Bonan 2008).

1.4 Uniqueness of study

Spatial prioritization techniques are applied in conservation planning initiatives as a systematic way of allocating scarce resources. Most of the studies are based on a single dimension relying on the performance of a system or the biology of a species. In this study, however, an integrated systems approach was used to assess how different ecological processes are influenced by various factors ranging from socio-ecological systems suh as civil war, human population, poverty to environmental aspects of the system. In ecological studies, less human dimensions are considered highly and yet humans play a critical role in modifying systems as well as shaping values and attidudes towards a species or process. Besides the multiple dimensional aspects, this study all involved multiple scales namely the regional scale performed at the level of the Congo basin, landscape scale (Greater Virunga Landscape) to local scale (household). For example, human activities such as poaching, human population density, low household incomes that could compromise the conservation of elephants and protection of its habitat due to positive land allocation and management decisions were incorporated in the elephant population dynamics model.

Understanding how water resources and biodiversity respond to changes in land use, climate, and generally human behavior helps to develop integrated instruments, incentives, and institutions that responsive to the desired outcome. In this study, recognition was given to the dimensions of achieving effective policy options that deliver positive benefits from natural resources management. Typically, investment in conservation of nature will depend on how well conservation values, vulnerability of rural communities where conservation is done, economic cost, human and social capital, and species ecology are well understood. This study also dealt with the most critical ecosystem services (e.g. water, biodiversity, carbon) by identifying regions or hotspots where intervention is needed and where prioritization in terms of strengthening conservation is required.

1.5 Study goal and specific objectives

The goal of this study is to assess hydrologic and ecological impacts resulting from land use and climatic changes using integrated modeling with remotely-sensed and spatial data. This information will be used to predict risks to ecosystems under future land use and climatic change scenarios. The analytical process will involve both local (L) and regional (R)-level analysis. The specific objectives for the study are to 1) model the hydrology, water quality and quantity changes in the Congo watershed systems; 2) model the distribution of endemic and threatened species in the Congo watershed; 3) assess the spatial and temporal dynamics of elephant population in a patchy environment (L); (R); 4) assess the extent of ecosystem services at a coarse spatial scale (R); and 5) evaluate the extraction of common pool resources under varying land use and climatic conditions (L). The hypotheses are 1) fluxes in hydrology and water quality processes are significantly different in the watershed systems; 2) species distribution in the Congo subwatersheds is significant different at a spatial scale; 3) elephant population dynamics is significantly affected by changes in habitat, water resources, wars, poaching and climatic conditions; 4) there is a spatial variation in ecosystem services within the subwatersheds; and 5) climate change, resource types and user characteristics have an influence on extraction rates of local Common pool resources

1.6 Ecosystem Services

Ecosystems are the conditions and processes through which natural ecosystems, and the species that constitute them, sustain and fulfill human life. These conditions and processes result in the production of ecosystem goods (e.g. fodder, timber, biomass fuel, natural fibres), and ecosystem services that provide actual life-support functions such as biodiversity protection, carbon sequestration, nutrient recycling, pollination, waste assimilation and detoxification. According to the Millennium Ecosystem Assessment (2003) definition, ecosystem services are the direct and indirect benefits that people obtain from ecological systems categorized as (1) provisioning services (i.e. goods produced or services provided by ecosystems); (2) regulating services (i.e. benefits obtained from regulation of ecosystem services); (3) cultural services are nonmaterial benefits from ecosystems such as spirituality, aesthetics, and recreation; and (4) supporting services are services necessary for production of other ecosystem services, such as soil formation and nutrient cycling (Millennium Ecosystem Assessment 2003; Wilson and Troy 2005). Despite a growing recognition of the importance of ecosystem services (Costanza et al. 1997; Carpenter et al. 2006), their value is often overlooked in local decision-making processes or merely assumed to be zero.

Incorporation of the costs and benefits of maintaining the ecosystem services in national accounts will ultimately have a strong influence on how resource management decisions are made. Many ecosystem services, however, are not adequately captured in the traditional commercial markets, which diminishes the contribution of the environment to national economies. As a result, the benefits that these ecosystem services provide to society are largely unrecorded, and only a portion of the total benefits make their way into economic statistics (Daily et al. 2009). Moreover, for some ecosystem services that do not pass through markets, there is often insufficient incentive for individuals to pay for their protection, let alone internalize the external costs of environmental damage in their investment decisions. Likely, the process of identifying and quantifying ecosystem services is increasingly recognized as a valuable tool for the efficient allocation of environmental resources (Heal et al. 2005; Millennium Ecosystem Assessment 2003). By estimating and accounting for the economic value of ecosystem services, social costs, and benefits that otherwise would remain hidden can potentially be revealed and vital information that might otherwise remain outside of the economic decision making calculus at local, national, and international scales can be revealed and internalized in policy options (Millennium Ecosystem Assessment, 2005). Achieving such an objective, however, requires better understanding of ecosystem services and the landscapes that provide them.

1.6.1 Water resources management in the Congo basin

Studies of input-output budgets of drainage basins have improved our understanding of the interactions between rock weathering, which supplies nutrients and plant and microbial growth, which retains nutrients in ecosystems (Vitousek and Reiners 1975; Driscoll et al. 2001; Falkenmark and Rockstrom 2004). Ecosystem productivity (Saugier et al. 2001; Luyssaert et al.2007), information transfer (Margalef 1968), hierarchical changes in ecosystem controls at different temporal and spatial scales (O'Neill et al. 1986; Peterson et al. 1998; Enquist et al. 2007) and the resilience of ecosystem properties after disturbances (Holling 1973). Land use activities can modify the balance between soil loss and deposition, leading to excessive soil erosion or sediment depositional rates. Forestry (Brooks et al. 2003; Moore and Wondzell 2005; Rashin et al. 2006), urbanization (Paul and Meyer 2001; Randhir 2003; Nearing et al. 2005), and agriculture (Paul and Meyer 2001; Brooks et al. 2003) affect both water quantity and quality. The degradation associated with agricultural production, such as altered sediment loads and an increase in nutrient discharge, is one of the top three causes of loss of aquatic biodiversity (Richter et al. 1997). Land uses change the natural functions of a watershed by impeding or altering the flow of water and impacting water quality (Randhir 2003). Common effects of increased human disturbances in a watershed include increases in water volume, decreases in the reaction time of stream discharge to storm events, increases in runoff affecting stream channels, and decrease in water quality. Although individual land use activities may not appear to have immediate and significant environmental affects, the cumulative and dynamic effects can negatively impact water quality and aquatic ecosystems (Randhir and Hawes, 2009). Land uses such as urban development, grazing, agriculture create impervious surfaces that result in higher surface runoff.

Surface runoff refers to the loss of water from an area by flow over the land surface. It occurs when rain falls with intensity greater than the rate at which it is able to infiltrate the soil. Runoff flow is composed of two main elements: base flow, which has its origin in ground water, and surface runoff, which is the accumulation of rainfall that drains to the stream. The characteristics of a watershed that affect the base flow and runoff include geology, soil type, vegetation cover, mean precipitation, drainage area and antecedent moisture condition (AMC) (Bellal et al. 1996). Runoff behavior is very important for successful design of the rainwater harvesting systems, water budget analysis and management, and in the design of water conservation strategies. Several hydrological models including EPIC (Williams, 1995), and SWAT (Arnold et al. 1996) use the SCS curve number method for estimating storm runoff. The watershed characteristics are considered in determining the curve number (CN) index, which expresses a catchment's response to a storm event (USDA 1986). Ponce and Hawkins (1996) provided a detail account of the conceptual and empirical foundations of the curve number method, emphasizing its wide use in the United States and throughout the world. Hollick (1982), however, cautioned against reporting runoff as a percentage of the annual rainfall due to its shortcomings, in that, it gives no indication of the relationship between runoff and rainfall intensity and duration making it difficult to extrapolate to new areas or drought years. Shanan and Tadmor (1979) warned against the use of annual runoff percentage in the design of microcatchment systems. The Soil Conservation Service (SCS) Curve Number Method (USDASCS 1972), also known as the Hydrologic Soil Cover Complex, is a versatile and widely used procedure for runoff estimation because it gives consistently usable results (Rao et al. 1996; Sharma et al. 2001; Gumbo et al. 2002; Senay and Verdin 2004; Sekar and Randhir 2008).

1.6.2 Water quality and quantity research and monitoring in the Congo basin

Water lies at the core of sustainable development concerns and its efficient and equitable management is crucial for human survival. Watershed systems provide valuable goods and services to support human population and ecosystems (Randhir and Hawes 2009). Freshwater ecosystems have lost a greater proportion of their species and habitat than ecosystems on land or in the oceans, and they face increasing threats from dams, water withdrawals, pollution, invasive species, and overharvesting (MEA 2005; Revenga et al. 2005; Abell et al., 2008). There is increasing need to improve and maintain the structure and function of watershed systems to achieve sustainable resource use, and ensure continuity of ecosystem goods and services. As such, water resource managers are under pressure to implement Integrated Water Resources

The major source of water supply in the region is through precipitation, followed by ground water and to a small extent glacial melt from mountain Rwenzori. Water in the reservoirs is sustained through direct precipitation, stream flow discharge, subsurface, and overland flow. It is however, lost through direct evaporation from water bodies, transpiration from terrestrial and aquatic plants, and animal and human withdrawals or consumption. Because the two major drainage basins in the region, that is, Congo and Nile River are linked to the oceans, water is also lost to the Atlantic, and Pacific Ocean through the Mediterranean Sea respectively.

The presence of icefields on top of the Rwenzori Mountain plays a critical role in catchment recharge and climatic modulation. The Rwenzori are a lofty 75 miles long and 30 miles wide mountain range lying north of the equator on the borders of Uganda and Democratic Republic of Congo. Unlike the other snow peaks of East Africa, the range is not of volcanic origin but is a gigantic uplifted horst-like block of pre-Cambian rocks, and its formation is connected with the complex tectonics of the rift valley (Whittow, 1966). Of the 37 glaciers on the Rwenzori (syn. Ruwenzori), Stanley peak is the highest at 5100 m a.s.l., while Speke glaciers are the most studied (Temple, 1968). During the period 1958-1967, the mean rate of recession of approximately 1.3 m per year was recorded for the lowest section of the Speke glaciers (Whittow et al. 1968; Temple 1968).

Field research carried out in the 1950s (Whittow et al. 1963), early 1990s (Kaser and Noggler 1996) and from 2003 to 2005 (Taylor et al., 2008) indicate that the area covered by alpine glaciers has reduced from 7.5 km^2 in 1906 to <1 km^2 in 2003. More recent studies (Taylor et al. 2009) showed that the rate of decline in glacial extent since 1990 is consistent with the overall trend of approximately 0.7 km^2 per decade since 1906. From moraine evidence, Osmaston (2006) estimated that glaciers covered nearly 260 km^2 between 10 and 20 ka before present (BP) (Last Glacial Maximum) and approximately 10 km^2 between 100 and 200 a BP (Little Ice Age). Glacial recession since the 19th century has led to increased seasonality and overall reductions in riverflow (Mark and Seltzer, 2003; Bradley et al., 2006). Excessive melting was reported to occur if continuous sunshine is experienced, which observations are consistent with those of Howell (1953) who studied the glaciers in the tropical Andes of Peru, and Platt (1966)'s work on Mount Kenya. Temple (1968) also noted that large variations in melt-water amounts were experienced seasonally over periods of a few days and between night and day, and occasionally maintained throughout the night in the ablation seasons. Flint (1959) argued that the cause of glacier retreat was a secular rise in temperature rather than any change in precipitation amounts. There is also evidence to suggest that the decrease in glacial retreat is related to high precipitation amounts associated with greater cloud cover and less insolation and melting, which is also corroborated by Water Development department empirical measurements (daily stage height, and monthly river discharge measurements) data collected on river Mobuku and Bukuju draining from Rwenzori from an altitude of 3400m.

Recent studies by Taylor et al. (2009), however, revealed that the observed acceleration in the rates of termini retreat of the Speke and Elena glaciers since the late 1960s is attributed, in part, to the convex–concave slope profile in which these valley glaciers reside. They argued that current glacial recession has a negligible impact on alpine river flow. Also, spot measurements of meltwater discharges indicate that icefields contribute considerably less than 2% of the river discharge at the base of the Rwenzori Mountains during both dry and wet seasons (Taylor et al.2009). They also noted that an anomalously high specific discharge of the River Mubuku (1730 mm per year) arises from high rates of precipitation exceeding 2000 mm per annum below alpine icefields within Heath-moss and Montane forest ecotones that occupy more than half of the river's gauged catchment area. For other tropical alpine icefields representing a tiny fraction (<1%) of alpine river catchment areas (e.g. Irian Jaya, Kilimanjaro, Mount Kenya), glacial meltwater discharges are similarly expected to contribute a negligible proportion of alpine river flow.

1.6.3 Biodiversity

It has been argued that long-term ecological studies of population dynamics in large mammalian herbivores provide a detailed understanding of the effects of intrinsic and extrinsic factors in determining population size and composition (Saether 1997; Gaillard, et al. 1998; Gaillard et al. 2000). Most studies, however, have focused on the relationships between population density, weather, and individual survival rates of different sex/age classes (Gaillard et al. 2000; Gordon et al. 2004), and very few studies on density-independent effects (Smith and Anderson 1998; Coulson et al. 2001) upon large mammal populations exist. More so, no specific study has examined effects of water supply, habitat quality and climate change on age-specific elephant population structure and size. It is known that malnourished animals are more likely to succumb to severe climatic events at high than at low population densities (Gaillard et al. 1997; Jacobson et al. 2004). More importantly, it has become evident that the impacts of both density-dependent and density-independent effects vary substantially according to a population's sex/age structure. This is because the survival of different sex/age classes is not equally affected by resource availability and weather changes. Adult elephants are likely to be relatively impervious to density and climate change effects, while the calves and juveNiles (and possibly senescent individuals) are highly susceptible to both. Also, calves and juveNiles are more susceptible to naturally-induced mortality while the sub-adults and seniors experience mortality mainly due to poaching, and killings resulting from human retaliation against crop damage and livestock predation. In order to avoid incursions in human dominated areas, and in response to scarcity of food in open dry savanna habitats, elephants are likely to seek refuge in forested habitats and wetland areas. It therefore, seems reasonable to predict that the natality rate of mainly forest and wetland habitat-dependent age groups will vary greatly from the mobile and savanna woodland/grassland-dependent age groups.

1.7 Common pool resources

Managing natural resources for sustainability (i.e. balancing current human wise use with opportunities for the resource use by future generations) is a key priority for managers and policy makers (Klare 2001; Palmer et al., 2004). Common-pool resources (CPR), however, have proved to be a challenge for managers and resource users themselves. Commons refer to institutional devices that entail government abstention from designating anyone as having primary decision-making power over use of a resource (Hess and Ostrom, 2001) whereas common-pool resources have been simply defined as resources available to anyone, making them difficult to protect and easy to deplete (McKean, 2000). The commons is usually used interchangeably with public domain because they both refer to land and allied resources owned by the government or free for anyone to use without monopoly of rights. Common-pool resources are treated as public goods due to the difficulty of developing physical or institutional means of excluding beneficiaries. The resource units possess all the attributes of a private good, one person's consumption subtracts from the quantity available to others making common-pool resources subject to problems of congestion, overuse, pollution, and potential destruction unless harvesting or use limits are devised and enforced.

Enforcement of property rights demands that the state clearly defines rights, roles, responsibilities, and revenues of the resource users, and the management authority. Implicitly, resource user groups are expected to share common interests, share information, collectively monitor and enforce rules and codes of conduct (Hess and Ostrom 2001). As such, it has been argued that smaller groups, presumed to know each other well, understand the resource, and are willing to enforce the property rights, and more likely to manage common pool resources sustainably as opposed to large groups that are highly heterogeneous in group characteristics. Agrawal and Goyal (2001) who examined the group size relationship to successful collective action hypothesis, noted that small groups may not perform any better over a long period of time due to exogenous factors operating beyond their jurisdiction boundaries. For example, construction of roads and railways that open access to remote areas makes policing of resource users from outsiders difficult. Similarly, development of markets for commodities of all kinds, and improvement in resource extraction technologies requires a new set of tools to enforce and monitor resource use, which may not be readily available to the resource user groups irrespective of size. As a response to these challenges, national governments have resorted to decentralization, and privatization of centrally owned and managed forest systems, leading to resource depletion and/or degradation due to excessive logging rates, and rapidly reducing primary growth of forests and pollution of the global atmosphere. Indeed, commodification and privatization of resources is a trend, and a problem to all resources. According to Agrawal and Gupta (2005), participation in community-level user groups is greater for those who are economically and socially better-off.

The sustainability of common-pool resources becomes important as we consider economic and ecological services that are critical to human welfare and wildlife needs. For example, FAO estimated that the world catch of fisheries was worth $80 billion annually (Carr, 1998), and Garcia and Newton (1997) estimated that 200 million people worldwide receive their income from fishing. Common pool resources provide direct benefits such as food, fuel, fodder, herbs, construction materials and income to millions of rural and urban dwellers. Common-pools resources provide a wide range of ecological services and functions enjoyed locally and globally. The ecosystem functions include habitat for wildlife, biological or system properties or process of ecosystems while the ecosystems services consist of flows of materials, energy, and information from natural capital stocks, which when combined with manufactured and human capital services support human lives. Contanza et al., (1997) estimated the average economic value of ecosystem services and natural capital in the world to be US$33 million and indicated that the real value could even double this amount.

In Africa where annual rainfall is low and its distribution is erratic, the products obtained from CPR have been critical elements in the livelihood and survival of many rural communities, particularly in times of drought (Bernus, 1988). At the household level, reliance on common-pool resources have long been a method of assuring household subsistence during hunger periods and in resolving imbalances in the diets of rural households (Williams, 1998). In addition, the sale of products collected from CPR provides an important contribution to household incomes (Bush et al., 2004). For example, the annual bushmeat consumption in DRC was estimated to be 1.7 million tones and worth over US$1billion (UNEP, 2011). The spatial dispersion of common pool grazing resources and the temporal fluctuations in their availability make them an important resource for livestock production in the region. The seasonal use of CPR creates opportunities for mutually beneficial exchange relationships between various user groups, including opportunities to access financial services. The exchanges of grain, crop residue and water owned by farmers for the manure produced by pastoralists' livestock have linked crop and livestock production for many years in the Sahel and served to increase land productivity (Williams et al, 1995).

1.8 Climate change

There is general consensus among scientists that the earth is warming, and as global temperatures increase, the hydrologic cycles are becoming more vigorous and varied world over. The Intergovernmental Panel on Climate Change (IPCC) report that both land and sea surface temperatures have significantly increased by 0.4 to 0.7oC in the past century (IPCC 2007). The IPCC also report that there has been a very significant increase in precipitation during the 20th century in the mid-to-high latitudes of the Northern Hemisphere (IPCC 2007). Much of the increase in precipitation that has been observed worldwide has been in the form of heavy precipitation events (IPCC 2007). Climate models are predicting a continued increase in intense precipitation events in some regions and decrease in drier regions of the world over the next hundred years (IPCC 2007). In the Albertine Rift, there is evidence of rising temperatures in the region leading to retreat of glaciers at the summit of the Rwenzori Mountains documented since the early 1900s (Taylor et al. 2006, 2008, Eggermont et al. 2007; Taylor et al. 2008; Russell et al. 2009).

1.8.1 Climate effects on water resources

Soil erosion rates may be expected to change in response to changes in climate for a number of reasons, the most direct of which is the change in the erosive power of rainfall (Pruski and Nearing, 2002). Soil erosion responds both to the total amount of rainfall and to differences in rainfall intensity, however, the dominant variable appears to be rainfall intensity and energy rather than rainfall amount alone. One study predicted that for every 1% increase in total rainfall, erosion rate would increase only by 0.85% if there were no correspondent increase in rainfall intensity (Pruski and Nearing 2002). However, if both rainfall amount and intensity were to change together in a statistically representative manner predicted erosion rate could increase by 1.7% for every 1% increase in total rainfall (Pruski and Nearing 2002). Higher temperatures may translate in higher evaporation rates, while more rainfall would tend to lead to higher soil moisture levels. Even changes in soil surface conditions, such as surface roughness, sealing, and crusting, may change with shifts in climate, hence impacting erosion rates. Finally, if farmers react to climate change by implementing different crops, crop varieties or even change land use patterns, the erosion and deposition rates and patterns within catchments may change completely and therefore net soil loss may change as well.

1.8.2 Climate change effect on biodiversity

Numerous reports especially those of the Intergovernmental Panel on Climate Change (IPCC 2001; 2007, 2013) and other peer-reviewed publications have indicated that earth's climate is changing as witnessed by higher atmospheric temperatures, decreased snow and ice cover, including retreat of many mountain glaciers in non-polar regions and increasing sea levels particularly toward the end of the 20th century. According to the IPCC Assessment Report five released in 2013, global surface temperature for the 21st century is likely to exceed 1.5 ^{o}C relative to 1850 to 1900 for all Representative Concentration Pathways (RCP). The global mean precipitation is also anticipated to increase at a rate of one percent to 3% per degree increase in temperature for all scenarios other than RCP2.6 (IPCC, 2013). Basic ecological and physiological processes in natural systems are strongly influenced by extreme conditions of weather and climate, particularly sudden shifts in precipitation and temperature thresholds. Scientists have used the current distribution of individual species to construct climate envelope models of threshold values for temperature and precipitation variables within which the species is able to survive and reproduce to estimate geographic areas that will be climatically suitable for species persistence under a variety of climate change scenarios. Also, the species climate envelope models have provided some cues on the species extinction risks.

Depending on how long these extreme climatic conditions persist, this will have profound and long-term effects on the physical substrate and biological conditions suitable for species to survive and reproduce. Based on the available information, biological species are expected to respond differently. Species that do well under the optimal range of the environment and resource gradients may be affected most as opposed to species that have a relative degree of tolerance to physiological stress. Indeed, it is too early to tell with certainty how species will respond to these extreme conditions. For example, grasses and some alpine meadows may be unable to find patches with suitable soil properties at high elevations, while insects and mammal herbivores that feed on those grasses may be able to move to mountainous areas with higher meadows. Also species that exhibit different life cycles such as spawning in fish, nesting in birds, embryonic and sex development in reptiles that occurs at maximum temperatures, body development in some bird species during extremely wet periods will be affected at various levels.

Evolutionary and morphological changes as a result of climate change will mostly likely depend on the environmental gradient, and genetic differences at the individual, species, or community level. One of the ecological theory postulates that tremendous success of some invasive species is attributed to their release from predators or diseases, and pathogens which significantly decrease selective pressures to allow the organism to lose many of its anti-predator adaptations. Most mammals respond to competition (intra- and interspecific), and predation by reducing their growth or reproduction rates, and body size in order to cope under extreme climatic conditions. In fact, some species may fail to breed at all. Wide-ranging species (e.g.

salmon, turtles, snakes, butterflies) that depend on different habitats for different phases of their life cycles will not be able to survive in highly fragmented and hydrological disconnected ecosystems. Equally, species with poor or low dispersal abilities will not move to other patches reducing the colonizing ability whereas species with specialized resource and habitat requirements will become vulnerable to extinction.

In plants, cyclic events associated with seasonal changes such as species and plant phenological events may be altered. Seed germination and dispersal may be delayed or flowers become sterile. These changes will be associated with dramatic changes in predator-prey relations, migration patterns, breeding cycles, and mortality rates. Pests and disease outbreaks or re-emergency is expected, and species resistance to diseases is likely to deteriorate since the animals will be already physically weakened, unless such an organism has genotypes and alleles that confer immunity to that specific disease. Similarly, species abundance, richness and diversity, community structures and composition will be affected as a consequence of climate change impacts. Home ranges of species especially birds, landscape species such as elephants, lions are predicted to shrink, and for the most part species are expected to move toward poles and higher elevations. Species are predicted to shift southwards during cool periods and northwards during warm periods which is to some degree consistent with global air circulation movements.

One of the major effects of climate change on coastal wetlands is sea level rise. If the sea level continues to rise at such a high rate (1.7 cm/yr), some of the most important wetlands designated as Ramsar sites will be lost, and as a result, avian, herpetofauna, fish, wildlife and plants that are adapted to this type of environment will be dramatically reduced or driven to extinction. Salt intrusion and subsidence of relatively dry uplands and salt marshes shift inland will dramatically influence the distribution and patterns of plant communities with a likelihood of salt marsh species migrating upwards and the subsequent loss of freshwater species. Since, wetlands significantly store more carbon, particularly in soils than in the atmosphere, there is the potential that if the climate were to warm and accelerate decomposition of peatlands, then these peatlands would become an additional major source of carbon through aerobic respiration and possibly fires, into the atmosphere. Warm conditions will accelerate the conversion of methane by the bacteria into carbondioxide and carbon through oxidation process especially in wet-dry pulsating wetland sites. Therefore, an increase in global warming leading to sea level rise could easily turn wetlands from carbon sinks to net sources of carbon, but also affect the species community structure (composition, distribution and abundance).

1.8.3 Climate change effects wildlife

Hot, arid environments may impose restrictions on food and water availability, as well as exposing animals to large fluctuations in ambient temperature. Therefore, large homeothermic animals that inhabit these habitats are expected to have adaptations that deal with these environmental stresses. These can include a lower average (mesor) Tb and adjustments to the Tb daily rhythm, which may enable an animal to reduce energy consumption and water loss (Dawson, 1955). Kinahan et al. (2007) who studied the body temperature daily rhythm adaptation in African savanna elephants, observed that elephants had lower mean Tb values (36.2 ± 0.49 °C) than smaller ungulates inhabiting similar environments but did not have larger or smaller amplitudes of Tb variation (0.40 ± 0.12 °C), as would be predicted by their exposure to large fluctuations in ambient temperature or their large size. According to Buss (1961), during the dry season, elephants concentrated near permanent water supplies where some plants were utilized intensively, and used woodlands primary for day time concealment and shade, but not as a principal source of food. With the onset of the rainy season, elephants returned to savanna lands where water was then available and extensive grasslands provided an abundance of their favored foods (Buss 1961; Field 1971). Stream distribution and water yield in the Albertine Rift, specifically the Greater Virunga Landscape on elephant population dynamics is not well known. It is, however, a known fact that elephants depend heavily on water for cooling off, consumption and, stream channels provide migration routes for elephants to other habitats or landscapes in the region. In terms of reproduction and body physiology, there exists a direct relationship to precipitation cycles attributed to reduced stress due to food abundance (Laws 1968; Field 1971; Kinahan et al. 2007; Eltringham 2008). On average, an adult elephant consumes between 200-250 litres of water a day (Buss 1955; Barnes et al. 1999).

1.9 Dissertation Plan

The dissertation is presented as seven major chapters namely: 1) Introduction, 2) Biodiversity values of the Congo basin, 3) Congo watershed analysis and hydrological modeling, 4) Age-class specific elephant population dynamics: influence of climate, wars, and poaching, 5 Ecosystem Services, 6) Extraction of commons, and 7) conclusions and recommendations. The first chapter introduces the problem and background on the ecosystem services focusing mainly on watershed, carbon sequestration, and biodiversity values at a large scale. It further describes climate change impacts on species, water resources, and the provision of ecosystem services at a a local, landscape, and regional scale. In chapter two, watershed and hydrological processes occurring in the Congo basin are discussed in more detail. The chapter presents results on water quantity and quality and further discusses the policy strategies for improved management of the Congo basin. Chapters four focused more on the impacts of anthropogenic effects specifically civil wars, poaching, and climate change on elephant population dynamics in Greater Virunga

Landscape. This is a species level analysis performed at a landscape scale. The impacts of these stressors on age class-specific population dynamics are presented in individual subsections, including policy options investigated and their implications for managing the species. Chapter five describes the ecosystem services of the Congo basin and also discusses the policy implications of using a single objective criterion such water resources management, carbon sequestration, and biodiversity on other ecosystem values. In chapter six, human behavior in terms of extraction and consumption of common pool resources is assessed. The chapter also identified the key income strategies, highlighting the importance of common pool resources to supporting rural livelihoods. Conclusions and recommendations are presented in chapter 7.

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CHAPTER 2

BIODIVERSITY VALUES OF CONGO BASIN

2.1 Introduction

The distribution and abundance of biodiversity derived from models or extrapolations based on field observations is very critical to conservation planning. The term “biodiversity” is used in a broad sense as it is defined in the Convention on Biological Diversity (Leadly et al. 2010) to mean the abundance and distributions of and interactions between genotypes, species, communities, ecosystems and biomes. The combined effect of land use change, overexploitation of forests and aquatic resources, increasing concentrations of greenhouse gases leading to climate change, modification of river flow and pollution of water bodies are projected by IPCC (2013) models to result in significant changes in the distribution and abundance of species, and ecosystems. Most of these biodiversity changes will involve large and sometimes highly visible modifications of ecosystems such as widespread conversion of tropical forest to pastures and croplands, reductions in the abundance of top predators in marine systems. Some species are projected to increase in abundance or expand their ranges, but the abundance or range size of many species will decline, often leading to substantially increased risk of extinctions.

Congo River is the largest river in the Central Africa and the second longest (4,374 km) in Africa (after the Nile). The Congo drains the vast Congo River Basin, an area of more than 3.7 million sq km (Tchimanga and Hughes 201), and at high water periods, discharges approximately 34,000 cubic meters (1.2 million cubic feet) of water per second into the sea. Its width varies with location and time of the year but ranges from 0.8 to 16 km. The Congo River has four sections: the headwaters, Upper Congo, the Middle Congo and the Lower Congo flowing through the countries of Democratic Republic of Congo, the People's Republic of the Congo, the Central Africa Republic and partially through Zambia, Angola, Cameron and Tanzania. The Congo basin is very rich in biodiversity due to a relatively large area of forest (approximately two million hectares). The forest ecosystems of the Congo basin span across much of Central Africa, from the Atlantic Ocean's Gulf of Guinea to the mountains of the Albertine Rift in the east. Covering 700,000 square miles in six countries, they constitute the second largest area of contiguous moist tropical forest left in the world and represent approximately one fifth of the world's remaining closed canopy tropical forest. Six percent of the Congo basin is protected and the area in DRC alone, the protected area increased from nine to twelve percent of the national territory (UNEP, 2011). This vast area hosts a wealth of biodiversity, including over 10,000 species of plants, 1,000 species of birds, and 400 species of mammals, 280 species of reptiles, and three of the world's four species of great apes (CBD, 2009).

2.1.1 Problem

In the Congo basin, over six percent of the watershed is forested and allocated to conservation of wildlife under protected area status. The protected areas, however, are not free from agricultural encroachment, threats from resource extraction by local communities, including timber and minerals. Mixed agriculture dominates other land use activities done outside protected areas followed by livestock grazing. Mixed agriculture or integrated crop-livestock systems are defined as those land use systems in which crop and livestock production are managed by the same economic entity such as a household where crop inputs are also used in livestock production. It has been argued that as population pressures increase and/or new market opportunities emerge, farmers adopt more labour intensive agricultural land management practices (Verburg et al. 2009). In this region, however, farmers are too poor to afford new technologies and the high return to labour from the energy sector and off-farm businesses has resulted in scarcity of farm labour forcing farmers to abandoned degraded land in such for virgin parcels of land. It has also been noted that intensification of agricultural productivity can provide improvement in food security (Tilman et al. 2011), but without incentivizing farmers to change from traditional land practices to sustainable land use (Tilman et al. 2002), there is an increased risk of soil fertility loss.

Land and water resources in Congo basin face unprecedented challenges from natural disasters, climate change, development pressures, and population growth. More importantly, oil and gas development, a new land-based development in the region has come on board. Appropriate use of public lands for economic and energy development while maintain the stocks and flow of ecosystem services requires a keen appreciation of how the landscape is changing over time and the causes of, and impacts of those changes. Wildlife species in the Congo basin and Africa in general are declining due to habitat loss and degradation, bushmeat hunting, killings associated with civil wars and human-wildlife conflicts. For example, the elephant population in DRC dropped from 62,000 in 2002 to 23,000 in 2006 (UNEP, 2011) mainly due to poaching and killings during civil war. Detailed analysis of different land use policy impacts on biodiversity values at varied time and spatial scales is very important. Furthermore, understanding the interaction between ecosystem services and human well-being across different time and spatial scales is an important step toward generating feasible solutions. Identification of land use land cover dynamics at broad scales followed by identification of hot spot areas that are highly sensitive to land use change is very necessary. It is anticipated that migrant population grows and the expected increase in real incomes of much of that population likely to find employment in oil and gas sector will have manifold consequences on the demand and supply of ecosystem services in the region. It is fair to speculate that these foreseeable demographic and economic trends will call for a doubling of local and regional demand for food, wood, fibres, fuel and water among other welfare needs. While such expansion may well be within the capacity of the regional resource base and technical capability, it will require considerable financial and human resources to remedy the ecological impacts thereof. In addition, anticipated social and economic changes will alter the conditions under which land managers operate and new resource conflicts are likely to emerge. The need for rational management of conflicting goals, uncertainties and risks using varied applications of science-based support to solve complex problems (Makowski 2009). Therefore studies of this nature are timely and could help to provide evidence-based policies that will aim to maximize stakeholder's benefits while minimizing potential conflicts in future.

2.1.2 Current approaches

Species distribution models (SDMs) are numerical tools that combine observations of species occurrence or abundance with environmental estimates (Elith & Leathwick, 2009). A large number of algorithms have been used in species distribution modeling ranging from profile, domain, regression, to machine learning methods. The most obvious difference in application of these methods, however, is the type of data used. Profile methods only use presence only data while machine learning and regression methods use both presence and absence or pseudo absence (also referred to as background data). BIOCLIM, which is a classical example of profile methods extensively used in species distribution modeling, computes the similarity of a location by comparing the values of environmental variables at any location to a percentile distribution of the values at known locations of species occurrence. BIOCLIM algorithm follows Sprengel–Liebig Law of the Minimum (Hackett, 1991; Van der Ploeg et al., 1999; (Hijmans & Graham, 2006) where the most limiting factor determines the overall response. The closer to the 50th percentile (median), the more suitable the location is. The distinction between machine learning and regression methods has been discussed extensively by many authors (Thruiller and Elith & Leathwick, 2009).

All species modeling methods have their own strength and weakness and their performance depends heavily on the quality of occurrence data being free from bias and errors (Graham, 2009) and large number of observations (Pearson, Raxworthy, Nakamura, & Townsend Peterson, 2007; Wisz et al., 2008). For example, both BIOCLIM and Domain algorithms were noted to be less useful when applied in climate change effect modeling (Hijmans & Graham, 2006; Pearson et al., 2007; Elith & Leathwick, 2009). The Domain algorithm computes the Gower distance between environmental variables at any location and those at any of the known locations of occurrence or training sites (Hernandez, Graham, Master, & Albert, 2006). The distance between the environment at a given point and those of the known occurrences for a single climate variable is calculated as the absolute difference in the values of that variable divided by the range of the variable across all known occurrence points. Irrespective of the above mentioned shortcomings, SDM have been very useful in providing ecological and evolutionary insights of species and prediction of their spatial and temporal distributions across landscapes. The remaining challenges are: improvement of methods for modeling presence-only data and for model selection and evaluation; accounting for biotic interactions; and assessing model uncertainty.

2.1.3 Justification of the study

Available empirical evidence, however, suggests that many primate populations are severely threatened by anthropogenic actions and the African continent is of particular concern in terms of global primate conservation (Chapman et al.2006). The main threats to species conservation in central Africa include deforestation (Martin 1991). According to Mayaux et al. (2005), the African forests are currently being converted at a rate of 0.4 to 0.5% per year and are commercial logging and clearing for agriculture being the main cause of deforestation. Despite the low levels of forest clearing involved, small-scale, shifting agriculture and the collection of natural resources by rural populations also contribute to forest loss and degradation of habitats. Similarly, the Congo Basin is known to be the biggest centre for bushmeat hunting than anywhere else in the world. Central Africa in particular, bushmeat contributes up 80% of people's protein intake (Pearce, 2005) and local consumption of bushmeat in DRC is high and 57% of the wild animals hunted are primates (Lahm, 1993). Commercial hunting has been reported to be on the increase in the past decade (Oates, 1996; Wilkie, 2000). There is also the fear among conservationists that with the advent of climate change, species population, already stressed by habitat shrinkage and hunting, will decline catastrophically. The good news is that research scientists, academics, and the donor community are gaining interest in understanding the impacts of climate change on biodiversity (Hannah et al., 2002; McClean et al., 2005). Most recent field surveys done in the region have concentrated more in the Albertine Rift and specifically so because this is the core conservation area for Wildlife Conservation Society that is highly commended for using scientifically researched information to inform conservation planning and management. Furthermore, the Congo basin is relatively insecure due to the protracted civil war making it difficult to conduct research in the region. The available information concerning the distribution and abundance of fauna and flora scanty and outdate, but also very difficult to find. Initiatives to house all field data from museum collections, herberiums, and research institutes is being done and centralized in defined locations such as the Global Biodiversity Information Facility, IUCN & Birdlife International avianfauna database, FAO's Fishbase, HerpNET, A.P.E.S database, African Mammals Databae, among others. In order to support conservation planning, this data needs to be transformed into usable information, particularly to identify potential areas for species colonization. Thus, knowledge of a species distribution in the Congo basin can provide an idea of where investment in biodiversity conservation should be targeted.

2.1.4 Objectives

The study aim was to identify potential species distribution sites where conservation managers should ensure to invest their resources. Specific objectives of the study were to: i) develop species distribution models for selected endangered and threatened species in the Congo basin; and ii) identify areas with high concentrations of biodiversity values.

2.1.5 Hypothesis

The key hypothesis was that the distribution of selected endangered and threatened species for the Congo basin is uniform.

2.2 Literature review

2.2.1 Biodiversity species

The Congo basin is widely known for habituating over 400 species of mammals. According to (White, 1983; Lebrun 2001) the Guineo-Congolian's two main blocks of the African rainforest found located in Cameroon and Gabon's portion of the Congo basin has the highest (8000) species richness. The Odzala National Park (DRC) was noted to have a high diversity and species-specific abundance of primates, and has the highest number of diurnal primates (10 species) in the forest block of Central Africa (Bermejo 1999) of highest diversity of plant species primate species is. The Congo River Basin is well known for its large population of forest elephants. For example, in Dzanga-Ndoki National Park in the Central African Republic, Nouabale Ndoki National Park in Congo-Brazzaville and Langoue in Gabon, these elephant populations remain relatively undisturbed. The bonobo or pygmy chimpanzee (*Pan paniscus*) the sun-tailed monkey (Cercopithecus solatus), black colobus (Colobus satanas) and the okapi (Okapia johnstoni) are only found in the Congo River Basin. The Congo basin is also known for the high diversity of plant species, insects, reptiles, amphibians, and small mammals. Most of the species occur inside the protected areas. Six percent of the Congo basin is protected (Figure 2.1)

2.2.2 Endemic and threatened species of the Congo basin

According to the IUCN Redlist, the Congo basin is has several mammal species that occur nowhere else namely the Bonobo, Grauer's gorilla, Black Mangabey, Golden-bellied Mangabey, Thollon's Red Colobus. The list of endangered and threatened species selected for modeling is provided in table 2.1. A brief mention is made of the most notable species in each taxa. For example, among the avianfauna, these include Congo Peacock, Congo Bay-owl, Schouteden's Swift, Itombwe Nightjar, Prigogine's Greenbul, Yellow-crested Helmet-shrike, Yellow-legged Weaver, Golden-naped Weaver, Chapin's Mountain Babbler, Black-lored Waxbill among others are restricted to this region. Reptiles unique to the Congo basin include the Upemba Bush Viper (*Atheris katangensis*), rough-scaled lizard (Ichnotropis chapini), Marungu Girdled Lizard

(*Cordylus marunguensis*) among others. Endemic amphibians include five species of frogs in the genus of Hyperolius namely *Hyperolius sankuruensis*, *H. robustus, H. schoutedeni, H. leucotaenius and H. constellatus*, Itombwe River Frog *Phrynobatrachus asper*, Kivu Screeching Frog (*Arthroleptis pyrrhoscelis*: Eli Greenbaum), the Lendu Plateau Clawed Frog (*Xenopus lenduensis*), the Itombwe Massif Clawed Frog (*Xenopus itombwensis*), and the Itombwe Golden Frog (*Chrysobatrachus cupreonitens*). Among the fresh fish, these include the Congo Blind Barb Caecobarbus (IUCN), a bagrid catfish Gnathobagrus and the cichlids Schwetzochromis (Cichlid Room Companion) and Teleogramma (FishBase).

2.3 Methodology

2.3.1 Study area

The Congo basin covers over 3.7 million square kilometers and includes the world's second largest tropical forest of approximately two million square kilometers (Hansen et al., 2009; Bwangoy et al., 2010). The Basin's forest zone includes parts of six central African countries: Cameroon, Central African Republic, Congo, Equatorial Guinea, Gabon, and the Democratic Republic of the Congo. The study area includes the core Congo River basin, located within the forest zone from 5° north to 6° south and from 13° to 26° east (Figure 2.1). The climate is warm and humid with two wet and two dry seasons with a mean temperature of nearly 25 °C and average rainfall if 1800 mm per year (Bwangoy et al., 2010). The climax ecosystem is tropical evergreen forest with little seasonal variation typified by irregular and very dense (70–100%) crown cover that often preclude the development of understory shrubs and grasses (Mayaux et al., 2004). Congo basin is one of the world’s important biodiversity areas highly recognized for habiting the lowland gorillas, primates (Brooks et al., 2006) and elephants

(Blanc et al., 2002). The total human population in the Congo Basin region was estimated at over 73 million people with an annual growth rate of >2.5% (Ndoye and Tieguhong, 2006) resulting in habitat fragmentation and loss due to agricultural expansion and unsustainable resource extraction. Other major threats to the ecosystem benefits include commercial trade in bushmeat and endangered species contributing to biodiversity loss.

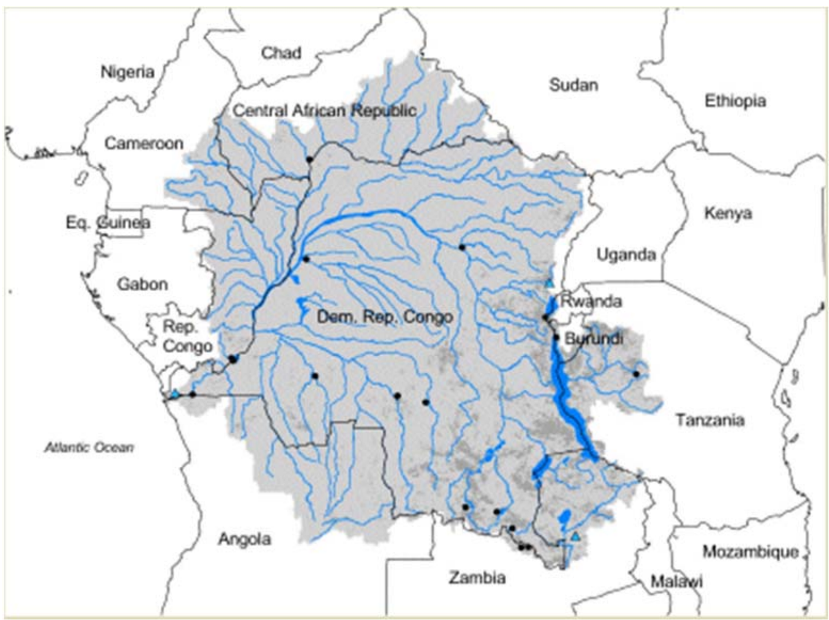

Figure 2.1 Map showing the Congo basin and overlapping countries

2.3.2 Conceptual model

In order to implement the species distribution models, the GIS processing followed the framework shown in Figure 2.2. The species selected for modeling were endangered and threatened species listed under the IUCN Redlist. A detailed description of the modeling procedure is discussed in the subsequent sections under this chapter.

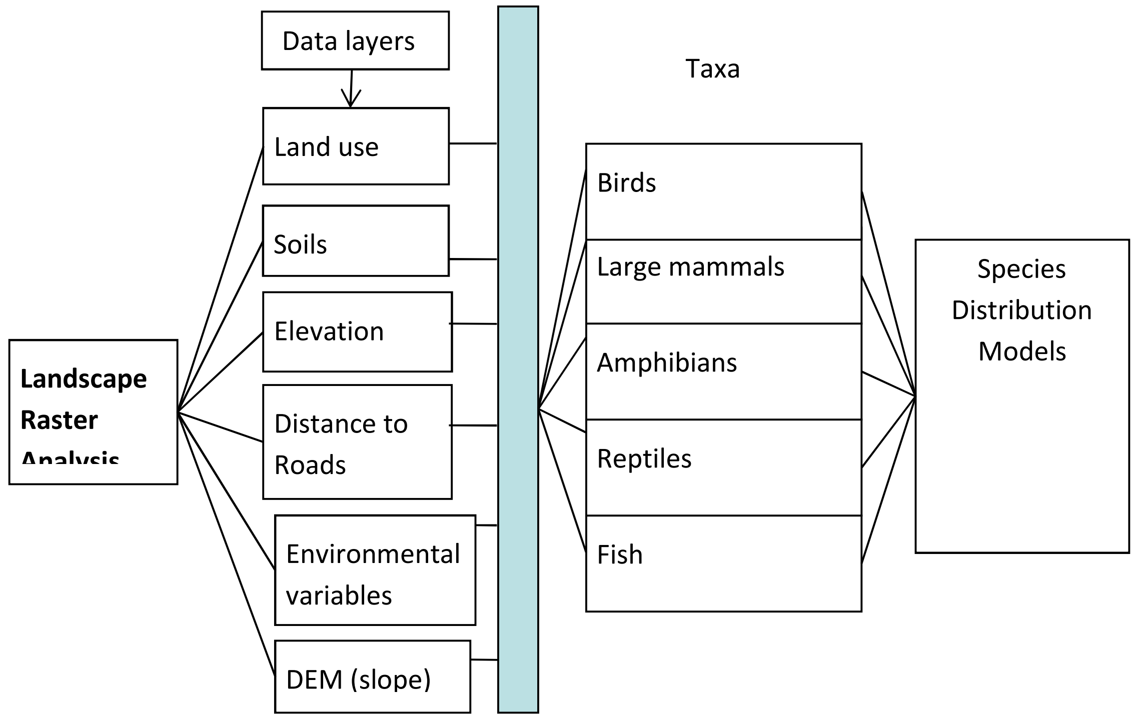

Figure 2.2 Species distribution modeling framework

2.3.3 Biodiversity data

Biodiversity data was needed in order to produce the biodiversity values for the Congo basin. There is a reasonable quantity of data on mainly large mammals, birds, and plants, but very scanty for fish, amphibians and reptiles. Importantly, most of this data is outdated, scattered, multi-scale, and cover only selected areas. In order to implement this study successfully, the distribution of key landscape species from five taxa: birds, large mammals, fish, reptiles, and amphibians were necessary. The spatial distribution of an organism forms a fundamental basis setting conservation priorities and land use policies (Anderson et al., 2009). The spatial overlap between habitats important for providing different ecosystem services and biodiversity benefits is of critical importance in designing strategies that maximize both conservation benefits and ensure sustained livelihoods. Majority of the species selected for modeling under each taxon had very few presence-only points. As such, the choice between using BIOCLIM and Domain methods that use presence only data and other methods such as logistic regression that use both presence and absence data or background points had to be made. Predicting species distributions from small numbers of occurrence records was discussed by Pearson et al. (2007) who noted that the performance of the models differed depending on the locations and observations included in the model. In their study, Pearson et al. (2007) concluded that models developed using small sample sizes as low as five should be interpreted as identifying regions that have similar environmental conditions to where the species is known to occur and not as predicting actual limits to the range of a species. To provide informative predictions, Anderson et al. (2003) noted that it is necessary for a model to successfully predict a high proportion of test localities (i.e. have a low omission rate) while not predicting as suitable a large proportion of the study area as to make the model statistically indistinguishable from a random prediction. In order to overcome data limitations, data re-sampling has been proposed as a plausible solution. According to Estrella et al., (2012), data re-sampling is not the best strategy. In order to overcome this data constraint, stratified random sample of pseudo-absence points (Phillips et al., 2009) were generated using HawthsTools (Beyer, 2004) in ArcGIS. The use of background data helps to establish the environmental domain of the study while presence data is used to establish conditions under which a species is more likely to be present than on average. On the other hand, true absence data coupled with presence records establishes where surveys have been done and the prevalence of the species given the survey effort. The only challenge with this approach, however, is how to select pseudo-absence localities for model parameterization for presence-only distribution modeling.

van DerWal et. al. (2008) suggested that it is important to consider the spatial extent from which pseudo-absence are taken, and concluded that 200 km was an optimal distance to stray in search of appropriate pseudo-absence points. On the other hand, Barbet-Massin et al. (2011) noted that pseudo-absences selected with a geographical exclusion of 2^{o} far' yielded significantly better models with few presences. Most of the birds, fish, amphibians, and reptiles taxa species selected for modeling had less than 1000 presence points. As such, 1000 pseudo-absence points were generated for all species instead of 10000 random pseudo-absence points recommended by Barbet-Massin, et al. (2011) and applied by Estrella et al. (2012). On the contrary, Barbet-Massin, et al. (2011) noted 1000 random pseudo-absence points yield reliable results if a generalized linear model (GLM) is used and both the presence and absence points are equally weighted in the modeling environment. In the case of large mammal species with relatively higher numbers (e.g. >10,000) of presence points, 10000 pseudo absence points were used. In order to increase of randomly sampled points, species range maps from IUCN Redlist database were used to constrain the region within which to extract the points.

2.3.4 Species selected for modeling

The number of species under each taxon that were selected for modeling is provided in Table 2.1 including the total number of spatial coordinates for the surveyed area where the species was recorded present. Five taxa were identified and species that are endangered or threatened were considered for modeling and each of these species had to have a minimum of five field observations taken from five different sites.

Table 2.1 Species selected for modeling under reptiles, amphibians, fish, birds and large mammals.

	Amphibians				
	Scientific Name	Comonn name	Survey points	Pseudo absence	comments
1	*Afrixalus equatorialis*	Congo banana Frog	13	1000	
2	*Cardioglossa cyneospila*	Long Finged Frog	48	1000	
3	*Hyperolius langi*	Lang's Reed Frog	9	1000	
5	*Phrynobatrachus asper*	Itombwe River Flog	21	1000	
7	*Leptopelis karissimbensis*	Common frog	18	1000	
			Birds		
1	*Afropavo congensis*	Congo Peacock	15	1000	
2	*Cossypha archeri*	Archer's-Ground-Robin	680	1000	
3	*Alethe poliophrys*	Red-throated-Alethe	187	1000	
4	*Apalis Kaboboensis*	Kabobo Apalis	51	1000	
5	*Apalis personata*	Montane-Masked-Apalis	1249	1000	
6	*Cynonomitra alinae (syn. Nectalinia alinae)*	Blue-headed-Sunbird	303	1000	
7	*Glareola nordmanni*	Black-winged-Pratincole	82	1000	
8	*Apalis ruwenzori*	Collared apalis	730	1000	
9	*Coracina graueri*	Grauer's cuckoo-shrike	25	1000	
10	*Graueria vittata*	Grauer's Warbler	244	1000	
11	*Bradypterus graueri*	Grauer's Rush-warbler	58	1000	
12	*Gymnobucco sladeni*	Sladen's barbet	20	1000	
13	*Gyps Africanus*	African-white-backed-vulture	301	1000	
14	*Apalis argentea*	Kungwe apalis	55	1000	
15	*Kupeornis chapini*	Chapin's Babbler	11	1000	
17	*Hemitesia*	Neumann's	256	1000	

	neumanni	Warbler			
18	*Parus fasciiventer*	Stripe-breasted-Tit	115	1000	
19	*Nectarinia purpureiventris*	Purple-breasted-Sunbird	246	1000	
20	*Chlorocichla prigoginei*	Prigogine's greenbul	9	1000	
21	*Prionops alberti*	Yellow-crested Helmetshrike	37	1000	
22	*Cinnyris regius*	Regal Sunbird	624	1000	
23	*Phylloscopus laetus*	Red-faced-Woodland-Warbler	761	1000	
24	*Cinnyris stuhlmanni*	Ruwenzori-Double-collared-sunbird	219	1000	
25	*Tauraco johnstoni*	Ruwenzori Turaco	789	1000	
26	*Batis diops*	Ruwenzori Batis	369	1000	
27	*Cryptospiza shelleyi*	Shelley's Crimson-wing	9	1000	
29	*Ploceus alienus*	Strange Weaver	175	1000	
30	*Terpiphone Bedford*	Paradise fly catcher	7	1000	
	Fish				
1	*Caecobarbus geertsii*	Congo Blind Barb	8	1000	
2	*Ancanthocleithron*	Congo squeaker	57	1000	
3	*Marcusenius schilthuisiae*	Barbus Jansensi	22	1000	
4	*Microctenopoma intermedium*	Blackspot climbing perch	63	1000	
	Large Mammals				
1	*Philantomba monticola*	Blue Duiker	112	10000	
2	*Cercopithecus mitis*	Blu Monkey	776	10000	
3	*Syncerus caffer*	Buffalo	4807		10000
4	*Potamochoerus larvatus*	Bush Pig	267		10000
5	*Tragelaphus scriptus*	Bush back	614		10000

6	*Pan troglodytes*	Chimpanzee	19418		10000
7	*Loxodonta Africana*	Elephant	28968		10000
8	*Rhinopithecus roxellana*	Golden Monkey	252		10000
9	*Graueri beringei*	Eastern lowland gorilla	4896		10000
10	*Alcelaphus buselaphus*	Hartebeest	8460		10000
11	*Crocuta crocuta*	Hyaena	89		10000
12	*Canis adustus*	Side-striped Jackal	97		10000
13	*Panthera pardus*	Leopard	358		10000
14	*Cercopithecus lhoesti*	L'hoest Monkey	2295		10000
15	*Panthera leo*	Lion	639		10000
16	*Okapi johnstoni*	Okapi	309		10000
17	*Tragelaphus spekeii*	Sitatunga	98		10000
18	*Kobus ellipsiprymnus*	Waterbuck	1094		10000
Reptiles					
1	*Atheris nitschei*	Great Lakes bush viper	17		1000
2	*Crocodylus cataphractus*	slender-snouted crocodile	8		1000
3	*Chamaeleo ituriensis*	Ituri Forest Chameleon	18		1000
4	*Chamaeleo johnstoni*	Johnston`s Chameleon or Ruwenzori Three-Horned Chameleon	25		1000
5	*Crocodylus niloticus*	Nile Crocodile	17		1000
6	*Leptosiaphos graueri*	Rwanda Five-toed Skink	11		1000
7	*Philothamnus ruandae*	Rwanda Forest Green Snake	12		1000

Only species categorized by IUCN as endemic, endangered, critically threatened, and vulnerable were considered.

Species distribution models (SDMs) were done based on presence/absence data using a generalized linear logistic model in R statistical package. The SDMs are used to produce predictions of potential distribution of species within their environmental niche. A number of predictive SDM packages such as BIOCLIM, BIOMOD, MAXENT, and reference papers related to modeling methods such as generalized linear model (GLM), Generalized Additive Model (GAM), Classification Regression Trees (CART), and Artificial Neural Networks (ANN) were extensively reviewed by Franklin, (1995), Guisan and Thuiller (2005), and Garcia, et al. (2011). In this study, the GLM was used due to the nature of the data (i.e. presence/absence) , specifically a binomial rather than Gaussian GLM distribution was considered because the data was noted to be normally distributed. The decision to develop species distribution models was dictated by the limited spatial data for the key landscape species that are either; endemic, critically endangered, threatened or vulnerable in the study area. On the other hand, the development of presence-only modeling methods (Thuiller, 2003; Thomas et al., 2004; Guisan and Thuiller, 2005) that require a set of known occurrences together with predictor variables such as topography, climatic, soils, hydrologic, and remotely sensed data was the motivating factor. Large quantities of occurrence data, particularly in natural history museum and herbarium that is freely available on the internet made it easy to compile the SDMs for some selected species and used to compile the ecosystem services map for the Congo basin.

High-resolution elevation data obtained during a Space Shuttle flight for NASA's Shuttle Radar Topography Mission (SRTM). The 3-arc seconds (approximately 90 meter at the equator) elevation model was derived from Defense Mapping Agency (DMA) map products and DMA digital terrain elevation data (DTED). This data is distributed free of charge by USGS and the Consultative Group of International Agricultural Researchers (CGIAR) - Consortium for Spatial Information (CSI) (http://srtm.csi.cgiar.org/) at varied resolutions (e.g. 30 m for the USA and 90m for the rest of the world) and geoformats. The 90 meter DEM was used to derive topographic variables such as elevation, slope, aspect, and the implementation of watershed and stream analysis. Annual precipitation, potential evapotranspiration, roads, human population, political and administrative boundaries are also included in the USGS Global GIS dataset and compiled on a CD-ROM can be accessed on request.

2.3.5 Land Use Land Cover Map

Most of the spatial data was acquired from United States Geological Survey (USGS) Global GIS dataset of 2003 to implement watershed analysis and hydrological modeling. U S. Geological Survey's (USGS) Global Land Cover Characteristics (GLCC) dataset is derived from Advanced Very High Resolution Radiometry (AVHRR) satellite imagery produced at 1km resolution (Loveland, Merchant, Brown, & Ohlen, 1991). A detailed description of the methodology used to compile the USGS global land cover map can be found on USGS's official website http://webgis.wr.usgs.gov/globalgis/landscan/landscan.htm. Other sources of LULC maps include Food and Agriculture Organization of the United Nations (FAO) and the United Nations Environment Programme (UNEP)'s AFRICOVER produced and distributed by the Global Land Cover Network (GLCN), and GLOBCOVER produced by European Space Agency (ESA) and Université Catholique de Louvain (ESA and UCL, 2010). ESA's 2009 global land cover map was generated using 12 months worth of data collected from 1st January to 31st December 2009 from Envisat's Medium Resolution Imaging Spectrometer (MERIS) instrument while FAO and UNEP land cover map was essentially derived from visual interpretation of remotely acquired high resolution satellite images and using AFRICOVER Interpretation and Mapping System (AIMS) to develop vegetation class based on FAO Land Cover Classification system (LCCS) (Kalensky, 1998; Latham, He, Alinovi, DiGregorio, & Kalensky, 2002).

USGS Land Use Land Cover (LULC) map was evaluated alongside others data sources such as AFRICOVER and GLOBCOVER for accuracy based on representation of vegetation types, classification level and area coverage, and processing techniques and found to be the best map available for the Congo basin. For example, USGS LULC map was compared with ESA's GLOBCOVER and the classification scales were quite different and vary in vegetation classification accuracy. USGS Global land use land cover map is composed of three different levels of vegetation classification, that is, the primary classification which is relatively finer with a total of 28 global cover classes, secondary classification which is coarse and the third level that is much broader at 11 classes. One obvious error identified in the USGS LULC was a slight mismatch between the polygon boundaries and the extent of the vegetation class. Technical errors might have arisen out of poor digitization of administrative and/or system boundaries and the vegetation cover classification by the remote sensing expert resulting in disaggregated classes. ESA's land cover map has 19 broad vegetation classes, but of particular concern is the misclassification of savanna grasslands and some patches of evergreen broadleaved forests into cropland/mosaic vegetation that were easily detected with the help of Google earth, high resolution satellite images from MODIS sensor and the researchers knowledge of the region.

The USGS land use land cover map was reclassified by condensing the original 28 classes to 12 that occur in the study area (Figure 6.3). Before any analyses were done, the geoprocessing environment and spatial references were set to Datum: GCS_WGS1984 and projection: WGS1984_UTM_ Zone 35S, which is the appropriate zone for the study area.

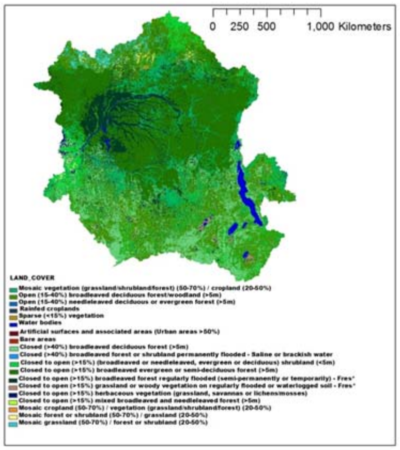

Figure 2.3 European Commission Global land cover produced in 2009

2.3.6 Environmental data

Data on environmental variables was acquired from WorldClim database (Hijmans et al., 2005; http://www.worldclim.org). WorldClim database consists of 19 bioclimatic variables derived from the monthly temperature and rainfall values in order to generate more biologically meaningful variables (Hijmans et al., 2005). These are often used in ecological niche modeling (e.g., BIOCLIM, GARP). The bioclimatic variables represent annual trends (e.g., mean annual temperature, annual precipitation) seasonality (e.g., annual range in temperature and precipitation) and extreme or limiting environmental factors (e.g., temperature of the coldest and warmest month, and precipitation of the wet and dry quarters). A quarter is a period of three months (1/4 of the year). The data layers were generated through interpolation of average monthly climate data from weather stations on a 30 arc-second resolution grid (often referred to as "1 km^2" resolution). Variables included are monthly total precipitation, and monthly mean, minimum and maximum temperature, and 19 derived bioclimatic variables. The WorldClim interpolated climate layers were made using: Major climate databases compiled by the Global Historical Climatology Network (GHCN), FAO, WMO, the International Center for Tropical Agriculture (CIAT). The SRTM elevation database (aggregeated to 30 arc-seconds, "1 km").

The biophysical variables selected were Digital Elevation Model (DEM), slope, Euclidian distances of both rivers, and roads, human population, and aspect. Aspect was computed in form of eastness = sine(aspect in radians)*3.14159/180), and northness = cosine (aspect in radians)*3.14159/180) a formula that was also used by Estrella et al., (2012). Flat terrain was classified as zero. The other variables that were considered but later dropped because they were categorical in nature and would affect the model constant and associated coefficients substantially, let alone the interpretation challenges include lithology, subwatersheds, soils and Compound Topographic Index (CTI). The Compound Topographic Index (CTI), also referred to as Topographic Wetness Index (TWI) is a steady-state wetness index. The CTI was considered for use in the model, however, it was not used in the final analysis because it is already embedded in the FAO soils data compilation. Temperature, elevation, and precipitation are commonly used physical variables that are used to quantify turnover along ecological gradients (Mitasova et al. 1996). These abiotic predictors have been leveraged widely in conservation to address known gaps in existing knowledge on the distribution of species and ecosystems (Hortal and Lobo, 2005). The use of physical variables to measure landscape scale changes relies on the assumption that the physical variables chosen act as a surrogate for biodiversity within the landscape and that differences within the descriptor variables (eg. temperature, elevation) are correlated with differences in species composition.

Table 2.2 Biophysical data used in species distribution modeling

	Environmental variables	Description	Reference/source
1	Bio2	Mean Diurnal temperature range	Worldclim
2	Bio6	Mean temperature of coldest month	
3	Bio10	Mean temperature of warmest month	
4	Bio5	Maximum temp of warmest month	
5	Bio16	Precipitation of wettest quarter	
6	Bio18	Precipitation of warmest quarter	
7	Elevation (m)	Physical	USGS (STRM) 90m DEM
8	Slope (m)	Physical	USGS DEM
9	Population density (persons/km^2)	Biotic	CIESIN
10	EucdistRDS	Euclidean distance to roads	USGS road network & ArcGIS geoprocessing
11	Eucdistrivers	Euclidean distance to rivers	Hydrosheds rivers & ArcGIS processing
12	Eastness	Component of aspect	Computed in ArcGIS
13	Northness	Component of aspect	Computed in ArcGIS
14	Soils	Soil types	FAO HWSD
15	Lithology	Rock structure	USGS Ecosystem datasets

Reliance on physical characteristics of the landscape for conservation classifications is frequently driven by asymmetry in information availability between physical and biological data (Jetz et al., 2012). Abiotic measures of environmental conditions are generally cheaper to acquire, provide more complete coverage, and are more readily available than detailed empirical meteorological and hydrological monitoring data.

The identification of areas with steep gradients (rapid rate of change) in environmental space is designed to minimize the distance that species will need to travel in order to reach different and potentially more favorable conditions, by ensuring that a wide variety of habitats are locally present. The objective is to retain connectivity between areas that contain dissimilar environmental conditions (eg. connecting high, cool and dry places with nearby low hot and wet places). The analysis does this by first quantifying the current condition at a site based on abiotic variables, and then assessing the similarity of each site to neighboring areas. By promoting connectivity between habitat types the analysis seeks to provide options in a time of climate change, by ensuring diversity in local habitat availability.

In addition to be useful for promoting the adaption by providing geographically proximate areas where conditions are different, they have also been suggested as areas of speciation and unique biological diversity (Kark, 2006). Gradient areas have also been suggested as potential areas where monitoring efforts should be focused in times of climate change (Kark, 2006), as the regions may serve as a warning to changes happening elsewhere. The incorporation of environmental gradients within conservation planning has also been recognized as a method for promoting persistence of landscape level processes (Rouget et al., 2006).

2.3.7 Environmental distance

The measurement of environmental distance provides a mechanism to quantify the observed differences in environment conditions across space. Methods for quantifying environmental distance can be broken down into two groups based on the type of input data used to quantify distance; 1) distance measures based on a species composition, and 2) distance measures based on abiotic conditions. Distance measures that use species composition directly rely on survey data that details species abundance or presence/absence and then quantifies the relative similarity of sites based on differences in species assemblages at a site. Measures that use abiotic predictors utilize information on abiotic factors like elevation, rainfall, and temperature to quantify differences between sites. Environmental distance measures have been used to support a wide variety of conservation applications. They have been used to delineate biological domains (Mackey et al., 2008), predict species composition (Ellis et al., 2012; Pitcher et al., 2012), identify regions at risk of climate change (Saxon et al., 2005), inform survey design (Hortal and Lobo, 2005), explain genetic diversity in populations (Mendez et al., 2010), to identify priority areas for the expansion of protected areas (Faith et al., 1987), and to promote connectivity in national scale adaptation planning (Game et al. 2011).

2.3.8 Spatial autocorrelation and multicollinearity

Modeling of species distribution based on range maps or survey data often suffer from spatial autocorrelation whereby locations close to each other exhibit more similar values than those further apart. Spatial autocorrelation occurs when the values of variables sampled at nearby locations are not independent from each other (Legendre & Fortin, 1989; Tobler, 1970; Legendre, 1993). Biological processes such as dispersal, extinction and speciation are distance dependent, and quite often modeled linearly in relation to the environment. More so, the statistical models fail to account for spatial structuring of the environment resulting in spatial autocorrelation (Legendre, 1993) and spatial dependency (Legendre et al., 2002). If undetected during species distribution modeling, spatial autocorrelation can result in biased parameter estimates and an increase in type I error rates due to violation of a key assumption of standard statistical analyses, which states that residuals are independent and identically distributed (Hawkins, Diniz-Filho, Mauricio Bini, De Marco, & Blackburn, 2007; Dormann et al., 2007). Several methods of dealing with spatial autocorrelation for both presence/absence and species abundance data such as autocovariate regression, spatial eigenvector mapping, and generalized least squares have been proposed and their weaknesses and strength discussed (Dormann et al., 2007). Legendre et al. (2002)

proposed that performing a Dutilleul's modified t-test for the correlation coefficient, corrected for spatial autocorrelation, effectively corrects for spatial autocorrelation and presence of deterministic structures in the data.

In order to ascertain whether the environmental variables and species presence/absence data were spatially autocorrelated, a pairwise Pearson correlation analysis was conducted using ENMTOOLs (Warren et al. 2010; a toolbox for comparative studies of environmental niche model; http://purl.oclc.org/enmtools) and only variables with less than (+/-0.75) correlation were retained. This was followed by the determination of the significance of the spatial component using Moran's I for all spatial predictors followed by backward/forward selection of a model for a set of spatial predictors in a generalized binomial linear model (GLM) that exhibited a Pearson's $R <$ 0.6 (Peres-Neto & Legendre, 2010). Following this procedure, non-correlated environmental variables were selected for each species distribution model.

A binomial logit model was considered appropriate for binary data (Nelder and Wedderburn, 1972). The Logistic multiple linear regression analysis is a statistical method that can be used to explore relations between species and environment basing on collected observations of a species and environmental variables at each particular location (Narumalani et al., 1997).

$$log\left[\frac{\pi(x)}{1-\pi(x)}\right] = logit(\pi) = \alpha + \beta_1 x_1 + \beta_2 x_2 + \cdots + \beta_k x_k$$

The generalized linear model (GLM) identifies variables important in predicting the probability of occurrence, by defining the presence or absence of such an occurrence as a dichotomous, dependent variable (Harvey et al. 1987). The GLM yields coefficients for each variable based on data derived from samples taken across a study site. The coefficients then served as weights in an algorithm in the GIS to produce a map depicting the probability of occurrence of modeled species. In order to generate the GLM probabilities, the logit equation is written exponentially.

$$P = p(d = {}^{1}/_{x}) = {}^{1}/_{(1 + (\exp -(b_0 + b_1 x_1 + b_x x_2 + \cdots b_k x_k)}$$

Where d is presence/absence (e.g. gorilla), which is the dependent variable, x1.............xk, are a set of predictor variables such as biophysical (i.e. slope, DEM) and environmental variables (i.e. annual precipitation, mean temperature of warmest quarter), and b1……….bk are coefficients derived from logit regression model output. Spatial biodiversity data used in species modeling was acquired from various sources. The birds, and large mammals distribution data was acquired from IUCN/Birdlife International (BirdLife International and NatureServe 2011), Global Biodiversity Information Facility (GBIF), and European Commission/The Institute of Applied Ecology (IEA)'s African Mammals data bank, Apes, Populations, and Environments (APES) database compiled by IUCN/Species Survival Commission (SSC), The Dian Fossey Gorilla Fund, Wildlife Conservation Society, Jane Goodall Institute, World Wide Fund for Nature (WWF), and African Wildlife Foundation among others. Spatial biological data for presence/absence of wildlife species recorded inside protected areas of the Congo basin and parts of the Albertine Rift provided by Wildlife Conservation Society (WCS) was used to develop species distribution models.

In order to constraint the selection process for the random points, the presence points were displayed in ArcGIS and created sampling polygons by digitizing around the location of individual and/or clusters of points (Figure 2.6 to 2.10). The polygons created where then used in the HawthsTools (Bayer 2004) to generate the random points. A data matrix of presence and absence points was then created and used to extract the associated environmental and biophysical variables corresponding to those points. The output of this analysis was the used to perform a logistic regression analysis by implementing a back/forward stepwise selection and used the Akaike Information Criterion (AIC) for model evaluation as well as the chi square values to test for model significance in predicting the species distribution. The best model selected using the AIC selection criterion was also compared with the results generated when a Bayesian Information Criterion (BIC) was used. In both cases, the model with the lowest AIC or BIC was considered to be the best model. Importantly, both AIC and BIC selection criteria generated similar results.

2.3.9 Model evaluation

Different measures of evaluating the quality of prediction of species distribution model results have been proposed (Fielding & Bell, 1997) depending on the goal of the study. Majority of the evaluation techniques (e.g. Cohen's kappa) are threshold dependent where predicted values above the specified threshold indicate species absence. The other methods such as the Area under Receiver Operating Curve (AUROC or AUC), a measure of rank-correlation are threshold independent. A high AUC value (>0.5) means that high predicted scores tend to be areas of known species presence and locations with lower model prediction scores tend to be areas of known presence. After fitting the model, the overall model fit and hypothesis regarding a subset of regression parameters was tested using a likelihood ratio test (LRT) statistic. Likelihood ratio tests compare the full model with a restricted model where the explanatory variables of interest are omitted. The p-values of the tests were calculated using the chi square statistic. The logistic regression constant and the coefficients were then used to compile the species probability distribution maps in ArcGIS. In the case of birds and large mammals, the species range maps provided by IUCN/Birdlife International (BirdLife International and NatureServe (2011), and the African Mammals Distribution Maps produced by the Institute for

Applied Ecology and the European Commission were used to examine the accuracy of the SDMs produced. The African Mammals Distribution maps and IUCN species range maps are considered to be authoritative and SDMs were compared with these maps by visual inspection. The maps were further presented to the experts working in the region for further evaluation (expert-based validation technique).

A typical challenge with species distribution modeling is the lack of adequate spatial field survey points per species to achieve reliable results in terms of probability of occurrence. The other challenge concerns the choice of species to model that could adequately represent the biodiversity values. For example, how would the results change, if common species are included in the analysis? Is IUCN species categorization the best selection criteria for species selection? How effectively do the selected species, regardless of the selection criteria, represent the ecosystem types within the entire basin? These questions could not be answered in the scope of this study, but are still important to think about.

2.4 Results

In species distribution modeling, multicorrinearity is a very import issue that needs to be considered seriously. As explained earlier on the first sections of this chapter, all variables selected for use in the compilation of the distribution models, were investigated for autocorrelation and later multicorrineality. The first results of this chapter focused on reducing noise and data redundancy by selecting the right suite of variables to use for modeling. Results of the predictor variables are shown in Figure 2. 4.

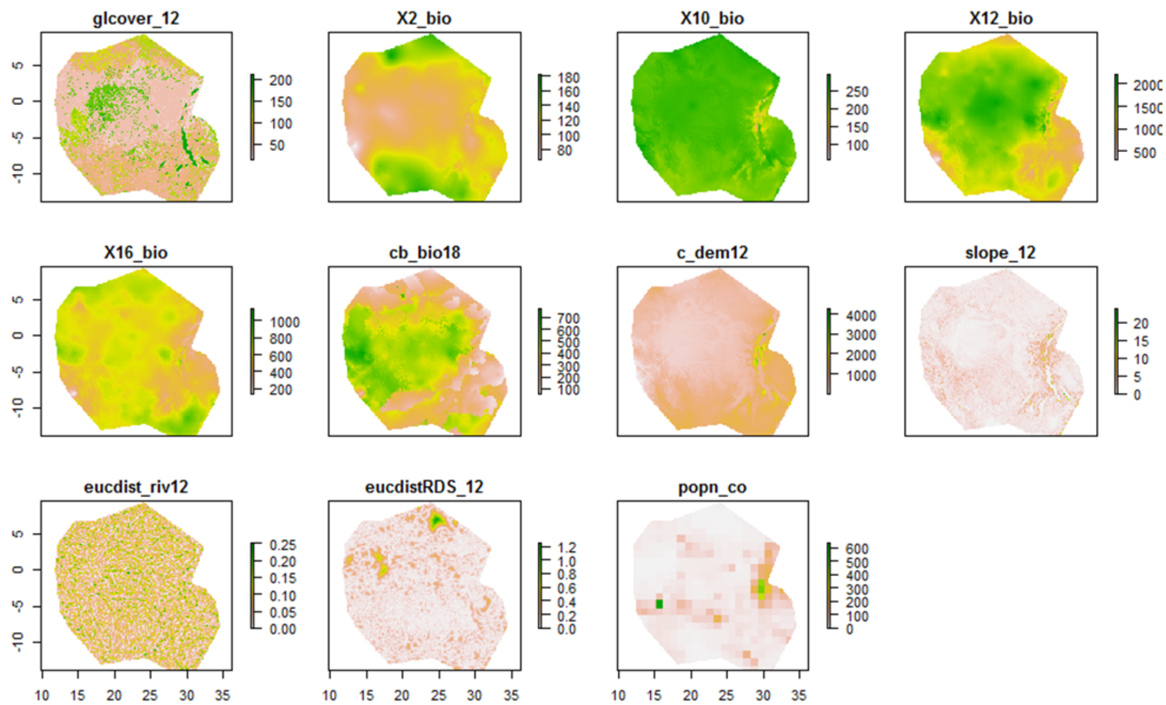

Figure 2.4 Pairplots of predictor variables selected for species distribution modeling

The pairplots were performed to identify variables that share similar information so they can be eliminated from the explanatory variables (Figure 2.5). In this case, annual precipitation (BIO12) was highly correlated (0.97) with precipitation of the wettest month (BIO16), while precipitation of the warmest month (BIO18) was highly correlated (0.9) with annual precipitation. Equally, the mean temperature of the warmest quarter (BIO10) was highly correlated (0.991) with elevation.

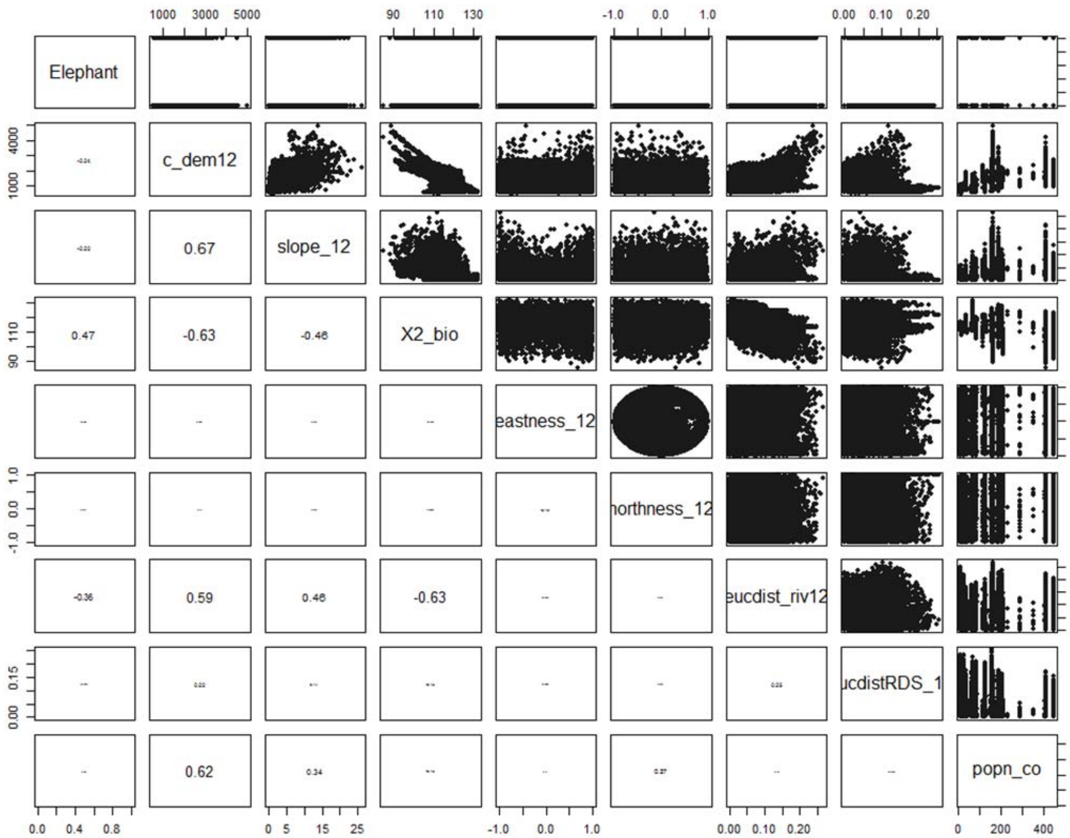

Figure 2.5 Pairplots of environmental variables used in species distribution modeling

In figure 2.6, Euclidean distance to rivers and mean diurnal temperature range (BIO2) exhibited a very noisy pattern and yet in the model, they demonstrated to be explaining the distribution of elephants very well.

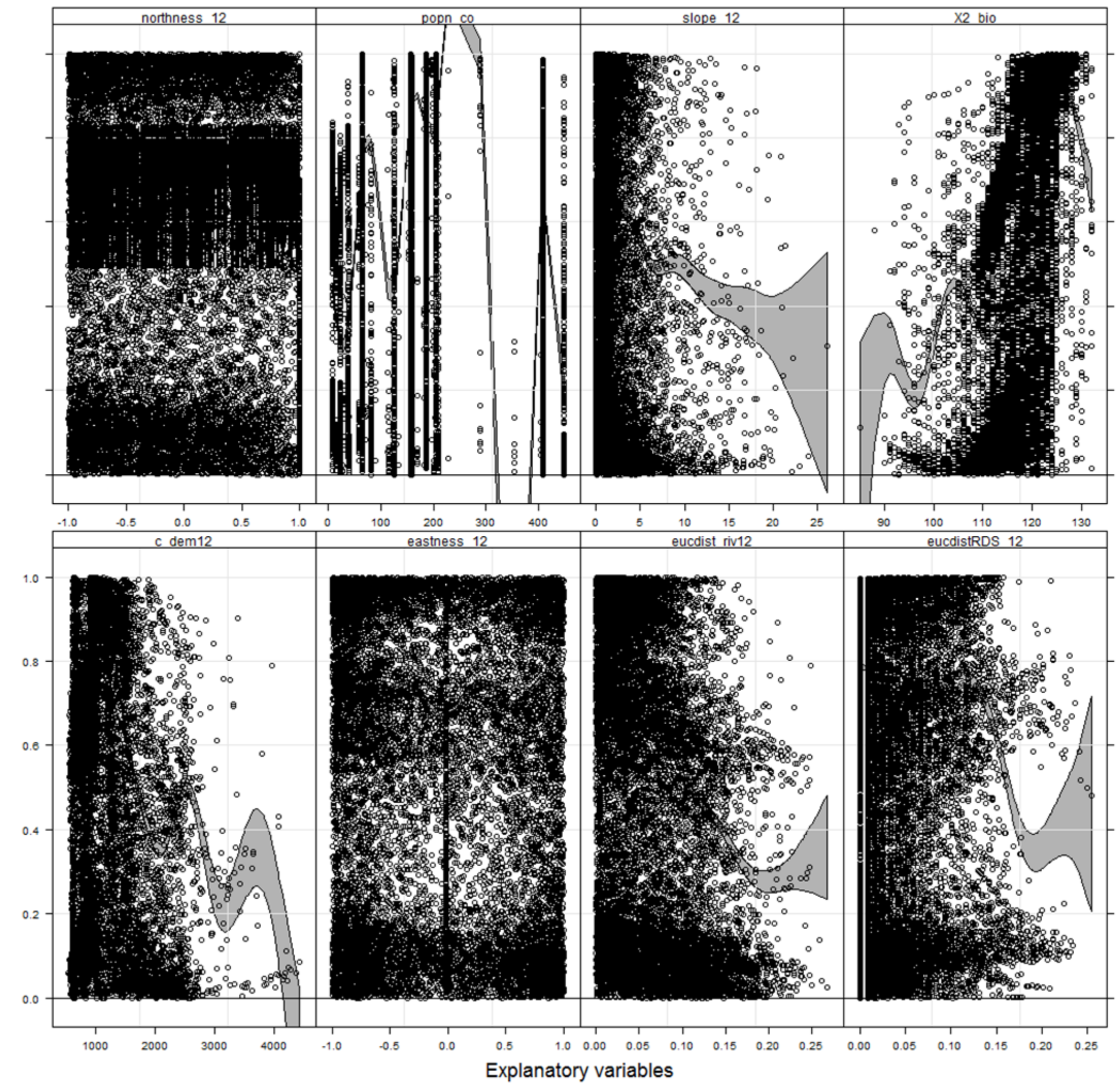

Figure 2.6 Residual plots of explanatory variables

Even after removing those explanatory variables that were redundant, some of the variables maintained in the model were overfitting the data. In order to eliminate noise in the data, residual plots were performed against the fitted variables (Figure 2.7). For example, slope was over fitted in one of the generalized linear regression model. This had to be investigated and eliminate the noise before use in the model. Different soil types contributed quite different information to the model. Soils being a factor variable with 15 different soil types, it was difficult to include this variable in the species models without knowing which particular soil group was making a meaningful contribution to the model. The deviations of individual soil types were quite varied and more concentrated in the lower regions of the fitted model.

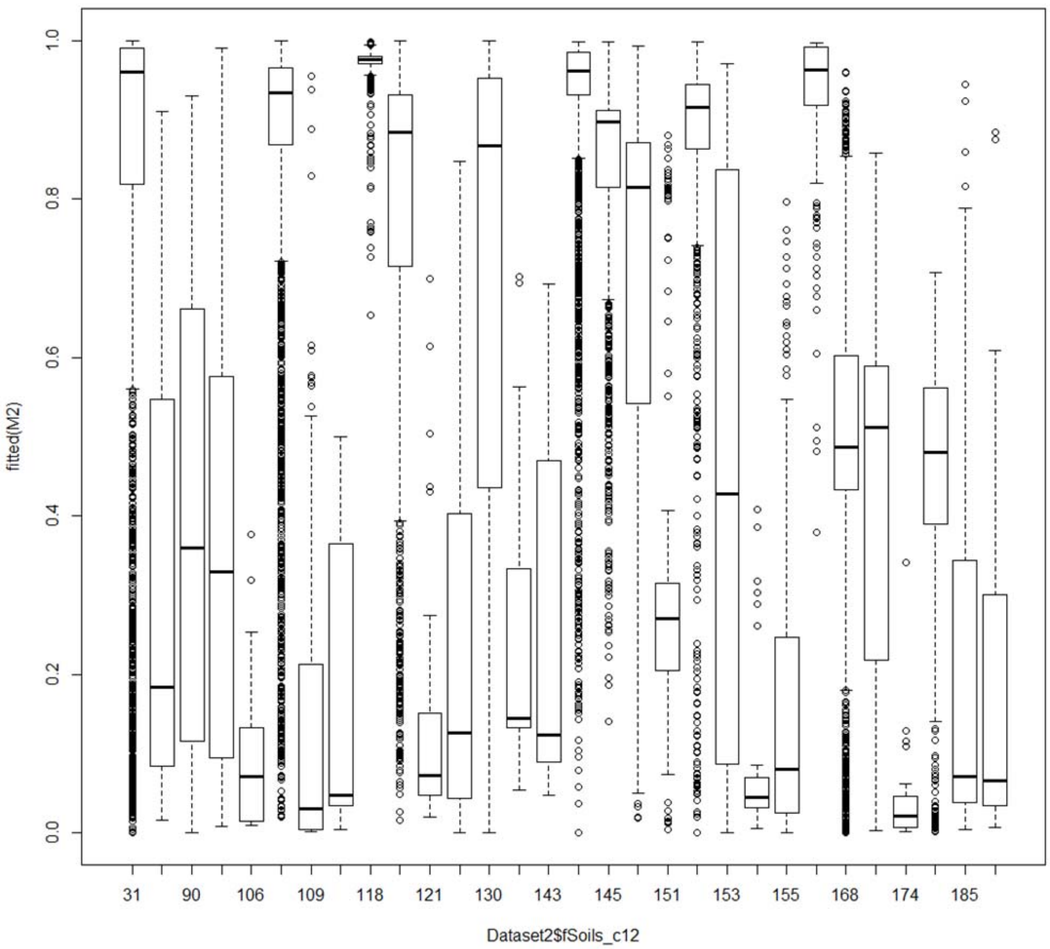

Figure 2.7 Fitted regression model plotted against soils

By repeating these processes for each individual species, the best variables for individual species model were selected. For example, the best variable that predicted well it distribution are presented in Table 2.3. All the variables selected in the model were highly significant.

Table 2.3 Regression model for the elephant distribution model.

	Df	Deviance	AIC	LRT	Pr(>Chi)	
<none>		25342	25528			
popn_co	1	25471	25655	128.6	< 2.2e-16	***
poly(X2_bio 2)	2	26330	26512	988.1	< 2.2e-16	***
slope_12	1	25356	25540	13.6	0.000229	***
eastness_12	1	25368	25552	25.8	3.84E-07	***
fSoils_c12	29	28119	28247	2777.2	< 2.2e-16	***
fLithology_12	10	25826	25992	483.8	< 2.2e-16	***
PA	1	28886	29070	3544.2	< 2.2e-16	***
eucdist_riv12:fVegcode	11	25490	25654	148.4	< 2.2e-16	***
fVegcode:eucdistRDS_12	11	25878	26042	535.9	< 2.2e-16	***
fVegcode:c_dem12	11	25455	25619	112.7	< 2.2e-16	***

Signifance codes: 0.000 ‘***’ 0.01 ‘**’ *’ 0.05, ‘.’ 0.1 ‘ ’ 1

In this model, the interaction of the elephants with habitats patches outside the protected area boundary were quite important in determining which areas are likely to be suitable for the elephant to move to. Elephants in the study site are very mobile and their mobility is triggered by the availability of resources mainly food and water. However, their movement is sometimes triggered by the conflicts in the region. Figure 2.8 shows the potential distribution of elephants generated by the model. The polygons superimposed on the map actually represent the species range as indicated by IUCN species data list. The model was doing a very good job of identifying those areas where the elephants have been cited or known to occur.

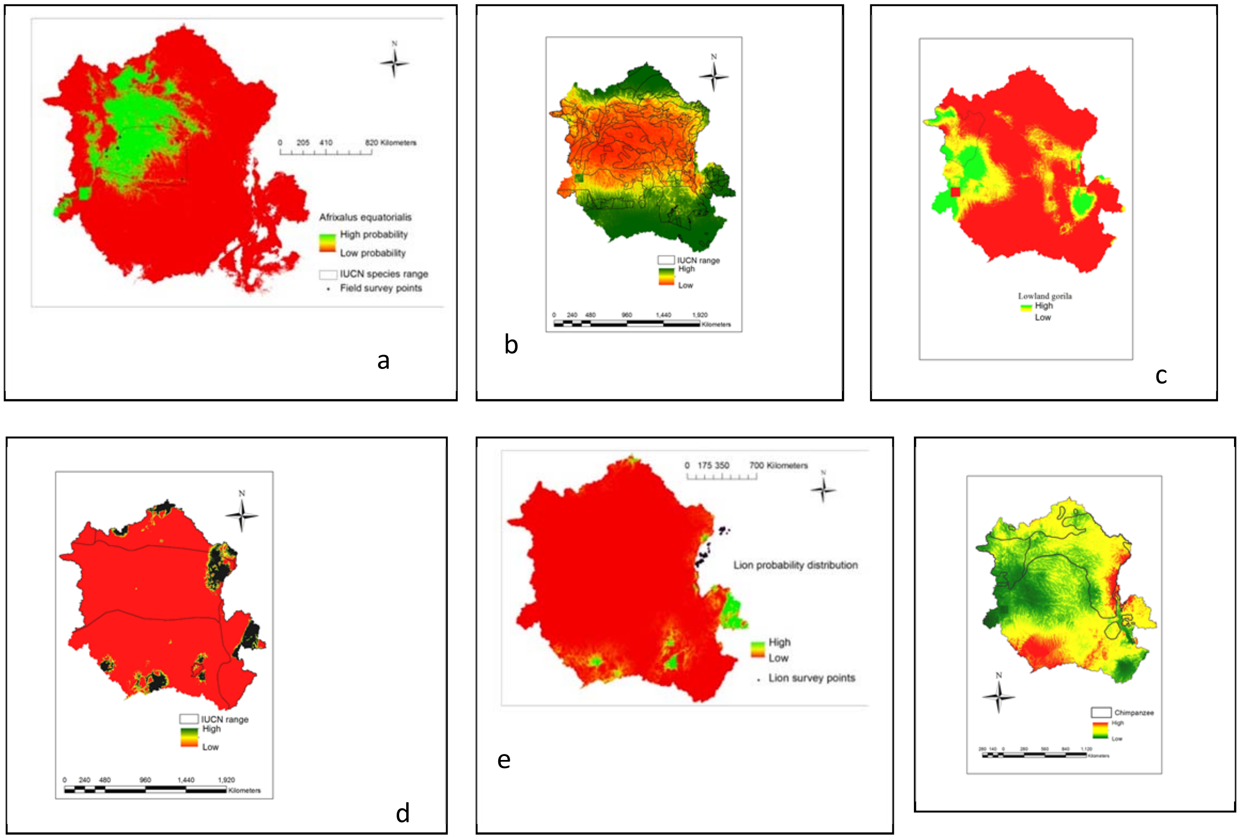

Figure 2.8 Species distribution models of a) *Afrixalus equitorias* (Congo banana frog), b) elephant, c) lowland gorilla, d) *Syncerus caffer*, e) *lion, f)* Chimpanzee

Figure 2.9 shows species distribution of selected fish species and one bird species, a Grauer's warbler. Fish and birds are important taxa in the Congo basin. Change is the ecological health of the basin is likely to dramatically affect the forest specialist and wetland birds, and the fish ecology. Congo River is connected to very many important rivers and stream, and provides a direct link to the ocean. The threats to the basin already discussed present a challenge to the management of fisheries, large mammals, birds, and the herptofauna. The streams are very critical to the movement of fish, and herptofauna, unfortunately, there is not much monitoring and research going with regard to these taxa.

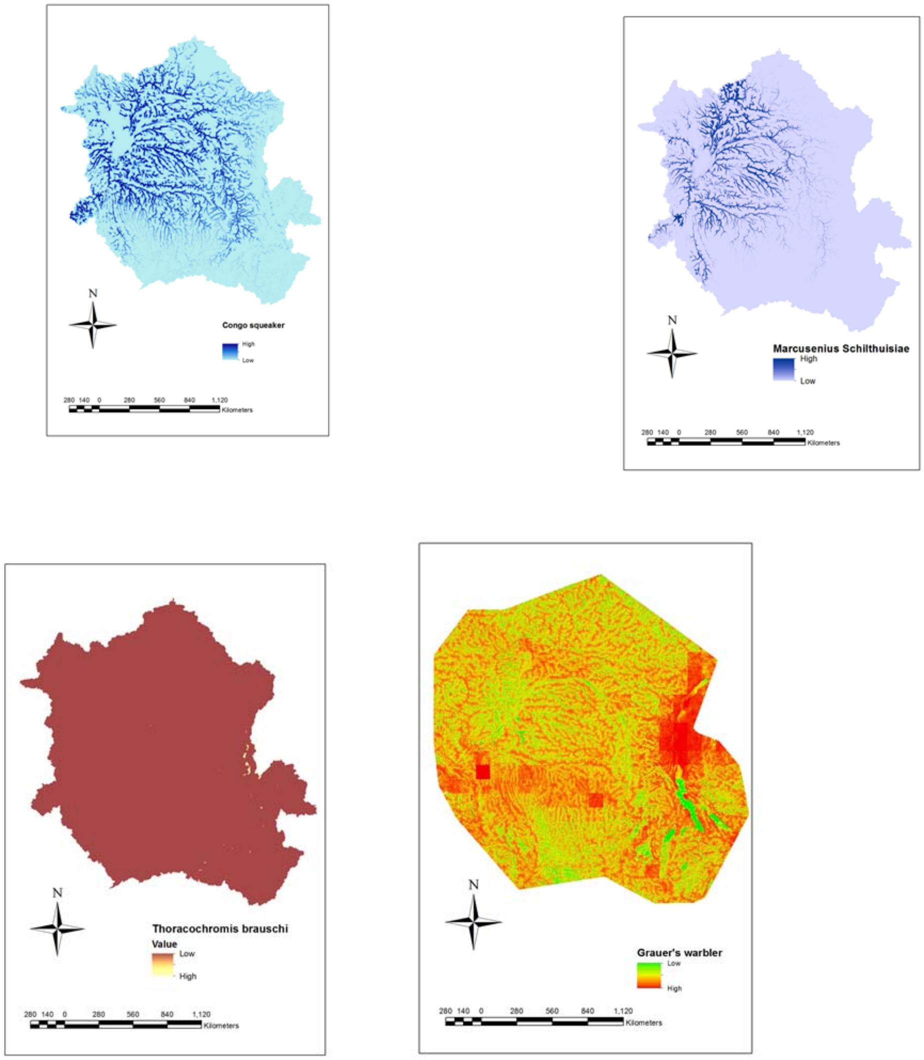

Figure 2.9 Species distribution models for Congo squeaker (top left), *Marchsenius schilthisiae* (top right), *Thorachromis brauschi* (bottom left), and *Grauer's warbler* (bottom right).

The distribution of amphibians and fish is more concentrated in riparian areas while that of large mammals and reptiles is more spread out all over the entire basin (Figure 2.10). The distribution of bird species was more predominant in the Albertine rift to the east of the basin and this is not to imply that the other parts are not suitable for birds, rather this is the most surveyed area and less field information is available in the other parts of the Congo basin.

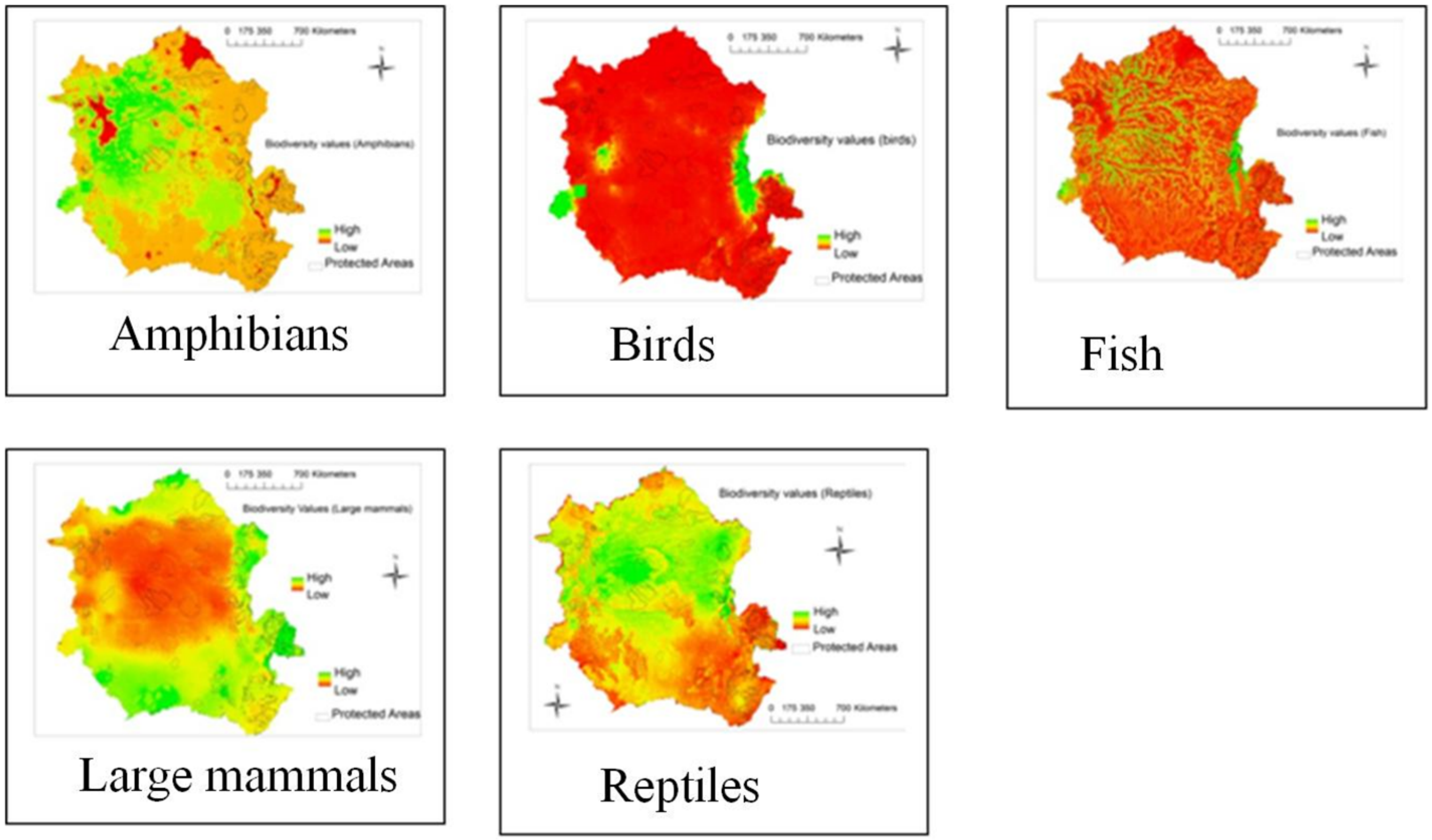

Figure 2.10 Biodiversity distribution maps for the five different taxa

2.5 Policy implications

Species distribution modeling is one of those new techniques of quickly producing information about a species by combining observations of species occurrence with environmental estimates to gain hind sight about their habitat needs and threats. Species distribution modeling, however, should not be considered a perfect surrogate for monitoring and research. It still relies heavily on species inventory data. While it is necessary to combine the knowledge of remote sensing and species modeling techniques to approximate habitat complexity of birds, large mammals and other taxa, it is logical that investment in conducting field studies continues. Results from this study demonstrated that there are still challenges in getting the species modeling techniques to adequately represent what is on the ground and these concerns have been well articulated by many scholars familiar with modeling (Austin 2002; Soberon 2007). Methods are still needed to address the challenges of uncertainty in species distribution modeling (Magder & Hughes, 1997).

Differences in resource needs, including habitat and water resources by different species in selected taxa present a challenge to conservationists and wildlife managers alike. The fundamental question is what species should be considered to be representative of the other species's needs when it comes to conducting SDMs. Endangered and threatened species were selected for modeling because they face eminent danger of being driven to extinction. This selection criterion, however, underscores the role of other species toward building adaptive ecological networks. Selection of landscape species and emphasis on flag or charismatic species also presents a risk of underestimating the importance of lower species toward detection of a system collapse. The other challenge is that donors are overwhelmed by human welfare demands to the effect that funding that used to be allocated to conservation is now being channeled to meet human basic needs, particularly in developing countries. In this study, it has been demonstrated that you can adequately represent ecological process using mechanistic models to learn more about the system.

2.6 Conclusions

The most important outcome of this study is that endangered and threatened species in the Congo basin are not uniformly distributed and that their ecological needs vary with respect to environmental, biophysical and ecological factors. In representing the distribution of nearly 50 species, only topographic factors such as elevation and slope, forest vegetation type, and precipitation accounted for the distribution and pattern of species in the Congo basin. Areas in the basin that are considered marginal at present seem to be the next available suitable habitat of quite a large number of species. Unfortunately, most of those areas are equally habited by humans and continue to be the next available land for agricultural expansion. The distribution of large mammals and reptiles was noted to be more spread out in the entire basin than the birds, amphibians and fish. The latter taxa had high potential distribution in riparian and waterway areas for obvious reasons that they highly depend on water. In the areas where the model demonstrated less suitability potential of species occurrence could be attributed to high extraction of resources taking place such as timber and mineral mining, but this is also attributed to the lack of data in those areas.

2.7 References

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CHAPTER 3

CONGO WATERSHED ANALYSIS AND HYDROLOGICAL MODELING

3.1 Introduction

Globally, fresh water resources are recognized as an important resource to sustaining both human lives and nature. Large river basins in Africa such as Congo, Nile, and lakes; Tanganyika, and Victoria are the major sources of freshwater. Unfortunately, these basins face immense pressure from national developments, which have resulted in severe changes in hydrological cycle and livelihood disruptions for the dependent communities (Bates et al., 2008). The situation is likely to become complex with the advent of climate variability and change (Tshimanga and Hughes 2013). Given its wide-scale impacts, climate change is likely to provide an opportunity for governments that share water resources to cooperate and design appropriate strategies to secure these watershed resources. Water resources are transboundary in nature and changes in one spatial location affects the entire system. As such, hydrologists, water engineers, urban planners, agriculturalist, social scientist among other professional scientists are beginning to appreciate the need for a multidisciplinary approach to water resources management. More importantly, it is no longer economically feasible for water management authorities to collect the information needed to plan for effective water resources management. Cognizant of this, new innovative ways of generating this information have gained a lot of popularity. Watershed analysis and hydrological modeling are some of the widely used techniques to try and understand the behavior of watersheds (Döll et al., 2008) through an integrated water resources management approach.

In this study, an attempt was made to analyze the Congo River watershed and identify unique hydrological characteristics that are important to the resource managers for water resources management planning. The Congo River Basin is located in central Africa covers nearly 3.8 million km^2 and lies between 9^oN, 12^oE to 13.30^oS, 34^oE, and overlaps nine countries. Congo is the second largest basin in the world after the Amazon and generates a monthly flow volume of 108 147.5 Mm^3 at the outlet (Reference is made to the Kinshasa-Brazzaville gauging site, Feteke et al., 1999; Tchimanga and Hughes, 2013) representing forty percent of the African continent's discharge (Crowley et al., 2006). The Congo Basin is historically known for its river navigation services dating back to the pre-colonial period. In addition, the Congo Basin is recognized for accumulating the largest quantities of carbon in Africa (Mitchard, et al. 2008; Baccini e tal. 2008) due to its high (44%) composition of forest cover. It is estimated that over 75% of the rainfall received in the region is due to the Congo Basin, which is recycled through evapotranspiration (Eltahir et al., 2004).

3.1.1 Problem

While land use change effected at a modest scale may enhance the provision of essential ecosystem goods, if poorly managed, it alters a range of other ecosystem functions and continued supply of those goods (DeFries et al., 2004). In the humid tropics, deforestation is the number one threat to biodiversity hotspots that are also known to exhibit the highest potential to supply ecosystem goods and services. Proximate causes of deforestation include timber extraction, agricultural and pasture expansion, colonization of formerly inaccessible frontiers, infrastructure especially roads and urban development (Lambin et al., 2005; Chowdhury, 2006). More recent estimates of Africa's tropical forest change from the latest UNFAO Forest Resource Assessment (FAO, 2005) indicate that annual deforestation rates are nearly five million hectares (DeFries et al., 2005) higher than what was reported by Achard et al. (2002). The annual deforestation rate in the portion of the Congo basin located in Democratic Republic of Congo alone was reported to be 0.2% equivalent to 311,000 ha (UNEP, 2011). Deforestation in the Congo Basin occurs at fine scales and is caused largely by shifting agricultural activities (CBFP, 2005; Makana and Thomas, 2006) that are correlated with local populations (Zhang et al., 2005). Commercial logging is also present but is highly selective and typically only detectable via the extension of new logging road networks into the forest domain (Laporte et al., 2007). Timber harvest has increased from 3.05 million cubic meters reported in 1990 to 4.45 million cubic meters in 2010 (UNEP 2011). During the same period, fuelwood extraction increased from 44.2 million cubic meters to 75.44 million cubic meters (UNEP, 2011). Substantial parts of the Congo basin landscapes are heterogeneous agricultural areas, in which crop planting and land management decisions are based mainly on interactions among soil characteristics, microclimate, and economic convenience resulting in natural, cultural, and economic capital complexities for resource managers and conservationists.

All these land use changes have had far-reaching consequences in terms of spatial extent, ecosystem impacts, food production and local livelihoods in tropical forests (Foley et al., 2005). In spite of the logging activities in the Congo basin, the regeneration potential is very high, and if properly managed and protected, secondary forests, including those impacted by small-scale shifting agriculture have high potential conservation and economic value to the region (Makana and Thomas, 2006). Furthermore, limited information on the basin-wide hydrological process due to lack of technical capacity and finances to conduct research and monitoring are major constraints to effective planning of water resources in the region (Hughes, 2007). The availability of remotely sensed data, however, became the strongest motivation for conducting this study. Thus, the purpose of this study was to quantify the ecosystem services, develop a framework for conservation prioritization by linking the ecological processes and anthropogenic activities to ecosystem productivity.

Typically, a comprehensive assessment of ecosystem services at a watershed scale is necessary to develop appropriate conservation policies and resource management decisions (Nelson et al. 2009). This research was needed to combine the rigor of small-scale studies with the breadth of broad-scale assessments (Hill et al. 2007; Nelson et al, 2008, Petter et al. 2012). Since the Congo river basin spans such a large area, overlapping heterogeneous ownership and national boundaries, a systems' approach is required that assesses multiple of attributes of ecosystem services (Shriver and Randhir, 2006), representing watershed patterns and processes, and their relative state at spatial scale. Assessment of the nature and degree of ecosystem services at a spatially explicit scale (Millennium Ecosystem Assessment, 2005) can be used to determine conservation needs to develop appropriate multiattribute policy (Balmford et al. 2002, Randhir and Shriver, 2009). Policies and best management practices (BMPs) should protect and enhance structure and functions (National Research Council (NRC), 2005) associated with ecosystem services based on spatial prioritization of needs (Clements et al., 1996; Lamy et al., 2002).

3.1.2 Current approaches

Planning and management of water resources at basin-wide scale require accurate hydrological information that is spatially and temporally explicit. In the past, water resource managers focused mainly on streamflow measurements using metering boards and occasionally collecting water samples for water quality laboratory analysis. These techniques, however, are time consuming and only cover a small portion of the basin, leaving the water managers with no choice, but to assume similar patterns of water quality elsewhere. More importantly, the resource managers would apply guess work concerning the changes in water yield. Today, knowledge of the important variables such as soil moisture and drainage capabilities, surface and subsurface runoff, and evapotranspiration, and how these parameters are impacted by land use land cover change is very fundamental. In order to address these challenges, a number of studies have been conducted in the Congo basin over the last three decades (Anthony et al., 1983; Matsuyama et al., 1994; Kazadi and Kaoru, 1996; Bricquet et al., 1997; Callede et al., 2001; Laraque et al., 2001) dealing with varied aspects of the watershed.

An integrated, wide scale, process-based hydrological models aimed at simulating the hydrology of the Congo Basin is the most appropriate strategy that needs to be pursued. This, however, requires accurate data measurements, technical training, and appropriate computing software. None the less, scientific studies have done using innovative techniques for modeling hydrological process in the Congo basin such as Hydrological Modeling System (HMS) in the Congo and Nile Basins (Asante, 2000), River-Transfer Hydrological Model (RiTHM) (Ducharne et al. 2003) and ArcGIS coupled with remote sensing (Chishugi and Alemaw, 2009; Tshimanga and Hughes, 2013). Other studies focused on reducing hydrological modeling uncertainties. For example, Werth et al. (2009) undertook a multi-objective calibration of the WaterGap Global Hydrology Model (WGHM) for the Amazon, Congo and Mississipi Basins where model calibration was done using both river discharge data and the Total Water Storage Change (TWSC) data from the Gravity Recovery and Climate Experiment (GRACE).

3.1.3 Justification of the study

Existing evidence suggests that land use land cover and climate change (Hoare, 2007; IPCC, 2013) are a major threat to water resources availability in Africa. The increasing rate of deforest on the continent and shifts in climatic seasons are expected to result in water scarcity and food insecurity on the continent. In the Congo basin, the Oubangui River flow, a major tributary of the Congo Basin was noted to be decreasing (Ladel et al. 2008) mainly due to siltation, affecting navigation resulting in slowed economic activities in the region. The major water resources management challenges are caused by the desire for governments, individuals and businesses to improve human welfare through economic development. Unfortunately, how decisions are made toward the allocation and utilization of land resources results in unforeseen environmental damages. In order to avoid this pitfall, resource managers, land use planners, and policy makers need reliable and timely information about the processes and staus of the ecosystem services. At the moment, the Congo basin is experiencing a number of development challenges ranging from timber extraction, overexploitation of resources by internally displaced people or refugees running away from civil conflicts. Furthermore, the government of DRC with support from World Bank received funding to develop and implement a REDD+ strategy. The Congo basin is highly biodiverse, however, the key concern is that if the government purely pursues a carbon sequestration policy for the basin, management may direct its activities toward enhancing carbon stocks at the expense of biodiversity conservation and water resources. Under these circumstances, some areas of the watershed are likely to suffer heavy disturbances than others. These differences in subbasin quality cannot be deduced without conducting a scientific research. The major issue of concern is the lack of a well formulated decision support system for water resources management within the Congo Basin and the limited capacity for watershed monitoring and hydrological research. Luckily, under the UN REDD+ policy, governments that enroll under this policy are required to develop Reporting Verification and Monitoring (RVM) system and a set of indicators. As such, an improved understanding of the hydrological process for the basin is a valuable contribution toward the development of baseline data and setting up of a robust RVM system. Such a system should be cognizant of the complexity of transboundary watershed systems and the need to consider the economic, social and environmental dimensions of water resources if the watershed ecosystem services are to be sustainably managed.

3.1.4 Uniqueness of study

In this study, a GIS-based SCS Curve Number method, and the Revised Soil Loss Equation (RUSLE) were used to compute surface runoff and soil erosion for the Congo basin at a spatially explicit scale. A unique aspect of this analysis was introduced by combining watershed and hydrological analysis performed at varied scales as opposed to studies that have either analyzed them separately or combined them afterwards. For this process runoff and soil loss were maped on the entire basin and subbasin level. Later, relationship between biophysical attributes of the subbasins such as stream order, and density, slope, elevation, road density, runoff, and soil loss and biotic factors such as human population density, and biodiversity values were computed to try an explain the observed spatial differences in hydrological values.

3.1.5 Objectives

The study aim was to identify area with the basin where soil loss and runoff are occurring at very high levels. Specific objectives of the study are to: i) identify and delineate priority areas with high water supply and quality in the Congo basin; ii) assess the differences in the level of water degradation within subwatersheds; iii) recommend strategies for improvement of degraded areas as well as maintain priority areas for water supply

3.1.6 Hypothesis

The hypotheses are: i) soil loss and runoff are not varied across the entire watershed, and subwatersheds; ii) different management strategies are needed in order to improve water supply and quality in the Congo basin.

3.1.7 Conceptual Model

The differences in subwatershed characteristics account for the differences in soil loss and runoff. Land use land cover, togography, soil types and environmental factors directly impact on water supply and quality both at a basin to subbasin level. A reduction in land cover results in increased imperviousness whereby infiltration is reduced resulting in high surface runoff. However, in the area where natural vegetation cover is replaced by crops, it is also possible to experience reasonable runoff simply because the crop doesnot provide adequate resistance to the flow of water. Depending on the type of soils and slope, runoff is expected to be high in subwatersheds with high slopes and reasonable amount of precipitation. In implementing this study, spatial data layers for the watershed (Figure 3.1) were used to compile surface runoff and soil loss.

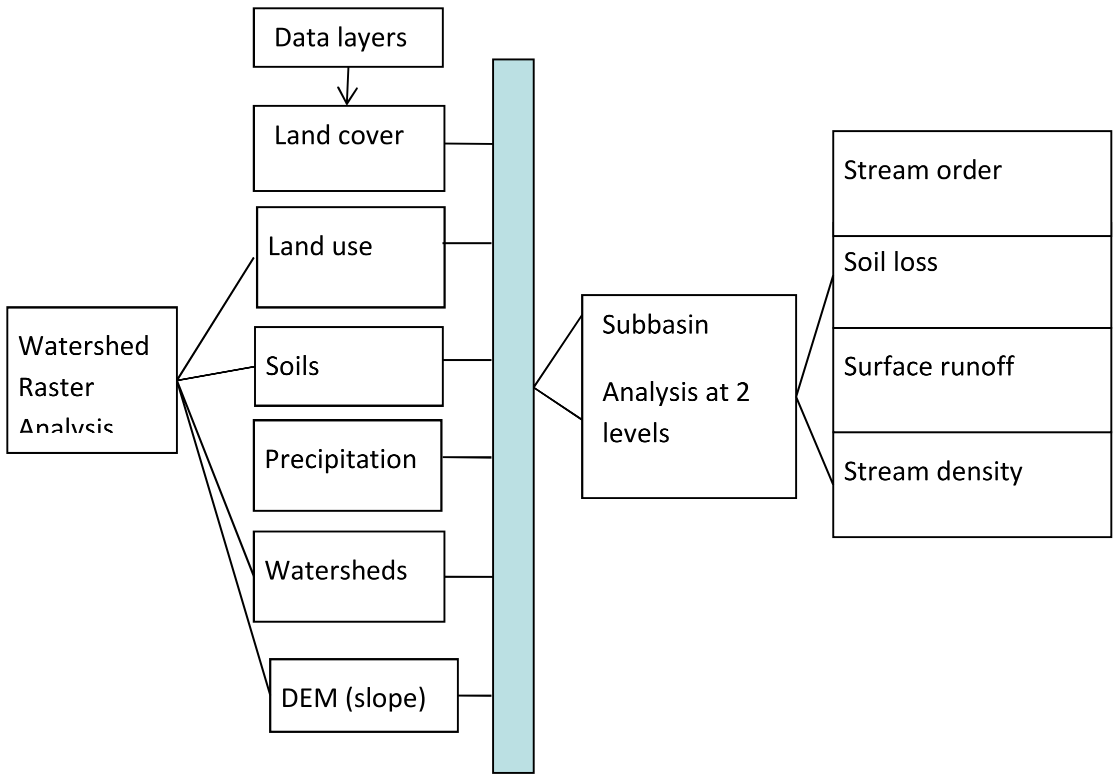


Figure 3.1 Watershed and hydrology modeling framwework for the Congo basin

3.2 Literature review

Africa is the world's hottest continent with deserts and drylands covering some 60 per cent of the entire land surface (UNEPA, 2008). According to the UNEPA (2008) report, only ten per cent of farm soils are prime agricultural land, and more than one-quarter per cent of the land has moderate to low potential for sustainable agriculture. Rainfall variability is high, ranging from near 0 mm/year in parts of the Sahara to 9500 mm/year near Mount Cameroon (Walling 1996). As a result, Africa's water resources are continuously affected by persistent droughts and changes in land use. At the same time, a growing population is increasing the demand on already limited water supplies, particularly in areas which suffer from water shortages. Although, rich in rivers and lakes systems, however, more than 1270 large dams have been built along the continent's many rivers (World Commission on Dams 2001), altering the sedimentation and flooding patterns (Walling 1996). In Africa, East Africa boosts of numerous lakes that support important fisheries which provide livelihoods for millions of people and contribute significantly to the food supply. Among these lakes include Victoria, Tanganyika, Albert, Turkana, and Kyoga. Currently, it is estimated that over 300 million people in Africa face water scarcity conditions (UNEPA, 2008) and about 75 per cent of its population relies on groundwater as the major source of drinking water, particularly in northern and southern Africa. However, groundwater represents only about 15 per cent of the continent's total renewable water resources. Droughts and famine are ever present, and tens of millions of Africans have suffered the consequences every season.

Generally, the hydrological processes in the Congo Basin are complicated due to the varied abiotic and biotic composition of the sub-basins that give rise to the Congo River system. The basin stretches over a large geographic area consisting of different combinations of physiographic and phonological characteristics. Seasonally, the basin has a bimodal pattern of rainfall distribution (Farnsworth et al. 2011) mainly attributed to various external and regional factors acting at a continental scale reinforced by the interactions with the ocean currencies and the monsoonal wind movement shifts (Balas et al., 2007; Farnsworth et al., 2011). The vegetation cover of the Congo Basin varies from dense deciduous forest occupying mainly the central part of the basin to mosaic vegetation types spread over the entire catchment area. The presence of dense canopy cover, accumulated litter, high rooting depth and densities greatly impact the water balance and runoff generated (Bonell, 2004; Roberts et al., 2004; Chappell et al., 2008). Hughes and Hughes (1987) noted that streamflow volumes in the channels of the flat central basin normally undergo high and low cycles in a year resulting in flooding of areas adjacent to the main rivers of the basin.

Land use, land cover change has been identified as one of the major determinants of global change with severe impacts on ecosystems and human vulnerability (Olson et al. 2008; Verburg et al. 2007). Nature conservation programs that are based on payment for ecosystems services and implemented at the international level such as REDD+, a demand-driven intervention on land use linked to global actors, public or private with local land use have become popular as a potential remedy to ecosystem services loss. Land use dynamics is a major determinant of land cover changes. Land cover has been defined as the layer of soils and biomass that cover the land surface (Verburg et al. 2009) while land use is the purpose for which humans exploit the land cover, including land management practices (Lambin et al. 2006; Verburg et al. 2009). Land use involves considerations of human behavior with crucial roles played by decision makers, institutions coupled by the initial conditions of land cover and the inter-level integration of processes operating at varied scales.

3.3 Methodology

3.3.1 Study area

The Congo basin covers over 3.7 million square kilometers and includes the world's second largest tropical forest of approximately two million square kilometers (Hansen et al., 2009; Bwangoy et al., 2010). The Basin's forest zone includes parts of six central African countries: Cameroon, Central African Republic, Congo, Equatorial Guinea, Gabon, and the Democratic Republic of the Congo. The study area includes the core Congo River basin, located within the forest zone from 5° north to 6° south and from 13° to 26° east (Figure 3.2). The climate is warm and humid with two wet and two dry seasons with a mean temperature of nearly 25 °C and average rainfall if 1800 mm per year (Bwangoy et al., 2010).

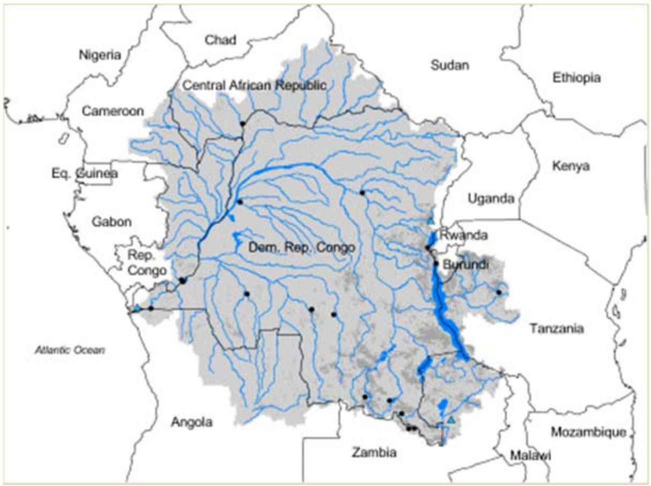


Figure 3.2 Map showing the Congo basin and overlapping countries

The climax ecosystem is tropical evergreen forest with little seasonal variation typified by irregular and very dense (70–100%) crown cover that often preclude the development of understory shrubs and grasses (Mayaux et al., 2004). Congo basin is one of the world's important biodiversity areas highly recognized for habiting the lowland gorillas, primates (Brooks et al., 2006) and elephants (Blanc et al., 2002). The total human population in the Congo Basin region was estimated at over 73 million people with an annual growth rate of >2.5% (Ndoye and Tieguhong, 2006) resulting in habitat fragmentation and loss due to agricultural expansion and unsustainable resource extraction. Other major threats to the ecosystem benefits include commercial trade in bushmeat and endangered species contributing to biodiversity loss.

3.3.2 Data sources and pre-processing

Before conducting the watershed and hydrological analyses, data which was acquired from different sources and collected under different scales had to be pre-processed and standardized.

3.3.3 Land Use Land Cover Map

Land use land cover is one of the most important ecological variables that is severely affected by human activities. Human activities that result in the conversion or use of land resources are considered to be the greatest challenge of the 21 century toward maintain ecosystem services (Lambin et al. 2001; Houghton et al. 2003; Foley et al. 2005). Land use land cover often used to be mapped by taking aerial photographs and cadastral techniques. With the advent of remote sensing technology, land use land cover is quickly monitored using remote sensing instruments mounted on satellites. Vegetation indices are radiometric measures of photosynthetically active radiation absorbed by chlorophyll in the green leaves of vegetation canopies (Tucker, 1979). Vegetative indices such as the Normalized Difference Vegetative Index (NDVI) and Leaf Area Index (LAI) are therefore good surrogate measures of the physiologically functioning surface greenness level of a region. Scholarly work by (Tucker, 1979) demonstrated how the Normalized Difference Vegetation Index (NDVI) generated from NOAA's Advanced Very High Resolution Radiometer (AVHRR) data can be used to map land cover and monitor vegetation changes and desertification at continental and global scales. NDVI has been shown to be very sensitive to ecosystem conditions (Goward, Tucker, & Dye, 1985; Ollinger, 2011) and of considerable value as an indicator of environmental change, particularly when most of the errors associated with atmospheric contaminants such as cloud cover and aerosols are reduced to minimum levels (Holben, 1986). The longest image time series of NDVI that has been produced is the archive developed by the Global Inventory Modeling and Mapping Studies (GIMMS) group at NASA's Goddard Space Flight Center from the Advanced Very High Resolution Radiometer (AVHRR) instruments on the National Oceanic and Atmospheric Administration (NOAA) Polar Operational Environmental Satellite (POES) series processed to yield a 30 calendar year record (1981–2011) with a spatial resolution of 1/12° (approximately 8 km at the equator) and a bi-weekly temporal resolution (Eastman, Sangermano, Machado, Rogan, & Anyamba, 2013; Zeng, Collatz, Pinzon, & Ivanoff, 2013).

Most of the spatial data was acquired from United States Geological Survey (USGS) Global GIS dataset of 2003 to implement watershed analysis and hydrological modeling. U S. Geological Survey's (USGS) Global Land Cover Characteristics (GLCC) dataset is derived from Advanced Very High Resolution Radiometry (AVHRR) satellite imagery produced at 1km resolution (Loveland, Merchant, Brown, & Ohlen, 1991). A detailed description of the methodology used to compile the USGS global land cover map can be found on USGS's official website http://webgis.wr.usgs.gov/globalgis/landscan/landscan.htm. Other sources of LULC maps include Food and Agriculture Organization of the United Nations (FAO) and the United Nations Environment Programme (UNEP)'s AFRICOVER produced and distributed by the Global Land Cover Network (GLCN), and GLOBCOVER produced by European Space Agency (ESA) and *Université Catholique de* Louvain (ESA and UCL, 2010). ESA's 2009 global land cover map was generated using 12 months worth of data collected from 1st January to 31st December 2009 from Envisat's Medium Resolution Imaging Spectrometer (MERIS) instrument while FAO and UNEP land cover map was essentially derived from visual interpretation of remotely acquired high resolution satellite images and using AFRICOVER Interpretation and Mapping System (AIMS) to develop vegetation class based on FAO Land Cover Classification system (LCCS) (Kalensky, 1998; Latham, He, Alinovi, DiGregorio, & Kalensky, 2002).

USGS Land Use Land Cover (LULC) map was evaluated alongside others data sources such as AFRICOVER and GLOBCOVER for accuracy based on representation of vegetation types, classification level and area coverage, and processing techniques and found to be the best map available for the Congo basin. For example, USGS LULC map was compared with ESA's

GLOBCOVER and the classification scales were quite different and vary in vegetation classification accuracy. USGS Global land use land cover map is composed of three different levels of vegetation classification, that is, the primary classification which is relatively finer with a total of 28 global cover classes, secondary classification which is coarse and the third level that is much broader at 11 classes. One obvious error identified in the USGS LULC was a slight mismatch between the polygon boundaries and the extent of the vegetation class. Technical errors might have arisen out of poor digitization of administrative and/or system boundaries and the vegetation cover classification by the remote sensing expert resulting in disaggregated classes. ESA's land cover map has 19 broad vegetation classes, but of particular concern is the misclassification of savanna grasslands and some patches of evergreen broadleaved forests into cropland/mosaic vegetation that were easily detected with the help of Google earth, high resolution satellite images from MODIS sensor and the researchers knowledge of the region. The USGS land use land cover map was reclassified by condensing the original 28 classes to 12 that occur in the study area (Figure 3.3). Before any analyses were done, the geoprocessing environment and spatial references were set to Datum: GCS_WGS1984 and projection: WGS1984_UTM_ Zone 35S, which is the appropriate zone for the study area.

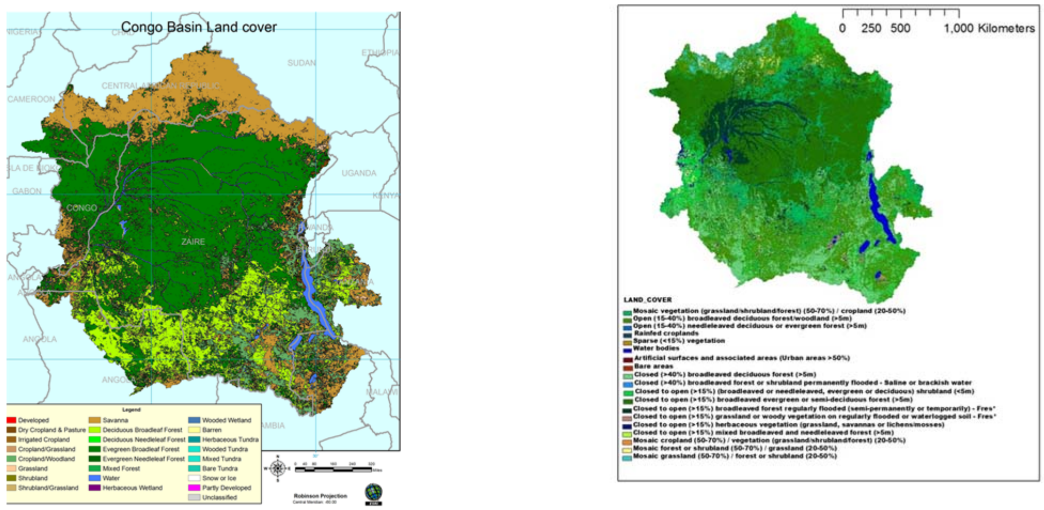


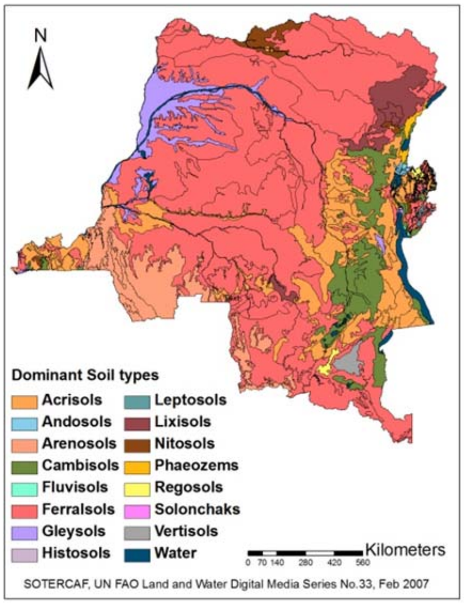
Figure 3.3 A comparison of different land cover maps for the Congo basin from USGS (left) produced in 2003 and European Commission Global land cover (right) produced in 2009

3.4 Watershed analysis

3.4.1 Soils of Congo basin

Watershed analysis, hydrological modeling and carbon stocks assessments rely heavily on a good land use land cover map representative of the landscape. Soils data was compiled from FAO's Harmonized World Soils Data (HWSD) version 1.1 and Soil and Terrain Database SOTER) for DR Congo (Figure 3.4), Rwanda and Burundi soils database produced recently by European Commission (FAO/IIASA/ISRIC/ISSCAS/JRC, 2009). The FAO's Harmonized World Soil database of 2008-2009 and that of JRC European Commission were generated at 1 km resolution (30 arc-second raster database with over 15000 different soil mapping units that combines existing regional and national updates of soil information worldwide) was used for modeling and analyzing the runoff and soil loss in the Congo River subwatersheds. I used the soils data from JRC/SOTER project to cross-validate the FAO-HWSD. The soils were reclassified according to the texture and drainage properties where soils in group A is sand, loamy sand or sandy loam with well to excessively well drainage properties, group B is composed of silt loam or loam texture and moderately well drained; group C is characterized by fine particles of sand clay loam and relatively low infiltration rate; and soils in group D are predominantly Clay loam, silt clay loam or sandy clay with relatively high infiltration rates (Brooks et al. 2003 p.449).

The USGS Global GIS dataset of 2003 geographic information system (GIS) database includes drainage basins, rivers and streams, large perennial water bodies. In this dataset, the basins and Sub-basins were delineated using the ArcHydro tools, a suite of GIS procedures that derive hydrological data and maps (HydroSHEDS, 2008) from high-resolution elevation data obtained during a Space Shuttle flight for NASA's Shuttle Radar Topography Mission (SRTM). The 3-arc seconds (approximately 90 meter at the equator) elevation model was derived from Defense Mapping Agency (DMA) map products and DMA digital terrain elevation data (DTED). This data is distributed free of charge by USGS and the Consultative Group of International Agricultural Researchers (CGIAR) - Consortium for Spatial Information (CSI) (http://srtm.csi.cgiar.org/) at varied resolutions (e.g. 30 m for the USA and 90m for the rest of the world) and geoformats. The 90 meter DEM was used to derive topographic variables such as elevation, slope, aspect, and the implementation of watershed and stream analysis. For this, analysis, flow accumulation value was computed for each position, and a 1000 km^2 threshold (as opposed to 500 km^2 that would result in very many subwatershed) was applied to obtain a drainage network in raster format.



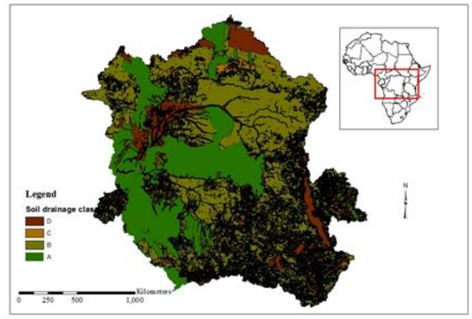


Figure 3.4 Dominant soils of the Congo basin (left) and the reclassified hydrologic soil groups starting with the most highly drained to the least drained (right)

This drainage network was vectorized using the STREAMLINK function, and each arc in the network is assigned with the maximum flow accumulation value from the set of raster elements used to derive it. Once the stream network was developed, then Pfafstetter watershed coding rules were applied to all the arcs draining directly into the ocean (Figure 3.5). The Pfafstetter convention of subdividing the area drained by the river was intended to increase ordinal values from downstream to upstream, and to assign odd digits to interbasins and even digits to basins. In this case, a zero digit is reserved for areas of internal, closed drainage. For the initial subdivision of a parent basin, there are five values (1,3,5,7,9) available for interbasins, and four values (2,4,6,8) for basins. This conforms to the topological fact that there will always be one more interbasin than basin, regardless of the manner in which they are labeled. In this study, level three and six were selected for further multivariate analyses and the summary attributes of each level are provided in table 3.1. Annual precipitation, potential evapotranspiration, roads, human population, political and administrative boundaries are also included in the USGS Global GIS dataset and compiled on a CD-ROM which can be accessed on request.

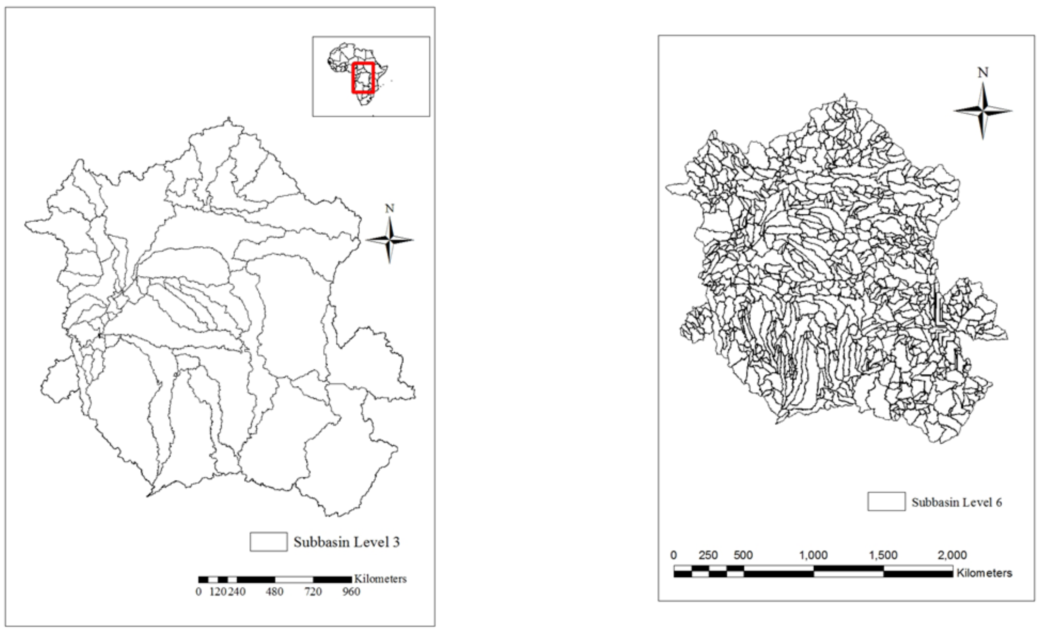


Figure 3.5 Level three and level six subwatershed

3.4.2 Runoff modeling

Surface runoff refers to the loss of water from an area by flow over the land surface. It occurs when rain falls with intensity greater than the rate at which it is able to infiltrate the soil. Runoff flow is composed of two main elements: base flow, which has its origin in ground water, and surface runoff, which is the accumulation of rainfall that drains to the stream. The characteristics of a watershed that affect the base flow and runoff include geology, soil type, vegetation cover, precipitation, drainage area and antecedent moisture condition (AMC) (Bellal et al. 1996). The Natural Resources Soil Conservation Service *CN* method is based on the relationships between rainfall depth, P in (inches), runoff depth, the stormflow, Q in inches, and storage factor, St (USDoA, 1986; Pilgrim and Cordery 1993; Schulze et al., 1992; Gumbo et al. 2002; Brooks et al. 2003; Senay and Verdin 2004):

Q is given as $Q = \frac{(\mathrm{P}-0.2\mathrm{St})^2}{(\mathrm{P}+0.8\mathrm{St})}$ and $S_t = \frac{1000}{\mathrm{CN}} - 10$ or $\mathrm{S_t} = \frac{25400}{\mathrm{CN}} - 254$ (if precipitation is expressed in mm).

The watershed characteristics are considered in determining the curve number (*CN*) index, which expresses a catchment's response to a storm event (USDoA 1986). The storage factor (also referred to as potential maximum retention, *S* in inches), represents an upper limit of the amount of water that can be abstracted by the watershed through surface storage, infiltration, and other hydrologic abstractions. It is important to note that *P* must be greater than 0.2*S* in order to produce runoff. If *P* is less than or equal to 0.2*S* then the runoff is equal to zero. Removing the initial abstraction (Ia) allows for a combination of *S* and *P* to produce a unique runoff amount (Pilgrim and Cordery 1993; Brooks et al. 2003 p.449). The maximum potential retention (*S*) is related to the soil and land cover conditions of the watershed through the curve number equation expressed as $CN = \left(\frac{25400}{254+\mathrm{St}}\right)$ and *S* is a dimensionless watershed parameter ranging from 0 to 100. A *CN* of 100 represents a limiting condition of a perfectly impermeable watershed with zero retention and thus all the rainfall becoming runoff. A *CN* of zero conceptually represents the other extreme, with the watershed abstracting all rainfall with no runoff regardless of the rainfall amount. The USDoA SCS has developed tables of runoff curve numbers corresponding to various land use, land cover types. These are listed in the SCS-SA User Manual (USDoA, 1986).

The Soil Conservation Service (SCS) Curve Number Method is a versatile and widely used procedure for runoff estimation because it gives consistently usable results (Rao et al. 1996; Sharma et al. 2001; Gumbo et al. 2002; Senay and Verdin 2004). The curve Number (CN) measures the relationship between initial abstraction and potential maximum retention of an area after a storm event. The presence of ArcHydro tool and a GIS-based Soil and Water Analysis (SWAT) tool have made it easy to compute surface runoff (Bhuyan et al. 2003; Zhan and Huang 2004; Fu et al. 2005) thereby provide timely information needed by decision makers such as natural resource managers and policy makers. This study heavily relied on GIS tools to compile runoff for the study area. The soil drainage data, however, had to be reclassified into four hydrologic soil groups that are suitable for Curve Number method modeling, following the US Natural Resources Soil Conservation Service drainage classification. The dominant soil group for DR Congo based on the EC/JRC SOTERCAF data is shown in Figure 3.4. The SCS (NRCS) method divides soils into four classes or hydrologic soil groups (A, B, C and D) according to their minimum infiltration rate/drainage properties, which is obtained for bare soil after prolonged wetting (Brooks et al. 2003 p.449). The soil drainage layer is then intersected with the land cover layer and later the appropriate curve numbers that correspond to the soil drainage and land cover were assigned in ArcGIS 10. Using the above equations, both the storage factor, St and the runoff Q were computed using the raster calculator function in ArcGIS. All raster layers were resampled to the 1-km by 1-Km resolution (i.e. 0.0083333338 if analysis is done in decimal degrees) before analysis. The results of runoff compiled using USGS, and European Commission Global land cover maps were quite different (Figure 3.6 and 3.7). The other possible sources of land cover land use maps freely available are AfriCover and FAO land cover, however, these share most of the data sources and quality with USGS Global land cover land use (LULC). As such, the results presented in this study for the runoff are based on both the USGS, and EC/JRC land cover maps for comparison purposes. A large portion of the Congo basin is covered by forests (i.e. evergreen broadleafed forests, evergreen deciduous forests) followed by savanna woodlands, cropland/pasture, shrubland, wooded wetlands and grasslands. Only a small portion of the entire 4.1 million square km is partly developed or built-up.

3.4.3 Soil loss modeling

The Revised Universal Soil loss Equation (RUSLE) has been extensively used to estimate soil erosion loss and to guide natural resource managers to develop soil and water conservation strategies (Millward and Mersey, 1999; Angima et al., 2003; Pham, et al., 2003; Lufafa etal., 2003). Due to the rapid development in computing power and Geographic Information System (GIS) applications, RUSLE has been used to study soil loss at a regional and global scale (Yang et al., 2003; Pham et al., 2003). The magnitude of soil erosion from terrestrial ecosystems is influenced by several factors interacting with each other such as climate, topograthy, soils, vegetation, human perturbations, tectonic instability and volcanic activities (Ferro & Minacapilli, 1995). Soil erosion rates are modified by sloe characteristics, particularly length, gradient, aspect, surface roughness, storm charactersitcs (Moore and Burch 1996) and amount of kinetic energy (McIsaac et al. 1987; Agassi and Ben-Hur, 1991). Soil loss was computed according to the equation stated below.

$$\mathrm{A(i,j)} = \begin{bmatrix} \text{Climate data} \\ \text{USGS (precipitation)} \\ \mathrm{R(i,j)} \end{bmatrix} \times \begin{bmatrix} \text{SRTM} - \text{DEM} \\ \text{(Slope, aspect)} \\ \mathrm{LS(i,j)} \end{bmatrix} \times \begin{bmatrix} \text{FAO Soil data} \\ \text{(soil properties)} \\ \mathrm{K(i,j)} \end{bmatrix} \times \begin{bmatrix} \text{USGS Land Use Land Cover} \\ \mathrm{P(i,j)} \end{bmatrix} \times [\text{MODIS NDVI}]\mathrm{C(i,j)}$$

Where: subscript *i and j =*i^{th} *and jth cell*

A = the average annual soil loss per unit area within the cell (t/ha/yr)

R = the rainfall runoff erosivity factor (MJ mm/ha.h.yr)

K = the soil erodibility factor (ton.ha.h/ha.MJ.mm)

L = the slope length factor

S = the slope steepness factor (dimensionless)

C = the cover/crop management factor (dimensionless)

P = the conservation support practice factor (dimensionless)

3.4.3.1 Soil erodibility (K- factor) (ton h MJ^{-1} mm^{-1})

Soil erodibility in the RUSLE equation is an empirical measure that expresses the inherent susceptibility of a soil to water erosion as determined by the intrinsic soil properties. The K factor is rated on a scale of 0 to 1, where 0 indicate soils with the least susceptibility to erosion while 1 indicates soils which are highly susceptible to soil erosion by water. It is one of the most difficult parameters to compute, particularly for areas with less field-based data on soil properties. Several methods have been proposed for estimating the erodibility from soil properties, which can be measured in the laboratory, such as particle size and aggregate stability (Torri et al., 1997). In this study, the K values for application in the Universal Soil Loss Equation, an empirical equation was derived using six components: percent silt plus very fine sand, percent organic matter, percent sand, soil structure and permeability, and this has been presented in the form of an easy-to-use nomograph (Wischmeier et al. 1971; Yang et al. 2003; Pham et al. 2003; Mokua 2009). Despite its wide use in the USA and now globally (Yang et al. 2003), this method of deriving the K values was also doubted about its application suitability on soils in tropical regions such as Hawaii, Tanzania and Nigeria (Vanelslande et al. 1984). Soil erodibility, can be determined on the basis of nomograms or calculating relations; Wischmeier, (1971) with due considerations of the granular-metric fractions of 0.002 – 0.1 mm, 0.1 – 2 mm, the organic matter content, the soil's structure and permeability. Determination of the soil erodibility involves assigning values that correspond to the soil types contained within the research area (Zhang et al., 2004). A simpler relationship was proposed by Renard et al., (1997) expressed in equation shown below.

$$100K = 0.0034 + 0.0405 * Exp[-0.5(\frac{LogD_g + 1.659}{0.7101})^2]$$

Where: **K**: is the soil erodibility (tons ha h / ha MJ mm); **D$_g$**: is the geometric mean weight diameter of the primary soil particles (mm). Dg is given as: $D_g = Exp(\sum f_{i\times} \ln \frac{d_i}{2} + \frac{d_{i-1}}{2}))$

Where: **d**$_i$ is maximum diameters of the particle size class *i;* **d**$_{i\text{-}l}$ is minimum diameters of the particle size class *i;* **f**$_i$ is sub-unitary percentage of the particle size class *i*. The same equation has be modified and re-written (Pham et al., 2003) in various ways and applied to compute K resulting in relatively the same results. In this study, the equation developed by (Wischmeier and Smith, 1978; Mhangara et al., 2011) and has been applied in South Africa (Mhangara et al., 2011) was used to derive the K values due to availability of data on soil structure, organic matter and permeability in the FAO World Soils data base (FAO/IIASA/ISRIC/ISSCAS/JRC, 2009). The equation is as follows:

$$\mathrm{K} = 2 \times 10^{-4} \times \mathrm{M}^{1.14} \times (12 - \mathrm{OM}) + 3.25(\mathrm{S} - 2) + 2.5(\mathrm{P} - 3))/7.59 \times 100$$

Where M = [%very fine sand + %silt] × [100 - %clay].

OM is percentage of organic matter; **S** is code according to the soil structure (very fine granular = 1, fine granular = 2, coarse granular = 3, lattice or massive = 4); and **P** is code according to the permeability/drainage class (fast = 1, fast to moderately fast = 2, moderately fast= 3, moderately fast to slow = 4, slow = 5, very slow = 6). **S** and **P** are represented as T_texture, and drainage in the FAO Soil database respectively. The soils in the region do not have very fine sand. As such, only the proportion of silt and clay were used to compute M. Both FAO's Harmonized World

Soils Data (HWSD) version 1.1 and Soil and Terrain Database (SOTER) soils database produced recently by European Commission (FAO/IIASA/ISRIC/ISSCAS/JRC, 2009) do not provide organic matter content directly, but can be derived from the top soils organic carbon content. According to Nelson and Sommers, (1982) this conversion factor assumes that organic matter contains 58% organic carbon. Thus, Organic Matter (%) = Organic Carbon (%) x 1.72414, of which, 1.72414 is merely a conversion factor.

3.4.3.2 Rainfall erosivity (R)

Erosivity is influenced by climatic factors, particularly precipitation depending on the amount, intensity, kinetic energy, and seasonality, temperature, evapotranspiration, wind velocity and vegetation cover. Rainfall erosivity factor, R (MJ mmha^{-1} year^{-1}) was determined through a raster calculation in ArcGIS 10 using Moore's equation (Moore, 1979). The equation is given as $R = 0.029(3.96P + 3122) - 26$ where, R = rain erosivity (joules per m^{-2}), p = annual rainfall amount (mm) and has been used in studies in Africa at a local or regional scale (Lufafa et al., 2003; Mhangara et al., 2011).

3.4.3.3 Slope length and steepness

Several methods have been developed and used to calculate slope length, and slope angle (Hickey et al., 1994; Hickey, 2000; van Remortel et al., 2001, 2004). In particular, Hickey (2000) produced an ArcInfo™ Arc Macro Language (AML) programs for creating a RUSLE-based LS factor grid using an input DEM (www.cwu.edu/~rhickey/slope/slope.html/). The AML code, however, requires an ArcInfo Workstation license in ArcGIS 9.3 software and is not compatible with ArcGIS 10. The same code, however, can now be edited to run from the command prompt of any computer provided the extension file for the executable file is properly renamed. The ***S*** constituent was calculated directly from the slope angle grid using two RUSLE algorithms

(McCool et al., 1997; Renard et al. 1997) that are differentially applied according to a break point at the experimentally modeled 9 percent gradient (Wischmeier and Smith, 1978). For slopes of less than 9 percent gradient, ***S*** is equal to: *10.8*sin(slope_angle + 0.03)* whereas slopes of 9 percent or more, ***S***, is equal to: 16.8*sin(Slope_angle - 0.50). The ***LS*** factor is subsequently calculated as the product of L and S elements. For this analysis, LS was calculated using both techniques developed by Renard et al., 1997; Pham et al., 2003), and the Arc Macro Language code (Hickey, 2000; van Remortel et al., 2004). The results were not different, except variations in the range of values.

$$LS = (\lambda/22.1)^{\S}(65.41S^2 + 4.56S + 0.065)$$

Where ***S*** is the land surface slope (m/m), λ *is the slope* length (m), and § is parameter dependent upon slope. The value of § varies with slope and it is estimated by:

$$\S = \frac{0.3S}{S + \exp(-1.47 - 61.09S)} + 0.2$$

3.4.3.4 Soil and Crop management (C-factor)

Erosion is greatly influenced by the extent to which the soil is protected from the energy of the rainfall or surface runoff by the vegetative cover. The soil and crop management factor (C) factor represents resistance of the ground surface to movement of soil mixture. The Normalized Difference Vegetation Index (NDVI), a spectral ratio between near infrared and red reflectance that is extracted from satellite imagery and is highly correlated with vegetative cover and biomass (Wang et al., 2002; Zhang, 2002; Sesnie et al., 2008; Karaburun, 2010). This relationship has be found to be useful toward creating the index of the effectiveness of crop management. Direct measurement of cover in the field using various methods such as aerial photography, high altitude ground photography, stereoscopic photography and those methods that measure the amount of light able to penetrate a crop canopy have been suggested. A quadrat siting frame, similar to instruments used in botanic surveys, and uses an inclined mirror which by reflection allows the operator to look upwards through the canopy and assess the density was developed and successful used in Zimbabwe in the late 1980s (Stocking, 1988). In this study, however, an attempt was made to conduct field-based measurements, but rather monthly NDVIs of 2009 acquired from MODIS satellite imagery were used to derive the C-factor following the equation developed by (van der Knijff et al., 2000). The NDVI derived C-values were later compared with those produced by FAO. Since the original C-factor of RUSLE ranges from 0 (full cover) to 1 (bare land) and the NDVI values range from 1 (full cover) to 0 (bare land), the calculated NDVI values were inversed using ESRI's ArcGIS 10 raster calculator present in the Spatial Analyst.

$C = Exp(-\alpha \frac{NDVI}{\beta - NDVI})$ Where α, and β: are iterative parameters determining the shape of the NDVI-C curve.

According to Ioannis et al., (2009), the appropriate value of α is 2 and that of β is 1 that give reasonable results. The values of the resultant C-Factor, however, were more than zero and extended beyond 1. Thus, a scaling factor, Z = 0.42696 ≈ 0.43 was used to spread the values of C within the range of 0 and 1. Consequently, the equation above was modified to $C = ZExp(-\alpha \frac{NDVI}{\beta - NDVI})$

3.4.3.5 Conservation practice-erosion (P – factor)

The conservation practice factor (P) represents erosion inhibition effect, and reflects resources users' value of soils and the control measures implemented to minimize soil erosion. The P-factor also referred to as the support practice factor is the ratio between soil loss with a specific support practice and the corresponding loss with upslope and downslope tillage. Assignment of P-values to the land use was done following the P-values proposed by FAO.

In order to account for any differences in water quantity and quantity at the subbasin level, a number of variables were selected to perform statistical analyses (Table 3.2). Drainage density using the formula $DD = \frac{Total\ length\ of\ channels\ in\ km}{basin\ area\ (km^2)}$ streamorder density using Terrain analyzing using Digital elevation model (TauDEM) a plug-in ArCGIS 10x. The rest of the valuables were compiled according to the methods explained in chapter two of this thesis.

Table 3.1 Biophysical data used in multivariate statistics analyses at the subwatershed level

	Variables (presented as means)	**Type**	**Reference/source**
1	NDVI	biophysical	MODIS 2009
2	Soil moisture	biophysical	ESO
3	Elevation (m)	Physical	USGS (STRM 90 m) DEM
4	Slope (m)	Physical	USGS (STRM 90 m) DEM
5	Runoff (mm)	Biophysical	Researcher generated
6	Soil loss (ton/ha/yr)	Biophysical	Researcher generated
7	Above & below ground carbon	Biophysical	NASA/JPL, Mitchard et al. 2008
8	Annual temperature (^oC)	Climatic	USGS
9	Annual precipitation (mm)	Climatic	USGS
10	Population density (persons/km^2)	Biotic	CIESIN
11	Road density	Physical	USGS
12	Streamorder	Physical	Researcher generated
13	Biodiversity (density/km^2)	Biotic	Researcher generated
14	Carbon (tons/ha)	Biotic	Saatchi et al. 2008; Baccini et al. 2012

Temperature, elevation, and precipitation are commonly used physical variables that are used to quantify turnover along ecological gradients. These abiotic predictors have been leveraged widely in conservation to address known gaps in existing knowledge on the distribution of species and ecosystems (Hortal and Lobo, 2005).

The use of physical variables to measure landscape scale changes relies on the assumption that the physical variables chosen act as a surrogate for biodiversity within the landscape and that differences within the descriptor variables (eg. temperature, elevation) are correlated with differences in species composition. Reliance on physical characteristics of the landscape for conservation classifications is frequently driven by asymmetry in information availability between physical and biological data (Jetz et al., 2012). Abiotic measures of environmental conditions are generally cheaper to acquire, provide more complete coverage and are more readily available than detailed empirical meteorological and hydrological monitoring data.

3.5 Gridded Population

A relatively high resolution Gridded Population of the World (GPW), v3 for the year 2000 was acquired from Socioeconomic Data and Applications Center (SEDAC), a NASA's Earth Observing System Data and Information System (EOSDIS) hosted by Center for International Earth Science Information Network (CIESIN) at Columbia University (Center for International Earth Science Information Network - CIESIN - Columbia University, & Centro Internacional de Agricultura Tropical - CIAT, 2005). The population data is freely available for download at http://sedac.ciesin.columbia.edu/gpw. A brief description of how the GPW data was processed is provided here. The GPW data processing has undergone several upgrades under the CIESIN's Global Rural-Urban Mapping Project (GRUMP) leading to the production of the most recent version GPWv3 (Deichmann, Balk, & Yetman, 2001). The first GPWv1 was produced using basic methods developed by Tobler et al. (1995, 1997). (Tobler, Deichmann, Gottsegen, & Maloy, 1997) collected the most recent population estimates available for each country according to the most recent census conducted in that country and data range from the year 1967 to 1999, and were projected to 1994 based on annual growth rates by country also provided by the United Nations. The mass-conserving gridding algorithm employed to distribute population from administrative units (usually provinces) to cells is purely cartographic and is based on population alone. The methods were modified slightly by (Deichmann et al., 2001) in the production of more recent versions; GPW v2 & v3.

In these versions, the population data was also transformed from their native [country level] spatial units, which are usually administrative and of varying resolutions to a global grid of quadrilateral latitude-longitude cells at a resolution of 2.5, a big improvement from 5.0 arc minutes scale. Gridding for each country was implemented individually and later merged the national grids to produce continental and global raster datasets of population counts (i.e. persons residing in each grid cell (Deichmann et al., 2001) resulting in major improvements in population estimation due to increase in spatial resolution and geoprocessing capabilities (Small and Cohen, 1997; (Small & Cohen, 2004). The grids use latitude/longitude reference system. Therefore, the actual size of a grid cell in square kilometers varies as a function of latitude, with a maximum cell size of about 21 square kilometers at the equator, 15 square kilometers at 45° and 5.0 square kilometers at 75° (Yetman, 2004) . By dividing the grids of population counts by the grid area, population density grids for the entire world were computed and a gridded map produced (Deichmann et al., 2001). The GPW data has been widely used by scientists all over the world to implement varied studies ranging from urban development (Small, 2004) improvements in settlement programs, natural hazards and disaster assessments (Small & Naumann, 2001; Small, 2004) to national planning of water resources. Like any other data produced at a global scale, GPW has several sources of possible error in the population estimates, which include inaccuracies in the interpolation method, the timeliness of the census estimates, varied numbers of population estimates and their accuracy. Inspite of the errors discussed above, the slight modifications to the processing and input resolution (Deichmann et al., 2001), make CIESIN GPWv3 (2000) (http://www.ciesin.org/GPWv3.html) data spatially more reliable and a good dataset to use in this study. The raster data are at 2.5 arc-minutes resolution and contain two data types: 1) population densities in 2000, unadjusted, persons per square km coded as ds00g; and 2) population densities in 2000, adjusted to match UN totals, persons per square km coded as ds00ag. The raster data for population densities in 2000, adjusted to match UN totals in persons per square kilometer (ds00ag) was preferred to allow for consistent comparison across countries that lie within the study area (Figure3.6).

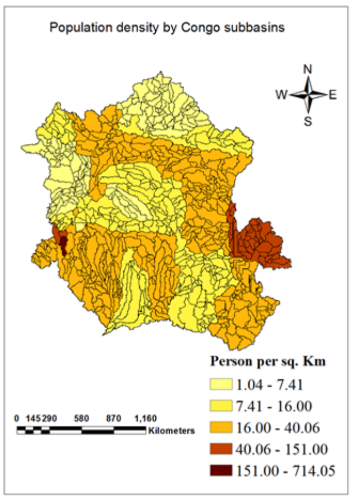


Figure 3.6 Human population density inside the Congo basin (CIESIN, 2010)

3.6 Environmental distance

Environmental distance measures have been used to support a wide variety of conservation applications. They have been used to delineate biological domains (Mackey et al., 2008), predict species composition (Ellis et al., 2012; Pitcher et al., 2012), identify regions at risk of climate change (Saxon et al., 2005), inform survey design (Hortal and Lobo, 2005), explain genetic diversity in populations (Mendez et al., 2010), to identify priority areas for the expansion of protected areas (Faith et al., 1987), and to promote connectivity (Van der Putten, 2012) in national scale adaptation planning (Game et al. 2011).

3.7 Results

The results of the water quality and quantity are provided and discussed under this section.

3.7.1 Sub-watershed zonal statistics

A number of environmental variables and biophysical variables were considered for the zonal bivariate and multivariate analysis (Table 3.1). Soil moisture is highly recognized as an important environmental factor that influences hydrological and agricultural processes, runoff generation, drought development and many other processes. It is also known to impact climate systems through atmospheric feedbacks, particularly serving as a source of water for evapotranspiration over the continents, and is involved in both the water and the energy cycles.

Ecological analysis frequently involves dividing the landscape into habitat types, biomes, ecoregions, or watersheds. Areas are grouped together based on relative similarities in biotic and abiotic composition within geography (Gosz, 1992). In this case, the Congo basin was our main geographical extent of analysis. The Congo watershed was delineated into 760 subwatersheds to utilize information available on the heterogeneity in the landscape and attempt to identify ecological gradients and quantify the continuous change across the entire watershed. A number of environmental variables and biophysical variables were considered for the zonal bivariate and multivariate analysis (Table 3.1). At the sub basin level, there were no huge differences in the means of the variables that were examined. For example, the mean NDVI for subwatersheds under level three (7085) was almost the same to that of subwatersheds in level six (7029). Similarly, the mean precipitation for level three subbasins (1581.54 mm) was almost similar to that of basins in level six (1519.0). Generally, there are no major differences between interbasins and basins except the mean area of the subwatersheds, which is typically expected considering the Pfafstetter’s coding rule.

Table 3.2 Subbasin level 3 and 6 attribute summary

Variable	Level 3 Mean	std	Level 6 Mean	Std
No of subbasins	108		720	
Area (km^2)	86797.33	85993.77	6171.6	6060.0
Stream order (X1)	9.06	1.73	7.4	1.6
Drainage density per m^2 (X2)	0.64	5.50	0.7	9.5
Slope % (X2)	1.97	1.10		
Elevation (m) (X3)	624.98	267.54	721.2	323.7
Road density/ m^2 (X4)	0.15	1.27	0.2	2.2
Population Density/m^2 (X5)	24.93	78.35	24.7	40.5
NDVI (X6)	7085.31	722.37	7029.0	903.0
Soil loss (tons/ha) (X7)	66.96	40.09	71.2	49.1
Runoff (mm) (X8)	1320.52	208.21	1269.9	260.9
Biodiversity (X9)	5.26	1.15	5.1	1.1
Carbon density(tons/ha) (X10)	91.85	49.95	86.1	55.8
Precipitation in mm (X11)	1581.54	237.40	1519.0	279.9
Temperature ^{o}C (X12)	24.09	1.13	23.7	1.5

Soil moisture is highly recognized as an important environmental factor that influences hydrological and agricultural processes, runoff generation, drought development and many other processes. It is also known to impact climate systems through atmospheric feedbacks, particularly serving as a source of water for evapotranspiration over the continents, and is involved in both the water and the energy cycles.

3.7.2 Runoff map for the Congo basin

The results of runoff compiled using United States Geological Survey (USGS) land cover map, and European Commission Global land cover maps showed very minor differences in the maximum and minimum runoff (Figure 3.7). The maximum and minimum runoff derived using the USGS was 2178 mm, and 346.8 mm compared to 2186 mm and 391 mm respectively when European Commission Joint Research Council (EC/JRC) land cover map was used. Differences in the development of these two sources of land cover maps are discussed under runoff modeling section in this chapter. In terms of spatial distribution of runoff, in both maps, high run off is shown to occur in the central northern part of the basin. This is an area served by the main tributary of the Congo River and highly forested. Generally, runoff in the Congo basin showed a high spatial variability. Areas of high runoff occur in subwatersheds with low vegetation cover, dense human settlement, and steep slopes.

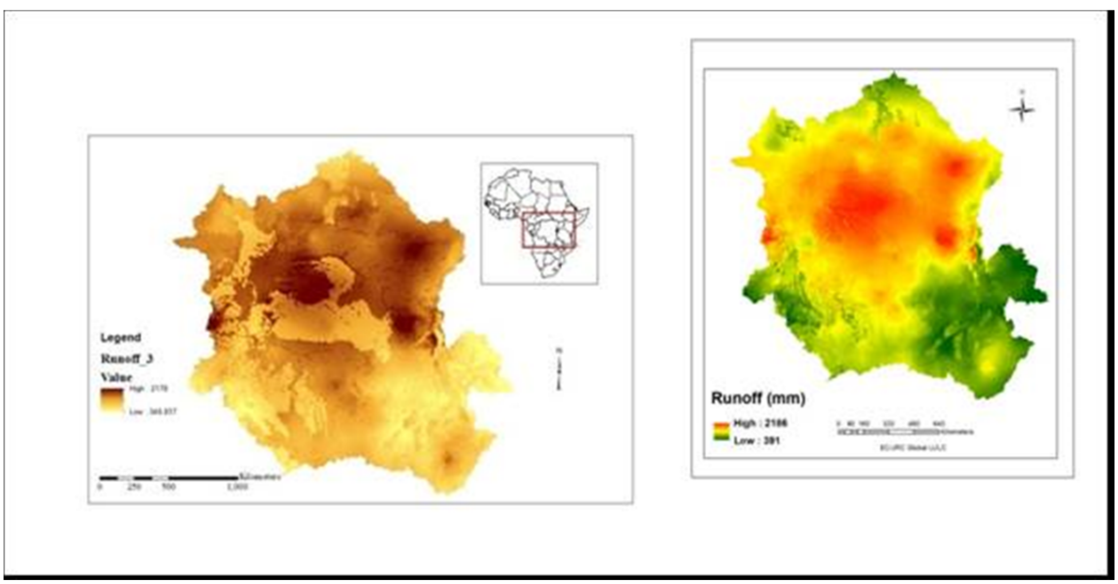


Figure 3.7 Runoff map for the Congo basin based on USGS land use land cover (LULC) map (left) and runoff based on EC/JRC Global LULC map (right).

The explanation for the high runoff could be attributed to the nature of soils, this area being a wetland and with lower slopes (>20%), the soils are already saturated with water and any addition water supply from precipitation will just runoff due to less infiltration. Sediments containing more clay and organic matter content tend to resist erosion more than the soils with more sand and silt due to the binding potential of the soil properties. The north eastern part of the basin also experienced relatively high runoff due to the high slopes dominated by human settlement and agriculture. The area has a reason amount of impervious surfaces. The southern part of the basin equally has high slopes but with less human settlements. The relatively dense vegetation cover and low human settlement make these areas less susceptible to runoff.

3.7.3 Soil loss map for the Congo basin

Results of the soil model showed that areas dominated by human settlement and cultivation on steep slopes experienced high soil loss whereas areas with dense forest cover and high slopes experienced very low soil loss. Soil loss occurred mainly in dense populated areas, the eastern part and around the largest cities like Kinshasa located in the western part of the basin. In some of the areas under heavy logging, particularly in the southern part, soil loss was occurring at a relative high amounts (60-70 tons/ha). In areas dominated by commercial crops mainly tea, a cover crop, soil erosion was very low (0.9 -7 tons/ha).

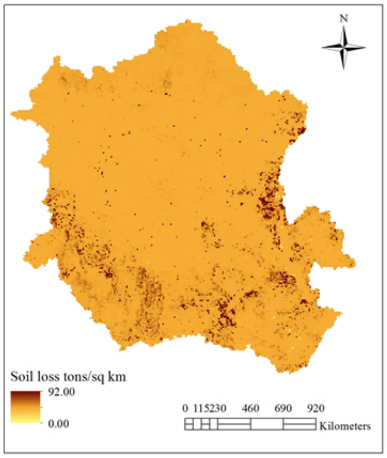


Figure 3.8 Soil risk map for the Congo basin

Results of the zonal statistics performed in ESRI's ArcGIS ver 10.2.1 for the subwatershed delineated at level three (interbasin) shown in Figure 3.9 showed some spatial variability within each and among the biophysical variables. In the case of surface runoff, the subasin with the highest runoff (1552 -1882 mm) are indicated in white background is located in central part f the basin, an area where the main Congo River is located. Subwatersheds in the southwest and southeast recorded the lowest runoff in the range of 651- 1058.9 mm. The subwatershed in the southeastern and northeastern region recorded the highest soil loss of 105-138 tons/sq km. There were differences in the quantity and distribution of different ecosystem services at the subwatershed levels three (Figure 3.9) and six (Figure 3.10). For example, soil loss was noted to occurred in higher amounts (168 tons/km^2) at level six of the subwatersheds in some of the subbasins than those in level three (138 tons/km^2).

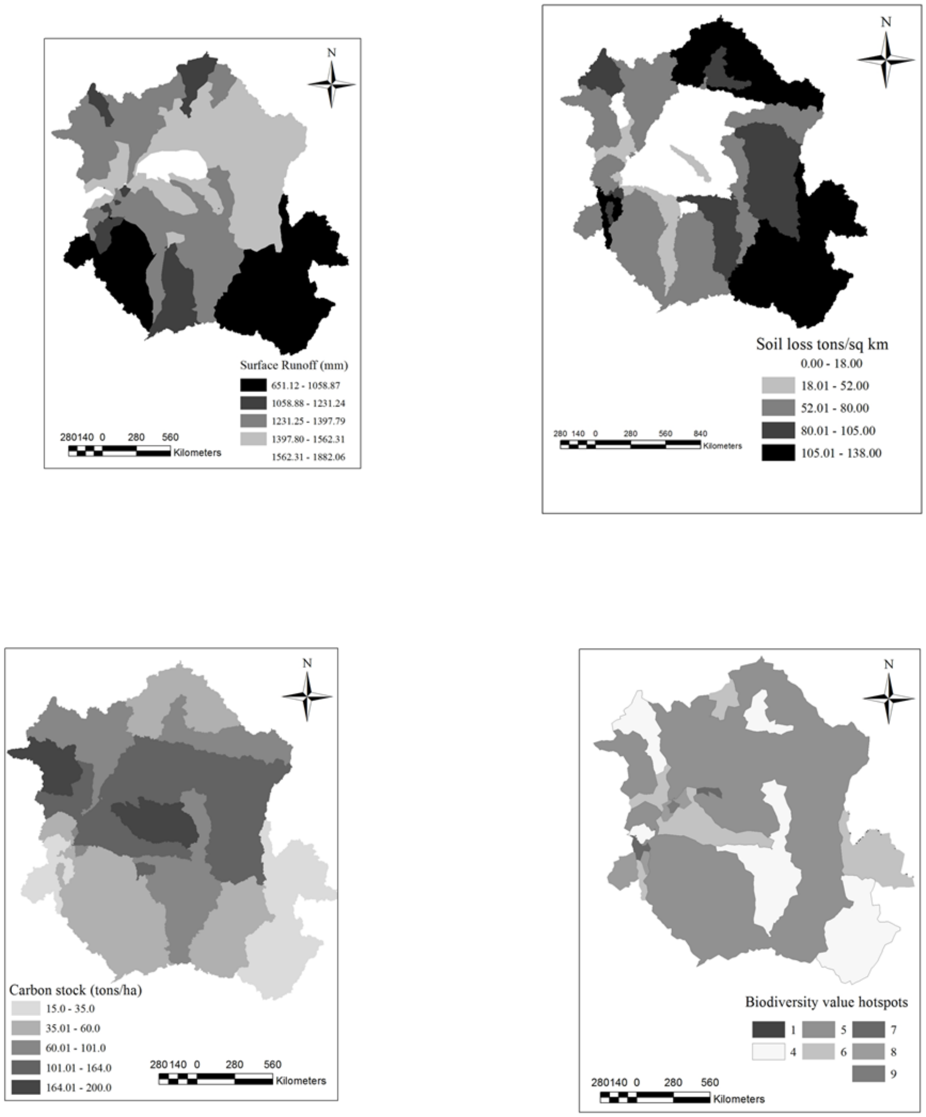


Figure 3.9 Level three subwatershed biophysical characteristics

There were also noticeable differences at the interbasin and basin scale within the two subwatershed levels. At the subwatershed level six (basins), a similar pattern and spatial distribution of runoff, soil loss, carbon stocks and biodiversity was recorded (Figure 3.10).

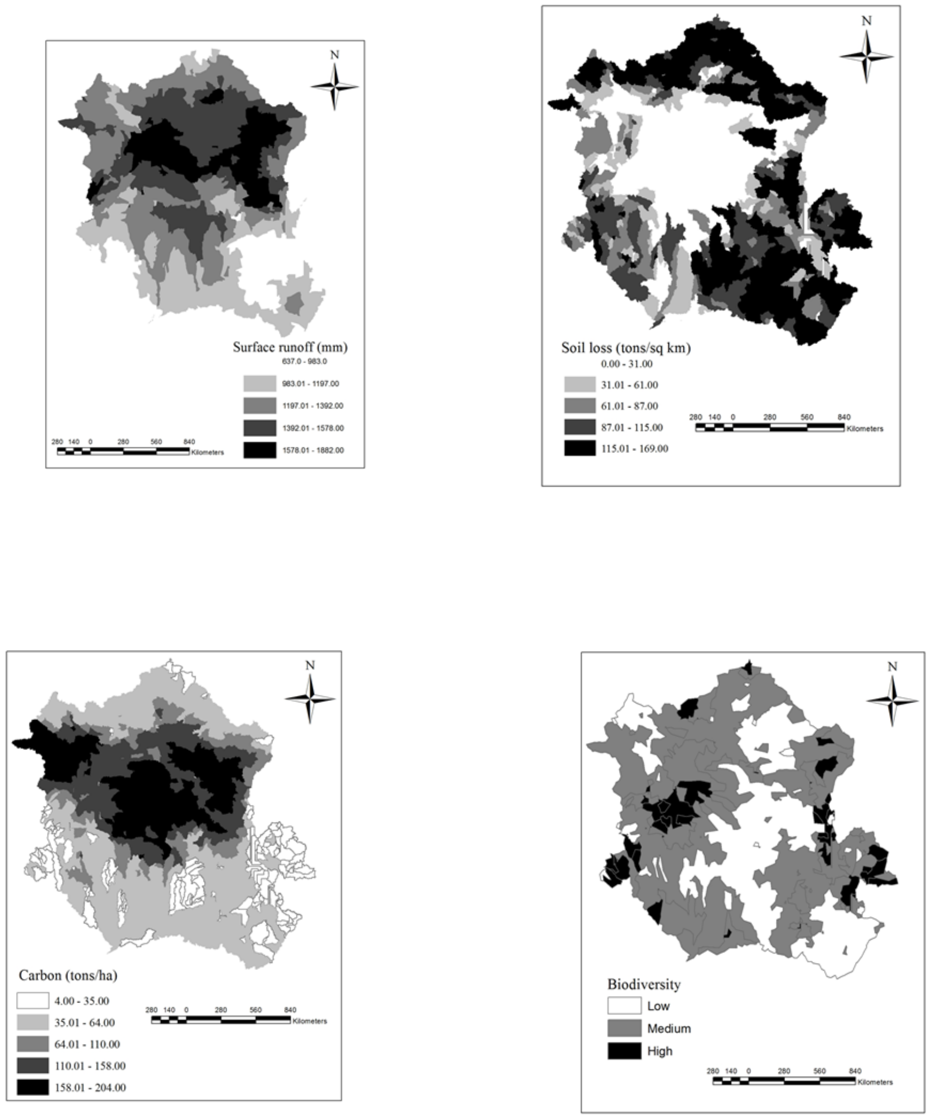


Figure 3.10 Level six subwatershed biophysical characteristics

3.7.4 Multivariate Analysis

A number of bivariate and multivariate statistics were conducted to assess the relationships between individual variables at the subwatershed levels three and six. Pearson correlation matrices and principle component analysis were performed to determine the variables that are highly (R^2 >0.6) with each other before implementing a regression analysis. Pearson correlation analysis is often used to determine the direction and magnitude of relationships between two variables under investigation helps to reduce data redundancy since highly correlated variables carry the same information. The table 3.3 provides a summary of the relationships measured by this statistic. In this case, temperature was negatively correlated with elevation (r = -0.81), precipitation (r = -0.68), and positively correlated with runoff (r = 0.62). This implies that as elevation, and precipitation increases, temperature decreases, which is a natural phenomenon. At high elevation there is a lot of vapor and get closer to the cloud cover, temperature gets reduced. Runoff was also positively correted with precipitation (r = 0.75).

Table 3.3 Pearson correlation coefficients for the subbasin variable in level 3

	X3	X6	X5	X11	X4	X8	X2	X7	X1	X12
Elevation (m) (X3)	1.00									
NDVI (X6)	-0.50	1.00								
Population Density/m^2 (X5)	0.08	-0.19	1.00							
Precipitation in mm (X11)	-0.69	0.65	-0.13	1.00						
Road density/ m^2 (X4)	0.08	-0.07	-0.01	-0.07	1.00					
Runoff (mm) (X8)	-0.57	0.63	-0.18	0.75	-0.05	1.00				
Slope % (X2)	0.28	-0.37	0.24	-0.28	0.04	-0.39	1.00			
Soil loss (tons/ sq km (X7)	0.50	-0.52	0.17	-0.62	0.01	-0.45	0.28	1.00		
Stream order (X1)	-0.16	0.00	0.02	0.06	-0.20	0.01	-0.03	-0.06	1.00	
Temperature °C (X12)	-0.81	0.56	-0.13	0.68	-0.08	0.62	-0.36	-0.47	0.13	1.00

The spatial distribution of the variables information above can be visually observed in the scatter plots (Figure 3.11).

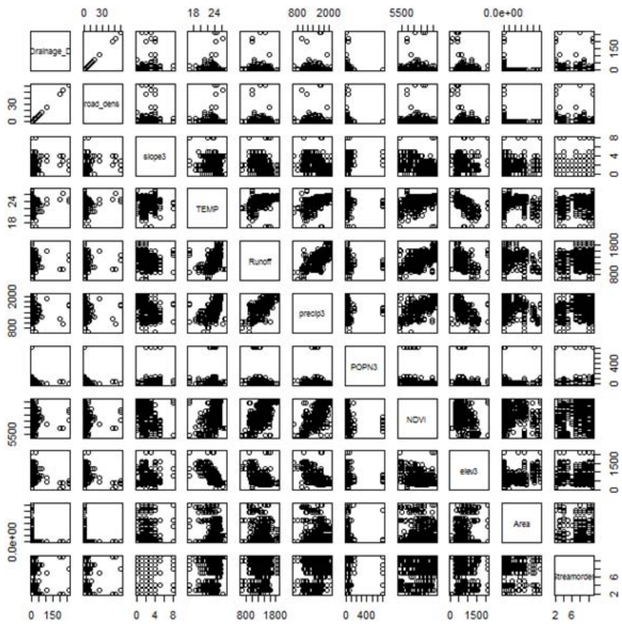


Figure 3.11Spatial distribution of subwatershed characteristics

Soil erosion was noted to occur in farmed and deforested areas mainly in the southern part of the basin, near urban centres such as Kinshasha in the west, territory of Luberu, and the surrounding areas of Kasennyi south of Lake Albert. The International Agency for Inspection of Nuclear Engergy expressed concern that soil erosion threatens the safety if nuclear plants in Kinshasha (UNEP, 2011). The main causes of soil erosion noted are mainlydue to poor land management, intensive agriculture without soil and water conservation practices.

3.8 Policy implications

The results from this study show that some of of the basin are exeperincing soil erosion to the maximu of 70 tons per ha. The worst affected areas are those where vegetation cover is being removed through timber harvesting, commercial and subsistence agriculture, and highly populated areas. Equally, the souther part of the basin is experiencing reduced surface water while the eastern part of the basin which also coincides with the urban areas is experiencing a lot of runoff. If soil and water conservation measures are not encouraged the water scarce areas will get worse. Extraction of minerals and timber are very intensive land use activities that expose the soils to direct solar radiation as well as changes the soil textural and drainage properties due to compaction. The construction of logging roads and the digging of mining tunnels and ditches also contribute to land cover clearance and is the likely cause for the high soil erosion that is being revealed by the results. Given the low scale of soil loss across the entire watershed, this loss is likely to continue unnoticed because the region has very few functioning research centers and even the few that do exist such as *Institut National pour l'Etude et la Recherche Agronomiques* (INERA) located in North Kivu, eastern DRC, lack adequate funding and their priorities are focused more on increasing agricultural production and experimenting with new crop varieties. The other important research center that is entired devoted to monitoring and researching on hydrological process, *Centre de Recherche Hydrogigue* (CRH), Uvira is equally domant due to lack of research funds. The only active research institute is the *Observatoire Volcanoligue de Goma* in charge of monitoring volcanic eruptions on Nyiragongo Mountain. There are a number of weather stations formerly established by the Belgian-colonial government, but these are dysfunctional due to insecurity, limited funding and inaccessibility.

3.9 Conclusions

The study has showed that the central region of the Congo basin is experiencing more runoff and less soil erosion. This area, however, is characterized by inundated wetlands whose soils are moderately to fairly drained. The natural presence of the main

Congo tributary hinders people from conducting activities that would facilitate runoff and soil erosion. At the sub basin level, some of the subwatersheds that are dominated by human settlements and cultivation, runoff and soil loss are relatively high compared to other places. The main conclusions are that runoff in the Congo basin is quite varied and not uniform as it was hypothesized. At the sub basin level, subwatersheds in mid central part of the basin are experiencing a lot of runoff and this is not necessary due to human disturbance given the high forest cover, rather it due to the presence of a high water table and saturated soils implying that no more infiltration can take place. These subwatersheds also have high streamorder to the size of 7-9. It is possible that after a storm event, the free water is drained into the river channels. The other conclusion that can be inferred from the results is that soil loss is that occurring at a very high magnitude compared to other parts of Africa such as the Lake Victoria basin where Lufafa et al (2003) reported very high soil loss.

Lastly, it is reasonable to say that this study has made some substantial contributions to the understanding the hydrological process in the Congo River Basin as well as generating useful information for the management of water resources in the region. The lack of empirically collected information on the streamflow, surface runoff, and soil erosion studies still continue to be a challenge for scientist that attempt to understand the hydrology of the Congo basin (Tshimanga and Hughes 2013). To circumvent this problem, this study relied on expert opinion that provided useful comments on the runoff and soil risk map. The next step is to disserminate these findings among the relevant institutions to get them to appreciate the challenges and begin to take positive steps toward developing strategic action plans for the management of Congo basin.

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CHAPTER 4

AGE-CLASS SPECIFIC ELEPHANT POPULATION DYNAMICS: INFLUENCE OF CLIMATE, WARS, AND POACHING

4.1 Introduction

The African savanna elephants (*Loxodonta Africana Africana*; Blumenbach) are among the long-lived mammals with a relatively slow rate of reproduction and high investment in the offspring. Elephants are a species of global conservation concern due to a dramatic reduction numbers over the last 100 years and now occur in specific locations as highly discontinuous populations (Blanc et al., 2007). In Africa alone, it is estimated that there are less than 500,000 individuals left in the wild and a bigger number is located in southern Africa. These declines have largely been attributed to legal and illegal trade in ivory, and more recently to competition for space and conflict with people. In the Greater Virunga Landscape (GVL), a part of the Albertine Rift (Africa), elephants are such an important component of the ecosystem. Elephants have been reported to play a major role in vegetation distribution and change (Buss, 1961; R. Laws, 1970; Wing & Buss Irven O., 1970). It is therefore not surprising that most studies in the last four decades focused on understanding the role of elephant toward vegetation changes (Buechner, H.K., Buss, I.O., Longhurst, W, M.,Brooks A.C., 1963; Field, 1971; Guldemond & Van Aarde Rudi, 2008; R. Laws, 1970; Rasmussen, Wittemyer, & DOUGLAS-HAMILTON, 2006). The next set of earlier studies focused predominantly on elephant reproduction and population dynamics in Africa (Buss, 1961; Hanks & McIntosh, 1973; R. Laws, 1966; R. M. Laws, 1970); Western, Moss, & Georgiadis, 1983; Western, Moss, & Georgiadis, 1983; (Moss, 2001; Western, Moss, & Georgiadis, 1983). Despite the effort made to understand the ecology of elephants, less emphasis was placed on anthropogenic influences on elephant population dynamics. In fact some hypothesis that attempted to explain the importance of humans in influencing elephant population changes proved to be controversial (Caughley, 1976)

Recently, controversies also emerged concerning the differences between savanna and forest elephants, proponents of the separation between these two arguing that the later is a different species while the opponents suggesting that they should be treated as subspecies and not necessarily different species. Three papers published in the last two years on the genetics of the African elephant (Rohland et al., 2010; Ishida et al., 2011a; and Ishida et al., 2011b) argued for the separation of the African elephant into two species. While there are still outstanding queries from the AfESG's "Statement on the Taxonomy of extant Loxodonta" (AfESG, 2003) which have not been satisfied, a more practical problem is where exactly to draw the geographical line between the two potential species. Until this query and the outstanding research questions from the 2003 AfESG statement have been fully clarified, the AfESG considers it premature to divide the African elephant into two species. Some recent genetic research has suggested that the forest elephant is genetically different from the savanna elephant and represent a separate species of elephant (Comstock et al. 2002).

Different mechanisms and management strategies such as culling programs, collaborative natural resource management, community conservation, and legalized one-off sale of stock piles, (which are discussed in detail in the subsequent sections), have been tried to address the challenges of elephant conservation, particularly elephant poaching without much success. These strategies climaxed with an international convention to regulate trade in endangered species and their products. The Convention on International Trade in Endangered Species of Wildlife and Fauna (CITES) was instituted to regulate trade in species and their products (CITES 1989; Stiles 2004). The African elephant is currently listed as Vulnerable (A2a; Ver 3.1; Blanc, 2008) in the IUCN Red List. At the 7th Conference of Parties (CoP) held in 1989, the elephant was listed on Appendix I and trade in live animals or its body parts was banned.

4.1.1 Problem

It is widely recognized that understanding the interaction between elephants and the vegetation cover is critical to the design of management and conservation strategies of the species and its habitat. It is equally important to understand the factors that influence the elephant behavior and the essential demographic parameters that determine their population dynamics. Single attribute models are crucial to understanding wildlife population dynamics and provide useful information in the design of management strategies. To be reliable, demographic models should preferably rely on easily obtainable variables that are directly linked to the ecological processes regulating a population. Fluctuating ecological conditions are often key factors influencing carrying capacity, mortality and reproductive rates in ungulates. On the other hand, anthropogenic factors such as war, fires, climate change, and socioeconomic changes, which indirectly affect natality and mortality rates, need to be considered in modeling elephant population dynamics. Information on elephant range and numbers is vital for the effective conservation and management of Africa's elephants.

The elephant is a keystone species that plays a pivotal role in structuring both plant and animal community and often dominates mammal biomass in the habitats it occupies (White 1994). While the effect of the elephants on its habitats is often beneficial (Chapman et al. 1992; Ruggiero and Fay, 1995; Eltringham 2008), it can have a detrimental impact on vegetation where high densities build up in confined areas (Hoft and Hoft 1995). In the Greater Virunga Landscape

(GVL), elephant populations are mainly confined to protected areas, however, most of their range stretches into human settlements and agricultural areas. This poses additional challenges for wildlife authorities and wildlife managers (Plumptre et al 2007, 2008) to manage such a flag species with wide home range. War, poaching, and climate change pose an eminent threat to the conservation of wildlife, unfortunately, the combined action of these factors are not well understood. Thus, wildlife management authorities need to be aware of the consequences of war, climate change, and poaching on the age structure of elephant population in the landscape to adequately manage and conserve the remaining population.

Elephants do not observe administrative or political boundaries, adjusting their management to socio-political realities can present a challenge. Countries in the Albertine Rift region have established collaborative management frameworks that allow all interested parties to be involved in the development of management plans (e.g. Central Albertine Rift Strategic Plan; Transboundary Core Secretariat 2006), and implement them jointly for the common good (Plumptre et al.2008). In the study area, region-wide information is required because elephants move long distances across protected areas and international boundaries, and a policy or management decision made in one country can ultimately affect elephant populations in other countries. Changing land use pattern or different management approaches such as trophy hunting in border areas or culling exercises to reduced elephant densities in one area may cause severe impacts on other populations beyond sovereign boundaries. It is assumed that elephants should be able to move freely to other areas within the landscape once food and water resources are limiting, however, the presence of sustained civil rebellions in the region impose a movement constraint and expose the elephants to more death risks. Furthermore, wars have destroyed livelihood strategies for the rural population causing them to depend heavily on wildlife resources for their welfare. As such, poaching of elephants is exacerbated by the scale of poverty, increase in human population and the demand for ivory.

4.1.2 Current approaches

Matrix models have proved useful tools in population dynamics to predict population growth given their capabilities to adequately describe the evolution of species population by age structure (Crouse, Crowder, & Caswell, 1987). Although the use of matrix models in population dynamics started way back in the 1940 (Bernardelli, 1977; D. Lewis, 1942; E. Lewis, 1977), the core principles of species population modeling were developed by Leslie (Leslie, 1945; Leslie, 1948). The Leslie model (Leslie 1948) was developed based on three core hypotheses, that is, age is a continuous variable starting from zero and subdivided into discrete age classes; 2) time is a discrete variable; and 3) the time step (projection interval) is exactly equal to the duration of each age class and all individuals in a population follow this time step. Leslie (1945) noted a strong limitation in previous population models, where, fecundity and death rates were treated as constant among all individuals in a population, which is not necessarily true. Matrix models have been used to determine the best fishing strategies and design of endangered species rescue programs. In ecology, matrix models were popularized by (Lefkovitch, 1965) who introduced structuring of populations and further developed by (Usher, 1966). Projection matrix models are now commonly used in modeling population demographics ranging from large mammals, fisheries to human population because they are relatively easy to formulate, compile complex data in a structured and analytically tractable manner, provide parameters with direct biological meaning, can be applied to general or specific, experimental and/or theoretical situations, including ecological and evolutionary questions (Salguero-Gómez & De Kroon, 2010).

Unlike the elephant–vegetation models, Baxter & Getz (2005) used a grid-based model of elephant–savanna dynamics to assess the interaction of woody plant demographics, tree–grass interactions, stochastic environmental variables such as fire and rainfall, and spatial contagion of fire and tree recruitment in the southern African savannas. Although developed primarily as a tool for investigating elephant impacts, Baxter & Getz (2005) concluded that the model can also be used to investigate the behavior of systems involving different environmental conditions and tree functional types and to explore other scenarios such as changes in fire management or rainfall regime(Baxter & Getz, 2008). Other elephant study techniques involve the use of satellite collars equipped with a Global Positioning System (GPS) and a radio transmitter to monitor habitat use and movement of elephants (Boettiger et al., 2011). Remote sensing is another recent technique being used widely to understand the spatial and temporal characteristics of landscapes with respect to wildlife use, climate change, and other human induced changes that affect ecosystems (Boone, Thirgood, & Hopcraft, 2006). Lastly, Statistical bioclimatic envelope models, which represent one specific type of species distribution models (also referred to as habitat models, or ecological niche-based models (Guisan and Zimmermann, 2000) whereby the biogeographical distributions of species are related to broad-scale variation in climate by given modeling techniques (Franklin, 2010; Franklin, 1995; Guisan & Thuiller, 2005; Guisan & Zimmermann, 2000). Species distribution models (SDMs) are numerical tools that combine observations of species occurrence or abundance with environmental estimates (Elith & Leathwick, 2009; Franklin, 2010; Heikkinen et al., 2006). The most commonly used techniques in species modeling include BIOCLIM (Araújo, Pearson, Thuiller, & Erhard, 2005; Beaumont & Hughes, 2002; Beaumont, Hughes, & Poulsen, 2005), HABITAT (Walker & Cocks, 1991), DOMAIN (Carpenter, Gillison, & Winter, 1993), MaxEnt (Elith et al., 2011; Phillips & Dudík, 2008), and machine learning techniques such as Artificial Neural Networks (ANN), Generalized Additive Models (GAM) (Garcia, Burgess, Cabeza, Rahbek, & Araújo, 2012).

4.1.3 Justification of the study

In the past, a number of decisions concerning the management of elephants, particularly culling or cropping programs of the 1960s (R. Laws, 1970; Macnab, 1991) and lifting of the ban on the trade in elephant products, particularly ivory on the international market (CITES 1997; Aarde and Ferreira 2009) were based on trends in numbers and poaching derived from Elephant Trade Information System (ETIS) and Monitoring of Illegal Killing of Elephants (MIKE) program, which information doesn't account for the spatial and temporal variables that influence the elephant population dynamics. Most studies that have examined the impact of civil strife (de Merode et al., 2007; Draulans & Van Krunkelsven, 2002; Eltringham & Malpas, 1980; Plumptre, Bizumuremyi, Uwimana, & Ndaruhebeye, 1997), poaching(Maisels et al., 2013; Michelmore, Beardsley, Barnes, & DOUGLAS-HAMILTON, 1994), habitat change (DeFries, Foley, & Asner, 2004) , and climate change (Foley et al., 2005; Wiens & Bachelet, 2010) have focused on each stressor independent of the others and yet all these factors occur simultaneously and need to be treated as a system. van Aarde et al. (2008) rightly argued that death and birth rates vary between elephant populations and the demographic variations confounded by unknown effects of movement across international borders makes the interpretation of elephant population variations unrealistic. In Greater Virunga Landscape, elephants in Virunga National Park (DRC) move back and forth to Queen Elizabeth National Park regularly and vise versa (Plumptre et al. 2008). More so, with the exception of the 2010 elephant census conducted by WCS in partnership with UWA and ICCN (Plumptre et al. 2010), in the past, quite often the censuses in these two major national parks would be done at different times and by different observer teams resulting in very huge differences in the observed elephant populations. This region also continues to suffer from war perturbations bringing untold suffering to the people, including physical and economic displacement, encroachment on natural ecosystems, and loss of wildlife due to hunting and reiteration killings resulting from loss of livestock and crops to wildlife raids. It was therefore important to investigate the impact of civil strife, poaching, and climate change on elephant population dynamics over a long period, and provide recommendations for the managers to plan for these negative impacts and design better management strategies.

4.1.4 Uniqueness of study

One of the central activities in wildlife management is habitat improvement and the major cause of wildlife population decline is habitat loss and fragmentation due to human developments. Changing patterns of land use have fragmented the habitat of animal species such as elephants to the extent that they exist as semi-isolated subpopulations in remaining patches of suitable habitat connected only by relatively narrow migration corridors. Dynamics of wildlife populations depend on both habitat quality within the remaining patches, including watering sources and environmental conditions. Localized impacts on vegetation have cascading effects on biodiversity because changes in vegetation cover and composition induced by humans affect habitat suitability for many other species. In order to achieve effective management of large wild herbivores, it is important that wildlife managers have a clear understanding of how changes in habitat quality, severity of war, climate change, and poaching operating varied spatial scales can influence wildlife population, specifically elephants. It also requires long-term planning based on deep understanding of how population processes such as birth, and death rates, age structure are affected by changes in habitat size and quality, climate, civil wars, and poaching and how these influence management decisions.

Understanding the dynamics of age-class specific elephant population with respect to multiple stress factors operating at varied temporal and spatial scales is very useful, but quite complex to implement using species distribution models. This study demonstrated that you can use a systems thinking and experimental laboratory learning analysis (STELLA) to integrate the density-dependant population demographic theory and anthropogenic factors to understand at a local scale population dynamics of elephants in a defined system boundary. The system boundary does not have to be closed, but a good conceptual design to account for the immigrations and emigrations, habitat change, war impacts, water resources availability and climate change, all treated as one system is all that is very important. High human population density, hunting intensity, absence of law enforcement, poor governance, and proximity to expanding infrastructure are the strongest predictors of African elephant population decline (Maisels, et al. 2013). Mortality and natality rates at an age-class specific level is modeled bearing in mind the factors that increase mortality such as poaching, war, and changes in resource level mainly habitat quality and water resources, and indirect causes of poaching of elephants. Conceptually, poaching is exacerbated by poor or ineffective governance of natural resources resulting in low enforcement of rights, reduced financing for natural resources management and conservation, and human population increase leading to encroachment of conservation areas, including obliteration of wildlife corridors.

More recently, an effort was made by MIKE to relate the PIKE trends with the socioeconomic drivers of elephant poaching (CITES 2014: SC65 Doc. 42.1 – p. 23). The MIKE study noted poverty, governance and market demand as the key drivers of elephant poaching across African sites. In this study, CITES (2014) concluded that the above factors identified explained nearly two thirds of the variation observed in PIKE levels across African sites. In particular, poverty and governance explained spatial patterns in poaching levels, while market demand explained the observed temporal trend and concluded that the empirical relationships were insufficient in providing a direct link between the observed PIKE and the causal factors, but they do provide a good basis from which to investigate causation of poaching in Africa. More so, poverty, poor governance and increased market demand were identified as key factors likely to facilitate or provide incentives for the illegal killing of elephants and illegal ivory trade. This study, however, did not attempt to examine all these casual factors as a system.

4.1.5 Objectives

The general objective was to assess the effects of war, poaching, habitat and climate change on elephant population in GVL. Specific objectives were: 1) develop a comprehensive system dynamic model of age class specific elephant population; 2) assess the impact of habitat change on elephant populations in GVL; 3) assess the effect of poaching on elephant population; 4) assess the impact of war on elephant population; and 5) assess the effect of climate change on elephant population.

4.1.6 Hypothesis

The hypotheses tested were: i) Habitat change from forest to savanna grassland has no effect on age-specific elephant population; ii) war has a less negative impact on elephant population than habitat change and poaching; iii) there is no strong relationship between elephant population size, water resources and anthropogenic factors (i.e. climate change, war, poaching).

4.2 Literature review

This section provides a detailed review of the existing literature related to the African savanna elephant covering mainly the population trends at a continental, regional and local scales, habitat changes, historical account of the civil wars, poaching, and management and conservation initiatives.

4.2.1 African elephant population trends

In order to estimate the elephant population in the region and elsewhere, two methods commonly used are direct and indirect sample counts (Laws et al.1975; Barnes et al.1999; Blanc et al.2003; Blanc et al. 2007). Direct sample counts are commonly conducted from the air referred to as aerial count and performed according to Norton-Griffiths method (Norton-Griffiths, 1978), but may also be done on the ground either on foot or from vehicles. Indirect sample counts offer the most objective estimates, especially in forested landscapes where it is difficult to see animals from the air, however, it is very tedious and time consuming. Under this method, elephant dung is counted along transects using distance sampling techniques and results are combined with estimates of elephant defecation rate (nearly 20 times per day) and dung decay rate (149 days depending on the vegetation cover, temperature, and relative humidity influenced by topography; to provide a population estimate with confidence limits (Barnes, 2001; Inogwabini et al., 2000; Plumptre, 2000; Plumptre & Harris, 1995). A new indirect sample counting technique has recently been introduced for estimating the size of elephant populations in forest (Eggert et al. 2003). The technique relies on the extraction of genetic material (DNA) from as many dung piles as possible within a given area, and the use of DNA fingerprinting methods such as microsatellite, electrophoresis to identify the number of unique alleles or genotypes (individuals) at each locus in the sample collected. The rates of repeat samples obtained can then be used to estimate the population size for the area using the equivalent of a mark-recapture census technique (Eggert et al. 2003).

Analysis of the largest survey dataset ever assembled for forest elephants (80 foot-surveys; covering 13,000 km; 91,600 person-days of fieldwork) revealed that population size declined by 62% from 2002–2011, and the taxon lost 30% of its geographical range (Maisels et al., 2013). The population is now less than 10% of its potential size, occupying less than 25% of its potential range. According to the fifth report on the status of the African elephant (2013) produced by the African Elephant Specialist Group (AfESG) of the IUCN Species Survival Commission (SSC) based on data from the African Elephant Database (AED), the total elephant population is estimated to be 433,999, indicating a big decline from 472,134 reported in 2007 (Blanc et al.2007). Of this population, southern Africa has the highest population (267,966) followed by East Africa (130,859), Central Africa (16,486) and West Africa (7,107) in that order.

4.2.2 Elephant trends in Greater Virunga National Park

There is historical evidence to show that people used to live in the present day Virunga, and Queen Elizabeth National Parks dated nearly 50,000 years (Spinage 1970). To support this fact, Languy and deMerode (2006) discovered fossil remains of an early hominid at the point where Semliki River leaves Lake Edward in Virunga Park. In the late 1800s, most people living in the parks together with their livestock abandoned the area due to an outbreak of rinderpest and sleeping sickness. As a result, the numbers of wild animals started to increase. In the late 1950s and 1960s, large mammal numbers were at their highest and were forced to migrate to other areas because of corresponding increase in human population.

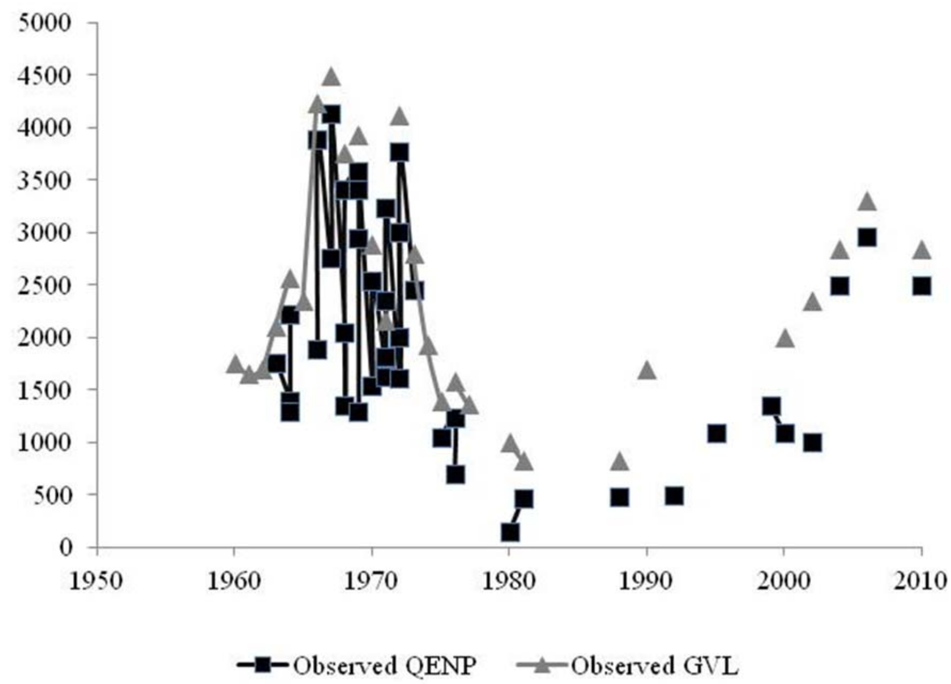


Figure 4.1 Elephant population trend in GVL and QENP (Uganda) from 1960s -2010

During this period, the biomass of large mammals recorded was 31.4 tonnes per square kilometer in the central sector of VNP and QENP (Cornet D'Elzius, 1996). It is believed that the collapse of the Uganda National Parks management in the 1970s led to massive killing of large mammals in QENP and the elephants migrated to DR Congo side to avoid being killed. The elephant population declined from around 3,000 to 400 in 1980 in QENP alone, only to rebound in the mid 1980s when the civil war in Uganda came to an end (Figure 4.1). Unfortunately, the eruption of a civil war DR Congo in 1996-1998 led to unprecedented poaching in VNP which continues to the present day. As a result, many large mammals moved back to QENP (Plumptre et al. 2007). Elephant group sizes vary with individual populations within a given national park. The mean group size for national parks located with the Albertine Rift, including GVL are 7-23 individuals (Buechner and Dawkins, 1961; Laws 1970) with a mean density of 1.9-3.8 elephants per square kilometer, and a home range of 130-260 km^2 per group. The individual elephant home range is 15-52 km^2 (Laws 1970; Douglas-Hamilton, 1973) during wet season and this could be as large as 1580 km2 during the dry season (Leuthold and Sale, 193).

4.2.3 Age-specific fecundity

The age of sexual maturity in elephants varies in different populations across Africa (Eltringham, 1982) with the potential for a marked effect on a population's growth rate. The age at sexual maturity in elephants is highly variable and is dependent on habitat conditions and elephant population density. Studies of a number of elephant populations in Eastern Africa's savanna parks have shown that the average age at sexual maturity ranges from 12 - 23 years (Laws et al. 1975; Laws 1981, Moss 1988; Nowak, 1999; Moss 2001). This is the widest range of variability in this characteristic that has been reported for wild mammals. The earliest reported age of female sexual maturity for an individual elephant was 7 years in Amboseli National Park (Kenya) (Moss, 2001). Under optimal conditions, females usually attain sexual maturity at 11 years, though the mean age at first birth was reported to be 14 years (Moss 2001). In areas with relatively harsh conditions, females may not become sexually mature until 18 or 19, up to 22 years (Laws 1981; Moss, 1988; Nowak, 1999, Whitehouse & Hall-Martin 2000).

Population modeling suggests that a delay in first parturition of even one year can slow growth rates (Moss 2001). Elephants of each species have a long oestrous cycle (Plotka et al., 1988) but a relatively short receptive period of 2–10 days (Moss, 1983). Synchrony in births within a population is common, although elephants breed throughout the year (Laws et al., 1975; Eltringham, 1979). Thus, elephants either adjust the gestation length or attune receptive periods in order to give birth synchronously (Rasmussen and Schulte 1998). Under favorable environmental conditions, females may benefit by giving birth simultaneously to enhance survivorship and to promote reciprocity in caring for the young. Gestation is a lengthy 22 -27 months followed by a lengthy lactation period of nearly 4-5years to provide extensive care for the offspring. The mean calving interval, a very important parameter that influences the growth rate of elephant population (Hanks & McIntosh, 1973) in Africa was recorded as 2.9-9.1 years (Eltringham 1982) and in Aldo National Park (South Africa), the calving intervalwas reported to be 3.3 years (Gough & Kerley 2006). In high-density populations or nutritionally stressed populations, the average calving interval is relatively long, potentially slowing population growth (Laws et al., 1975; Croze et al. 1981). Females, especially nulliparous adolescents assist in the protection and care of offspring within the herd, termed allomothering (Dublin, 1983; Gadgil and Nair, 1984; Lee, 1987).

In her study, Moss (2001) noted that elephant birth rate in Amboseli National Park (Kenya) varied from year to year, with a pattern of peaks and troughs at 4- to 5-year intervals, while the fecundity and calf survival varied by age of the females, and mortality also fluctuated from year to year. Male offspring remain with their natal group until the age of 12–15 when they join bachelor herds and associate with other family units or travel alone (Lee, 1991). Male elephants are physically capable of copulation and sperm production at 10–13 years of age (Laws, 1969). Age-specific rates of reproduction for individual females have been studied using hazard analysis (Moss 2001) techniques. The peak in fecundity among female elephants has been found to occur at different ages in different populations. For example, this peak occurred in 18 - 19 year old females in the Luangwa Valley, Zambia; 25 - 29 year old females in Addo Elephant National Park, South Africa; and 31 - 35 year old females in Northern Bunyoro, Uganda (Whitehouse & Hall-Martin 2000). Females may average four calves during their lifetime (the range 1 - 9), the number dependent on habitat conditions and elephant density (Laursen &

Bekoff 1978). Fecundity was reported to be relatively constant from the age of 16 to 40 and then declined slightly (Moss 2001), while the fecundity rate of the oldest females (50+ years) averaged 0.098 births per female per year while that of 40- to 50-year-old females averaged 0.196.

The theoretical maximum rate of increase for the African elephant was reported to be 7% (CALEF, 1988), a figure consistent with the annual rate of increase observed at Addo Elephant Park (South Africa) and the highest rate of increase reported for elephants living under natural conditions (Hanks & McIntosh, 1973). In another study, (Dunham, 2012) noted that elephant population in Gonarezhou National Park, Zimbabwe experienced a natural annual growth rate of between four and six percent. According to (CALEF, 1988), an elephant population increasing at the maximum rate soon reaches a stable age distribution in which 48% of the animals are older than 12 years of age, (the age of first reproduction) and 6.7% are calves (less than one year old). Several papers have reported frequencies of calves in natural populations of up to twice this percentage. Recent studies on elephant populations from East Africa and from Zambia have suggested that as population density increases, so does the mean age at puberty and the mean calving interval. At the same time there is also an increase in the proportion of old females that are reproductively inactive.

4.2.4 Habitat-Elephant interactions

Since elephants are such an important component of African ecosystems, it is widely acknowledged that understanding their key demographic parameters is essential for the conservation of the species and the areas in which they live. It is equally important to understand the variables that influence the population structure of elephants in order to design management strategies that strengthen their survival and protection. African elephants have large effects on vegetation and high numbers can lead to extensive habitat modification (Skarpe et al. 2004). Elephants are herbivores, spending 16 hours a day collecting plant food. Their diet is at least 50% grasses, supplemented with leaves, bamboo, twigs, bark, roots, and small amounts of fruits, seeds and flowers. Because elephants only digest 40% of what they eat, they have to make up for their digestive system's lack of efficiency in volume. An adult elephant can consume 140–270 kg (300–600 lb) of food a day. Driven by the need to manage these impacts, several models have been developed to better understand the interaction between elephants and trees (Caughley 1976; Pellew 1983; Dublin et al. 1990; Duffy et al. 2000; Baxter and Getz 2005) but the role of water is either assumed to be obvious or merely ignored.

4.2.5 Density-dependent and density-independent drivers of elephant population dynamics

An understanding of what factors cause elephant populations to increase, decrease or remain stable is fundamental to the design and implementation of management strategies. Long-term ecological studies of population dynamics in large mammalian herbivores provide a detailed understanding of the effects of intrinsic and extrinsic factors in determining population size and composition (Saether 1997; Gaillard, et al. 1998; Gaillard et al. 2000). Most studies have focused on the relationships between population density, weather, and individual survival rates of different sex/age classes (Gaillard et al. 2000; Gordon et al. 2004), and very few studies on density-independent effects (Smith and Anderson 1998; Coulson et al. 2001) upon large mammal populations exist and no specific study has examined the effects of water supply, habitat quality and climate change on age-specific elephant population structure and size in the Greater Virunga Landscape.

It is known that malnourished animals are more likely to succumb to severe climatic events at high than at low population densities (Gaillard et al. 1997; Jacobson et al. 2004). More importantly, it has become evident that the impacts of both density-dependent and density-independent effects vary substantially according to a population's sex/age structure. This is because the survival of different sex/age classes is not equally affected by resource abundance and weather stress. In general, we hypothesize that adult elephants are likely to be relatively impervious to density and climate change effects, while the calves and juveNiles (and possibly senescent individuals) are highly susceptible to both. It is also likely that calves and juveNiles will be more susceptible to naturally-induced mortality while the sub-adults and seniors will experience mortality mainly due to poaching, and killings resulting from human retaliation against crop damage. In order to avoid incursions in human dominated areas, and in response to scarcity of food in open dry savanna habitats, elephants are likely to seek refuge in forested habitats and wetland areas. It therefore seems reasonable to predict that the natality rate of mainly forest and wetland habitat-dependent age groups will vary greatly from the mobile and savanna woodland/grassland-dependent age groups.

There is evidence that ungulate populations subject to heavy harvest experience steep declines following seasonally harsh climatic conditions (i.e. high proportion of juveNiles) and possibly rapid increases following improvements in climatic conditions or a relaxation of harvesting because of the sudden graduation of the juveNiles into sexually reproductive age classes and the very high adult survival probabilities (Gaillard et al. 1997; Festa-Bianchet et al. 2003; Gordon et al. 2004). Recent studies on population dynamics of herbivores have also underlined the importance of time lags in both weather effects and density-dependence because of a combination of delays in the recovery of overgrazed vegetation and the effects of changes in the age structure of the population (Saether 1997). Elephants were listed under

Appendix I of the Convention on the International Trade in Endangered Species of Wild Fauna and Flora (CITES) in 1989 (Barnes et al. 1999) and classified as critically endangered by IUCN in 2002 (IUCN 2002; http://www.redlist.org/).

4.3. Impact of anthropogenic disturbances

Differentiation in age-class-specific mortalities and natality of elephants associated with anthropogenic factors provides additional insight to the mechanisms driving mortality and natality, however, determining the causes of illegal activities is often challenging. The influence of economic factors on elephant mortality may be complicated by variation in environmental factors that affect elephant demography (Wittemyer, 2011). Ungulate survival is strongly mediated by environmental stochasticity or population density (Gaillard, Festa-Bianchet, Yoccoz, Loison, & Toigo, 2000), and younger age classes are more susceptible to resource limitations than adults (Gaillard, Festa-Bianchet, & Yoccoz, 1998).

4.3.1 War impact on elephants

The African continent has struggled with political instability and conflict in recent history. Such instability encourages criminal activity including wildlife trafficking, poaching and other environmental crimes (Hanson et al., 2009); Chase and Beyers et al. 2011; Griffin 2011; Bouché et al. 2012). Even in relatively stable governments, corruption exacerbates environmental degradation (Ouma, 1991) whereby the ruling political party uses it as a strategy to gain political support from the business community and local populace. War and civil strife have been shown to contribute directly to increased wildlife poaching and environmental degradation, especially in developing countries. The three countries namely Democratic Republic of Congo (DRC), Rwanda, and Uganda whose territories constitute GVL have suffered major wars in the last 50 years. A chronology of the wars (Hanson et al., 2009) that these three countries have experienced is provided in Table 4.1 followed by a brief description of how each war affected the economy, people, and the environment, including wildlife management and conservation in the region.

Democratic Republic of Congo experienced several wars immediately after attaining independence (1960-65) affecting almost the entire country resulting in the death of its first Prime Minister Patrice Lumumba. During the period 1966-1974, there was some stability and economic growth, however, due to the resumption of war in 1977, the country suffered a serious economic recession and plundering of natural resources coupled with severe corruption under the reign of General Mobutu Sese seko Wazabanga (Akitoby & Cinyabuguma, 2004; Reno, 2002). Later, Political instability in Democratic Republic of Congo (DRC) intensified during its first (1996–1997) and second (1998–2003) civil wars (Reno, 2002). Since then, North Kivu continues to experience armed conflict resulting in the breakdown of governance structures, including wildlife management. Despite its endowment with vast natural resources and the most extensive network of navigable waterways, DRC's economic activity showed a drastic decline during the period 1960-2000. Before the decline in economic activities, real Gross Domestic Product (GDP) was growing at an average annual rate of 5.1 percent during 1966-74 (Akitoby and Cinyabuguma, 2004). In the wake of the economic recession and debt crisis followed by the invasion of the Shaba Province (the heart of mining activities) in 1977 and 1978, real GDP fell by 12 percent (Akitoby and Cinyabuguma, 2004). The Per capita GDP continued to fall steadily from US$380 in 1960 to US$240 in 1990 and further to US$85 (or 23 cents a day) in 2000 (Akitoby and Cinyabuguma, 2004), placing it among the poorest countries in the world with a dynamic population of 55 million people.

Rwanda's Volcanoes National Park, which borders Uganda and Democratic Republic of Congo, is part of the Greater Virunga Landscape. It is also important to note that the Congo River Basin located in the west, covers 33% of the national territory and the Nile River Basin in the east covers 67% of the country, both rivers receive 10% and 90% of Rwanda's national waters. Rwanda experienced one of the world's most tragic ethnic conflicts, leading to the 1994 genocide coupled with the invasion of the country by armed Rwandan nationals mainly from Uganda. Since then, the government of Rwanda struggled with the task of resettling more than two million people in the aftermath of the war of 1990-1994. Since there was unoccupied land available, deforestation and encroachment of protected areas for settlement put severe pressure on the environment, with large-scale movements of human population and livestock that led to the degradation of fragile ecosystems. It is estimated that nearly 50% of the original forest land has already been lost and protected areas such as Nyungwe and Virunga Volcanoes National Parks suffered major environmental degradation due to agricultural encroachment. It is estimated that over 850,000 refugees settled close to Virunga National Park (DRC) and Virunga Volcanoes (Rwanda) and 40 000 people entered the park each day to harvest forest products and hunt wild animals, including elephant, hippopotamus, and buffalo (Dudley, Ginsberg, Plumptre, Hart, & Campos, 2002); (McNeely, 2003). Another 332,000 refugees encamped in eastern Democratic Republic of Congo near Kahuzi Biega National Park (Prunier, 2004). The fuelwood demand of the refugees camped inside the Park was estimated at 600 metric tons/day leading to widespread depletion of forests in the lowlands.

Uganda has equally suffered major armed conflicts, starting with the military coup of Idi Amin, who later became president serving from February 1971-April 1979 when he was deposed by the Tanzanian army with the help of Ugandan exiles (Tindigarukayo, 1988). During this period, Uganda's economy declined tremendously resulting in smuggling of goods across the neighboring countries by army soldiers and local business people as a means of survival. It is reported that Uganda lost 75% of its elephants, 98% of its rhinos, 90% of its crocodiles, 80% of its lions and leopards, in addition to numerous species of birds (Kpundeh, et al. 1999). In 1981-86, Uganda suffered yet another civil war (Tindigarukayo, 1988) that led to the displacement of nearly 750,000 people and a decline in law enforcement by the forest department that was charged with management of central forest reserves, and Uganda National Parks, responsible for the management of national parks and wildlife reserves and sanctuaries. In 1996-1997, Uganda government battled yet another war orchestrated by the Allied Democratic Forces (ADF) who operated from DRC and attached western Uganda and logged itself in the Rwenzori National Park. This war contributed to the loss of tourism revenue from the Rwenzori National Park, displacement of people from high up to the lower slopes of the mountains resulting in a decline in wildlife in QENP due to bushmeat hunting. In 2006-2007, QENP suffered a major setback when a group of 8000 Basongora pastoralists with 50,000 herds of cattle returned to Uganda, escaping the war in eastern DRC and were temporarily settled in QENP by the government of Uganda (Rugadya 2009). Structurally, Uganda's population is growing at a high rate of 3.2 per cent and is projected to shoot up to 39.3 million in the year 2015 and 54.9 million in 2025 due to high fertility rate (UBOS 2012).

Several studies have attempted to assess the impact of civil strife on the environment, particularly wildlife and wildlife habitats in Africa (de Merode et al., 2007; Draulans & Van Krunkelsven, 2002; Eltringham & Malpas, 1993; Plumptre et al., 1997), Eurasia (Fell, 1997; Fell & Bader, 1997), and the Middle East (Westing, 1984; Westing, 1996) from 1960-1990 adequately reviewed by (Dudley et al., 2002; Hanson et al., 2009). For purposes of emphasis, studies specific to this landscape are reviewed here. In Uganda, (Eltringham & Malpas, 1980) noted that from 1973 to 1976, the elephant populations in Rwenzori and Kabalega (present day Murchison Falls

National Park) had fallen to about 26% and 17% respectively. The decline was attributed to a massive increase in poaching within the parks. Later, another study was conducted during the years 1982 and 1983, a period that lie within civil war of 1979-1987 to examine the conservation status of Uganda's game and forest reserves ((Eltringham & Malpas, 1993). In this study, (Eltringham & Malpas, 1993) reported massive declines in population of elephants and large ungulates, encroachment and degradation of wildlife reserves.

Table 4.1 Chronology of wars in Greater Virunga Landscape (DR Congo, Rwanda, and Uganda) from 1960-2010

War period	Profile of conflicts	Reference	Notes
1960-1965	DRC civil war immediately after independence	(Hanson et al., 2009)	economic collapse, breakdown in governance of protected areas
1971-1972	Military coup by Idi Amin in Uganda	(Gitelson, 1977)	Economic collapse, breakdown in law and governance of protected areas
1978-1979	Uganda - Tanzania war	(Ouma, 1991)	Economic decline, lawlessness, and environmental governance breakdown
1981-1985	Uganda civil war (NRA guerrilla war)	(Prunier, 2004; Tindigarukayo, 1988)	Economic decline, lawlessness, and environmental governance breakdown
1990-1994	Rwanda genocide + Liberation war	(Dudley et al., 2002; Prunier, 2004)	Displacement of people (refugee crisis in Virunga NP), military presence inside protected areas in Virunga and QENP
1996-1997	ADF in Rwenzori region + Tutsi and anti-Mobutu rebellion in eastern DR Congo	(Nackoney et al., 2014; Prunier, 2004)	Rwenzori NP closed for 2 years, reduced law enforcement and displacement of people
1997-2003	Great Lakes Region conflict (Africa's first world war)	(McNeely, 2003; Prunier, 2004)	Plunder of DRC resources, military influx inside protected areas, displacement of people (1 million)
2004-2014	Present day Kivu conflict	(de Merode et al., 2007)	Displacement of people (refugee crisis), military presence inside protected areas in Virunga and QENP (Basongora and 20,000 heads of cattle in 2007)

Similarly, a study of the impact of war on wildlife in Rwanda's *Parc National des Volcans* during the period 1990-1997 by (Plumptre et al., 1997) reported increased rates of mammal poaching in the western sector, and lower rates in the eastern sector of the park. A related study conducted by (Hall et al., 1998) in Kahuzi Biega National Park (DRC) during the same period reported similar results whereby elephant poaching and habitat encroachment increased and loss of forest areas to burning in an attempt to displace the rebel forces. A more recent study by (Nackoney et al., 2014) who used both satellite imagery and ground-based observations to analyze and compare forest loss and forest degradation during the civil war period (1990-2010) in Central Africa, noted that forest loss and degradation more than double, and wildlife suffered severe impacts from increased human settlement deep into the forest interior and reliance on bushmeat for protein intensified. In their essay, (Dudley et al., 2002) noted that intensive subsistence or commercial harvesting of wildlife and vegetation by refugees, combatants, and local residents during periods of civil strife in Africa exacerbates the effects of preexisting scarcities of natural resources on local communities and increases vulnerability of tropical forest ecosystems to systematic degradation. In support of these findings, (McNeely, 2003) suggested that the constant threat of war erodes the little, but highly valuable incentives by the communities and wildlife management agencies to investment in long-term management planning and conservation of wildlife and wildlife habitats.

4.3.2 Poaching

At a continental level, central Africa has shown worrying poaching trends for some time and has consistently displayed the highest levels of poaching in any sub-region since Monitoring of Illegal Killing of Elephants (MIKE) monitoring began (CITES, 2014). In 2006, Proportion of Illegally Killed Elephants (PIKE) levels were at 0.5, which means that about half the elephant carcasses encountered on patrol in MIKE sites were illegally killed (CITES 2014). In 2011, however, PIKE levels had risen to 0.9. The small and fragmented elephant populations in West Africa are particularly vulnerable to increases in poaching, which can severely distort sex ratios and lead to local extinctions. Historically, elephant populations of less than 200 are known to die out within a matter of a few decades (Bouché et al., 2011). This has happened in several elephant populations in West Africa, but a recent example is Comoé National Park in Côte d'Ivoire, where poaching associated with the country's recent civil war has reduced elephant populations to near extinction (Fischer 2005; CITES 2012a). Eastern Africa has experienced a three-fold increase in reported illegally killed elephants in MIKE sites from a PIKE level of about 0.2 in 2006 to almost 0.6 in 2011. In Tanzania, PIKE levels were higher than 0.7 across the country's five MIKE sites. Many of these reports on illegal killings came from the Selous Game Reserve in southern Tanzania, which is recognized as the largest game reserve in the world and also an UNESCO World Heritage Site (Baldus 2009). In 2011, more than 65 per cent of the 224 carcasses encountered on patrols had been killed by poachers (CITES 2012a). At the national park level, PIKE levels were higher than 0.9 in Ruaha Rungwa National Park (Tanzania) in the same year. Kenya showed similar poaching levels in 2011, with two thirds of the 464 carcasses reported in MIKE sites identified as illegally killed, particularly in the Tsavo National Park and the Samburu Laikipia ecosystem (CITES 2012a). Uganda harbors a much smaller elephant population and has not reported as many carcasses as its neighboring countries. Still, the Murchison Falls National Park and the Queen Elizabeth National Park reported PIKE levels of 0.8 and 0.9 respectively in 2011.

Overall there has been a marked decrease in numbers of large mammals in the savannas of the Greater Virunga Landscape. Elephant population declined from 5400 in 1960s (Malpas 1980; Mertens 1983; Cornet d'Elzius 1996) to 2100 counted in 2000 (Lamprey et al. 2003; Blanc et al. 2003). This is a direct result of civil wars in both Uganda and Democratic Republic of Congo (DRC), which led to rampant poaching. First, the effects of the civil wars affected each country at different times, that is, Uganda in the late 1970s and early 1980s, and then DRC in the 1990s and 2000s, leading to declines in mammal populations during those periods. Elephant numbers in Queen Elizabeth National Park declined to 150 individuals in 1981 but by 2002 there were about

1,000 individuals in the same park and recent counts in 2006 show an astronomical increase to 3,307 elephants (Plumptre et al.2010). In Virunga National park (DRC), elephant population declined from 3,000 in 1960 to at least 486 in 1998 (Barnes et al. 1999). Elephant populations, however, have been recorded to increase at a maximum of about 6% per year (Calef, 1988; Owen-Smith 1988).

Second, poaching exploded because of the lucrative markets for ivory in the middle and far east Asia that can be traced as far back as 500 A.D. (Simons 1962; Parker 1979; Blanc et al. 2003). Severe episodes of poaching in many areas started from outside protected areas but eventually penetrated into major national parks and reserves such as Queen Elizabeth National Park, and Murchison Falls National Park in Uganda. This was in response to the increased market which extended to Europe and America in the mid of the 19th century. Colonial authorities in most of Eastern Africa, however, responded quickly by prohibiting unregulated commercial hunting and instead licensed sport hunting (Simons 1962) at the beginning of the 20th century. The volume of international trade in mammoth ivory rose steadily between the late 1990s and 2007 (CITES 2014). It then dropped sharply in 2008 and 2009, but quickly recovered and continued to increase in subsequent years. Overall, the total volume in trade went from 17.3 tons in 1997 to 95 tons in 2012, a more than five-fold increase (CITES 2014). The total value of mammoth ivory in trade increased less steeply between the late 1990s and 2006, but also experienced a sharp increase in 2007 followed by decline and recovery in subsequent years. Import price per kg, calculated on the basis of declared value and weight at import, remained relatively stable until 2006 and then followed a similar pattern, increasing more than 2.4-fold from $36.7 per kg in 2006 to $125.89 per kg in 2012 (CITES 2014).

It is important to note that during the period 2011-2012, legal trade in Loxodonta Africana was conducted directly from African range States. During this period, direct export of 977 tusks and 16,660 kg of tusks from wild sources was made. In principal, the legal sale comprised wild-sourced hunting trophies (including tusks). According to CITES (2013), analyses of the Elephant Trade Information System (ETIS) data revealed that the years 2011, 2012 and 2013 represent the highest quantity of ivory ever seized and reported to ETIS over the last 25 years (Milliken et al.2012; Underwood et al. 2013; CITES 2014). Underwood et al. (2013) noted that over 70% of the Weights Index is from shipments of raw ivory weighing at least 100 kg mainly moving from Central and East Africa to Southeast and East Asia. 2011. Two years after CITES placed a 9-year moratorium on future ivory sales (CITES 2007), elephant poaching is on the rise. CITES restricted the moratorium to the four countries involved in the initial sale (CITES 2007), but never established whether poaching levels were so serious that any further trade could ultimately jeopardize elephant survival throughout most of Africa (Wasser et al., 2010).

4.3.4 Poverty

Declines in economic activity and associated changes in human livelihood strategies are highly associated with an increase species overexploitation (Wittemyer, 2011). Elephant population decline has been attributed to economic crises, which often drive intensification of subsistence poaching and greater reliance on natural resources. Similarly, increased human population and resettlement close to protected areas and inside conservation areas (Brooks and Buss 1962; Rogers and Lobo 1980) has been associated with an increase in hunting for bushmeat (Olupot, McNeilage & Plumptre, 2009) and poaching of elephants (Hanson et al., 2009; Prunier, 2004; Wittemyer, 2011). Olupot et al. (2009) who analyzed the socioeconomics of bushmeat hunting in Uganda’s hunting hot spots noted that except for households headed by hunters, bushmeat was less important as a source of protein than domestic livestock and fish for the households in the study sites. Hunters, however, heavily depended on bushmeat as a source of both income and food. Around Queen Elizabeth Conservation Area, which includes Rwenzori National Park, annual poacher incomes averaged US$219.29 and bushmeat contributed 21% (range = 0 to 45%) to this income (Olupot et al.2009). More important, poverty and cultural attachment were cited as the main reasons for bushmeat exploitation (Olupot et al. 2009). Overall, the hunters household incomes were too low (US$219.29) compared to the average household (US$1447.27) in the same village (Olupot et. al. 2009). Poverty around protected area adjacent communities is attributed to economic losses associated with wildlife conservation. Studies that have examined the contribution of forests to rural household incomes have shown that protected area adjacent communities derive 18-36% of the household income from utilizing protected area resources (Bush et al. 2004; Tumusime, 2005; Jagger, 2007) and Non Timber Forest Products in particular contribute 52% of the total forest income (US$194.11) earned (Bush et al.2004).

Rural developers, political ecologists, and social scientists have argued that protected areas [conservation areas] economically impoverish the already poor and vulnerable frontline communities because of the enormous costs (Naughton-Treves 1998; Osborne and Parker 2003) they are forced to bear in the interest of conservation. For example, Mackenzie and Ahabyona (2012) who calculated the economic loss to crop raiding borne by an average farmer living adjacent to Kibale National Park (Uganda) was US$888 per annum excluding other risks such as food insecurity (Hill 1997), physical injury (Sifuna 2005), and wildlife-human transmitted diseases. It is not surprising that the advocates of legal trade and one-off sale of ivory by some African countries argued strongly in favor of lifting the ban to allow them generate revenue to finance conservation (Bulte et al. 2007) and support livelihood programs for the protected area adjacent communities as compensation strategy. Contrary to this argument, the African Elephant Specialist Group provided evidence of elephant population counts suggesting that the ban on ivory trade had prevented the freefall of elephant population (Barnes et al. 1999; Stiles 2004; Blanc et al.2005). In general, there is now consensus that conservation needs to be looked at from a broader perspective including poverty reduction (Naughton et al. 2005), education and awareness, health, and equality.

4.3.5 Human population trends for DRC, Uganda and Rwanda

In all the three countries, the human population has been increasing over the past forty years. During this period (1960-2012), the mean annual population growth rate for Rwanda, DRC, and Uganda was 2.62, 2.80, and 3.22 respectively. According to the World Bank country-level socioeconomic indicator records (World Bank, 2013), the mean national population density for Uganda is 182 persons square kilometer, DRC (29), and Rwanda (464). The population in these three countries has been dramatically increasing over the last 50 years (Figure 4.2). As a result, human populations have increased around protected area, and the human-wildlife conflicts are more accentuated. Furthermore, the buffer zones are completed destroyed and wildlife corridors are either settled in or dysfunctional (Hanson et al., 2009).

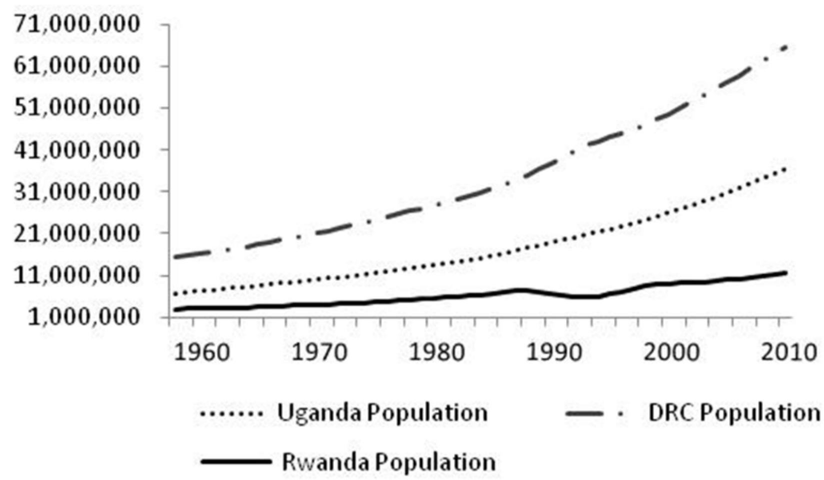


Figure 4.2 Human population trends for GVL overlapping countries (Source: World Bank 2013)

According to calculated gross national income (GNI) per capita, a criterion that World Bank uses to classify economies, DRC with GNI of US$190, Rwanda (US$570), and Uganda (US$510) follow in the category of low-income economies (World Bank, 2013). GNI per capita is based on purchasing power parity (PPP). PPP GNI is gross national income (GNI) converted to international dollars using purchasing power parity rates. An international dollar has the same purchasing power over GNI as a United States dollar has in the United States. GNI is the sum of value added by all resident producers plus any product taxes (less subsidies not included in the valuation of output) plus net receipts of primary income (compensation of employees and property income) from abroad. World Bank uses the Atlas method, which involves a conversion factor (an average of the country's one-year exchange rate) that is used to convert the economic values to United States dollars (US$) in order to reduce the impact of exchange rate fluctuations in the cross-country comparison of national incomes. Low-income economies are those countries with a GNI per capita of $1,025 or less in 2011, middle-income economies (GNI per capita of more than $1,025 but less than $12,475), and lower middle-income and upper middle income economies are separated at a GNI per capita of $4,036 whereas the high-income economies are those with a GNI per capita of $12,476 or more. Similarly, when the purchasing power parity (PPP) per capita (US$) criterion is considered, these countries PPP per capita values are US$340 (DRC), US$1270 (Rwanda), and Uganda (US$1310), which are also considered low (World Bank 2013). The poverty headcount ratio at rural poverty level calculated as percentage of the rural population for Uganda in 2006 was (34.2%) higher than the national average poverty line (31.1%) while that of DRC (75.7 cf. 71.3%), and Rwanda (61.9% cf.56.7%). These economic performance criteria clearly indicate that poverty in these countries is still high.

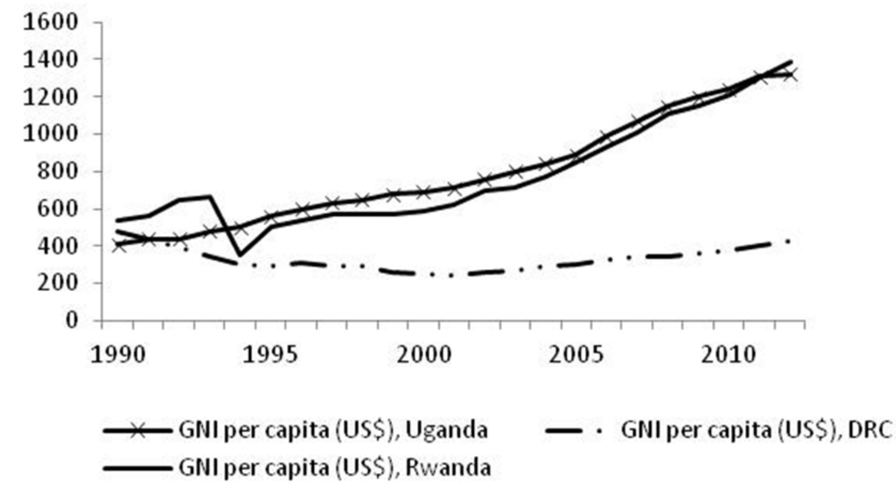


Figure 4.3 Gross National Income (GNI) per capita for DRC, Rwanda, and Uganda (current international dollars are based on the year 2011; source: World Bank, 2013)

4.3.6 Climate change and variability

The Greater Virunga Landscape experiences a varied climate, largely as a result of the influence of the high mountains. In the highlands, the climate is temperate, although at lower altitudes it is hot and humid. Rainfall distribution is bimodal, that is, March to May, and mid-

September to mid-December alternated with drier spells each year. Annual rainfall is highly varied at a local scale from 500 millimeters (mm) at Lake Edward, 900-1,500 mm on the plains south of the Lake Edward, decreasing highly on the volcanoes but on the west slope of the Ruwenzori orographic precipitation is almost 3,000 mm. Precipitation in the Rwenzori Mountains is bimodal; wetter periods occur from March to May and August to November. Apart from the seasonal control on precipitation exerted by movement of the ITCZ, there is a strong orographic effect on local precipitation. Mean annual precipitation from 1964 to 1995 recorded at Kilembe at an elevation of 1370 m above mean sea level (mamsl) is 1540 mm/year whereas this flux drops to 890 mm/year just 11 km away but 410 m lower in elevation at Kasese Airport (960 mamsl) (Taylor et al. 2009). These mountains have heavier snowfall than Mounts Kenya or Kilimanjaro, are permanently ice and snow-covered and carry small retreating glaciers. The high altitudinal range (700 – 5100 m) results in marked climatic variations with a consequent diversity of habitats. The mean annual temperature also varies from year to year between 20oC and 23oC with a 15^{0}C diurnal range.

Climatic variations, particularly in precipitation, affect the production of plant material and hence, indirectly, the carrying capacity of the ecosystem. Recent research also points to climate change and the increasing frequency of droughts as a major threat to elephant populations in the Sudano-Sahelian region (Bouché, 2012). Foley et al. (2008) explored how gender, age, mother's experience and family group characteristics determined calf survival in an elephant population during a severe drought in Tanzania in 1993. They noted that young males were particularly sensitive to the drought and calf loss was higher among young mothers than among more experienced mothers. They also noted high variability in calf mortality between different family groups, with family groups that remained in the National Park suffering heavy calf loss, compared with the ones that left the Park (Foley et al.2008). It is plausible to conclude that severe droughts can dramatically affect early survival of large herbivores and extreme climatic events are likely to increase selection of individuals that are demonstrate poor or slow behavioral change.

Numerical models (General Circulation Models or GCMs) representing physical processes in the atmosphere, ocean, cryosphere and land surface are the most advanced tools currently available for simulating the response of the global climate system to increasing greenhouse gas concentrations (IPCC 2001, 2007). While simpler models have also been used to provide globally- or regionally-averaged estimates of the climate response, only GCMs, possibly in conjunction with nested regional models, have the potential to provide geographically and physically consistent estimates of regional climate change which are required in impact analysis. A new set of scenarios referred to as Representative Concentration Pathways (RCP) were used established for use in the new IPCC climate model simulations conducted under the framework of the Coupled Model Intercomparison Project (CMIP5) of the world Climate Research Program (Taylor et al., 2012). The IPCC's Assessment Report Five (AR5) present four different sets of RCPs based on radiactive forcing scenarios namely RCP2.6, RCP4.5, RCP6.0, and RCP8.5 (van Vuuren et al. 2011). It should be noted that the baseline period for the IPCC AR5 is 1986-2005. In this study, RCP2.6, RCP6.0, and RCP8.5 were used in the implementation of climate change impacts on elephant population dynamics. Under RCP2.6, a more conservative pathway, the peak in radiative forcing is at approximately three watts per meter squared (~3 W/m^2), (~490 ppm CO2 eq) before 2100 and then declines to 2.6 W/m^2 by 2100 (Clark et al. 2007; Wise et al. 2009; van Vuuren et al. 2011). In the case of RCP6.0, concentrations stabilize without overshoot pathway to 6 W/m^2 (~850 ppm CO2 eq) at stabilization after 2100 (Hijioka et al. 2008) while RCP8.5 is considered to be the rising radiative forcing pathway leading to 8.5 W/m^2 (~1370 ppm CO2 eq) by 2100 (Riahi et al. 2011; van Vuuren et al. 2011). According to IPCC (2013), global surface temperature change for the end of the 21st century is likely to exceed 1.5 ^{o}C relative to 1850 to 1900 for all RCP scenarios except RCP6.0. The application of the RCP scenario results are discussed in detail under the methodology section.

In the study region, the temperature change under RCP8.5 is predicted to be 4-5^{o}C (IPCC 2013). While interpreting the results from the GCM, we need to be mindful that GCMs depict the climate using a three dimensional grid over the globe, typically having a horizontal resolution of between 250 and 600 km, 10 to 20 vertical layers in the atmosphere and sometimes as many as 30 layers in the oceans. Their resolution is thus quite coarse relative to the scale of exposure units in most impact assessments. Moreover, many physical processes such as those related to clouds, also occur at smaller scales and cannot be properly modeled. Instead, their known properties must be averaged over the larger scale by parameterization. This is one source of uncertainty in GCM-based simulations of future climate. The other concerns relate to the simulation of various feedback mechanisms in models concerning, for example, water vapour and warming, clouds and radiation, ocean circulation, ice and snow albedo. For this reason, GCMs may simulate quite different responses to the same forcing simply because of the way certain processes and feedbacks are modeled.

4.3.7 Water resources in the Greater Virunga Landscape

The major source of water supply in the region is through precipitation followed by ground water and to a small extent glacial melt. Hemp (2009) noted that the forests above 1300 meters receive nearly 1600 m^3 of water per year, of which, precipitation contributes 95 percent and approximately 5 percent from fog interception. Greater Virunga Landscape is 65percent woody, which makes these figures reasonable for modeling consideration. In comparison, an average annual water output of about 1 million cubic meters (5%) of what is recovered from fog water deposition comes from 2.6 km^2 of glaciers (Hemp 2009). Water in the reservoirs is sustained through direct precipitation, stream flows as direct rain, subsurface, and overland flow. It is however, lost through evapotranspiration from water bodies, and terrestrial and aquatic plants. Water availability is reduced through direct plant, animal, and human withdrawal to meet domestic and industrial use. The loss of forests in this landscape would have a greater effect on the water balance than the disappearance of the glaciers of which the hydrological effect would be negligible. Most of the headwaters are located in the Rwenzori Mountain and several rivers and streams drain into Lakes George and Edward. The presence of icefields on top of the Rwenzori Mountain plays a critical role in catchment recharge and climatic modulation. The Rwenzori mountain range covers a total area of 3000 km^2 and lie north of the equator on the borders of Uganda and Democratic Republic of Congo. Unlike the other snow peaks of East Africa, the range is not of volcanic origin but is a gigantic uplifted horst-like block of pre-Cambian rocks, and its formation is connected with the complex tectonics of the rift valley (Whittow, 1966).

4.3.7.1 Water from Rwenzori mountain glacier fields

The retreat of glacier ice on East Africa's mountains since the little ice age is believed to be partly due to global climatic trends (Kaser et al. 2004). A reduction in air humidity with all the consequent changes in energy and mass balance is suggested to be a major reason for the general recession of tropical glaciers since the end of little ice age and the rise in air temperature partially explained the glacier recession (Kaser 1999). The accelerated recession since the 1980s, however, is probably caused by increased air temperature and increased air humidity (Barnett et al. 2005). Glacier fields on mountain Rwenzori contribute to the river discharges through glacial melt. Of the 37 glaciers on the Rwenzori (syn. Ruwenzori), Stanley peak is the highest at 5100 m a.s.l., while Speke glaciers are the most studied (Temple, 1968). During the period 1958-1967, the mean rate of recession of approximately 1.3 m per year was recorded for the lowest section of the Speke glaciers (Whittow et al. 1968; Temple 1968). Field research carried out in the 1950s (Whittow et al. 1963), early 1990s (Kaser and Noggler 1996) and from 2003 to 2005 (Taylor et al., 2006) indicate that the area covered by alpine glaciers has reduced from 7.5 km^2 in 1906 to <1 km2 in 2003. More recent studies (Taylor et al. 2009) showed that the rate of decline in glacial extent since 1990 is consistent with the overall trend of approximately 0.7 km^2 per decade since 1906. From moraine evidence, Osmaston (2006) estimates that glaciers covered approximately260 km^2 between 10 and 20 ka before present (BP) (Last Glacial Maximum) and approximately 10 km^2 between 100 and 200 a BP (Little Ice Age). Glacial recession since the 19th century has led to increased seasonality and overall reductions in river flow (Mark and Seltzer, 2003; Bradley et al., 2006). Excessive melting was reported to occur if continuous sunshine is experienced, which observations are consistent with those of Howell (1953) who studied the glaciers in the tropical Andes of Peru, and Platt (1966)'s work on Mount Kenya. Temple (1968) also noted that large variations in melt-water amounts were experienced seasonally over periods of a few days and between night and day, and occasionally maintained throughout the night in the ablation seasons. Flint (1959) argued that the cause of glacier retreat was a secular rise in temperature rather than any change in precipitation amounts. There is also evidence to suggest that the decrease in glacial retreat is related to high precipitation amounts associated with greater cloud cover and less insulation and melting, which is also corroborated by Water Development department's empirical measurements (daily stage height, and monthly river discharge measurements) data collected on river Mobuku and Bukuju draining from Rwenzori from an altitude of 3400m.

Recent studies by Taylor et al. (2009), however, revealed that observed acceleration in the rates of termini retreat of the Speke and Elena glaciers since the late 1960s is attributed partly to the convex–concave slope profile in which these valley glaciers reside. They argued that current glacial recession has a negligible impact on alpine river flow. Also, spot measurements of melt water discharges indicate that ice fields contribute 0.5% of the river discharge at the base of the Rwenzori Mountains during both dry and wet seasons (Taylor et al.2009). Taylor et al. (2009) also noted that an anomalously high discharge of the River Mubuku (1730 mm per year) arises from high rates of precipitation exceeding 2000 mm per annum below alpine icefields within Heath-moss and Montane forest ecotones that occupy more than half of the river's gauged catchment area. For other tropical alpine icefields representing a tiny fraction (<1%) of alpine river catchment areas (e.g. Irian Jaya, Kilimanjaro, Mount Kenya), glacial melt water discharges are similarly expected to contribute a negligible proportion of alpine river flow.

4.3.7.2 Importance of water to elephants

Carrying capacity or the ability of a given area to support a certain population of animals on a continuing basis may be altered by both long and short term variations in climate and particularly in precipitation (Phillipson, 1975). It has been documented that standing crop biomass, energy expenditure and production by large mammalian herbivores in the African savannas show a high degree of correlation with mean annual precipitation and predicted above ground primary production (Coe et al. 1976). In terms of reproduction and body physiology, there exists a direct relationship to precipitation cycles contributing to lower stress due to food abundance (Laws 1968; Field 1971; Kinahan et al. 2007; Eltringham 2008). On average, an adult elephant consumes between 200-250 litres of water a day (Buss 1955; Barnes et al. 1999). Kinahan et al. (2007) who studied the body temperature daily rhythm adaptation in African savanna elephants, observed that elephants had lower mean Tb values (36.2 ± 0.49 °C) than smaller ungulates inhabiting similar environments but did not have larger or smaller amplitudes of Tb variation (0.40 ± 0.12 °C), as would be predicted by their exposure to large fluctuations in ambient temperature or their large size. According to (De Beer & Van Aarde, 2008) elephants seem to locate their home ranges in areas of the landscape where higher levels of resource heterogeneity occur during wet and dry seasons, which explains their spatial preferences and a decrease in home range size with availability and distribution of water sources. In Hwange National Park (Zimbabwe), elephant numbers at waterholes increased during dry years, a phenomenon that (Chamaille-Jammes, Valeix, & Fritz, 2007) attributed to reduced surface-water availability. This led them to conclude that surface-water availability drives the distribution and abundance of elephants within Hwange National Park, and therefore appear to be at the heart of the trade-off between elephant conservation and the extent of their impact on ecosystems (Chamaille-Jammes et al., 2007).

Hot, arid environments may impose restrictions on food and water availability as well as exposing animals to large fluctuations in ambient temperature. Therefore, large homeothermic animals that inhabit these habitats are expected to have adaptations that deal with these environmental stresses. For example, elephants have developed physiological adaptations such as lowering of the average (mesor) body temperature and regular adjustments to the body temperature daily rhythm, enable an animal to reduce energy consumption and water loss (Dawson, 1955; (Kinahan, Inge-Moller, Bateman, Kotze, & Scantlebury, 2007). According to Buss (1961), during the dry season, elephants concentrated near permanent water supplies where some plants were utilized intensively and used woodlands primary for day time concealment and shade, but not as a principal source of food. With the onset of the rainy season, elephants returned to savanna lands where water was then available and extensive grasslands provided an abundance of their favored foods (Buss 1961; Field 1971). How important the stream distribution, and water supply, including water glacial stock and recession on mountain Rwenzori to the elephant population dynamics is not well known. It is, however, a known fact that elephants depend heavily on water for cooling the body and direct consumption, and stream channels provide migration routes for elephants to other habitats or landscapes in the region.

4.3.8 Habitat loss

Empirical studies to date suggest that habitat loss, fragmentation and degradation have huge negative effects on biodiversity (Fahrig, 2003). Habitat degradation and degradation is big challenge to the wildlife managers in Africa because of the rapid change in demographics and socioeconomics of the continent. In the early 1980s, surrounding habitat in a radius of 50-km from the protected area boundaries enhanced the protected areas' effectiveness to protect wildlife and these buffer zones afforded them the capacity to promote species richness (DeFries Ruth, Hansen Andrew, Newton,Adrian C. & Hansen Mathew C., 2005). Seventy percent of the surrounding buffer areas coupled with the core forest habitats have declined in the last 20 years while 25% of the protected areas experienced declines within their administrative boundaries (DeFries Ruth, Hansen Andrew, Newton, Adrian C. & Hansen Mathew C., 2005). (Achard et al., 2002) who used Earth-observing satellites to determine deforestation rates of the world's humid tropical forests between 1990 and 1997 showed that Africa experienced an annual deforestation rate of 0.43% and a forest degradation rate of 0.21%. In a related study, (Hansen & DeFries S., 2004) described the changes in forest cover from 1982 to 1999 based on an 8-km pathfinder Advanced High

Resolution Radiometer (AVHRR) noted that the percentage tree cover had decreased during the period 1984-1997 and Africa experienced an annual deforestation rate of 0.09 percent. Habitat corridors are being emphasized in wildlife management as one way of maintaining spatially dependent ecological processes within landscapes where habitat has been seriously depleted. Corridors, however, can only be effective if they significantly contribute to the species sustaining processes of gene flow, resource access or the colonization of vacant patches (Drielsma, Manion, & Ferrier, 2007).

A number of studies have been conducted that attempted to relate habitat quality with elephant reproduction and mortality rates (Pettorelli et al., 2011; Rasmussen et al., 2006; Wittemyer, Barner Rasmussen, & Douglas-Hamilton, 2007). For example, (Rasmussen et al., 2006) compared the explanatory power of rainfall, a commonly used proxy for variability in ecological conditions with normalized differential vegetation index (NDVI) to predict time-specific conception rates of an elephant population in northern Kenya and noted that season-specific conception rates were correlated with both quality measures. (Trimble, Ferreira, & Van Aarde, 2009) evaluated the relative importance of conception, gestation, first year survival and subsequent survivorship for controlling demographic variation by exploring the relationship between past environmental conditions determined by integrated normalized difference vegetation index (INDVI) and the shape of age distributions at 17 sites across Africa. The results challenge Eberhardt's paradigm for population analysis, which suggested that populations respond to limited resource availability through a sequential decrease in juveNile survival, reproductive rate and adult survival. Contrastingly, elephants appear to respond first through a reduction in reproductive rate, a discrepancy that is probably attributed to evolutionary significance of extremely large body size in order to increase their survival rates, but decrease reproductive potential.

4.3.9 Conservation

Conservation management options for African elephants range from local to regional scales. The idea of setting up state managed wilderness areas or protected areas envisioned by John Muir in the 1860-70s became a very popular strategy for conserving wildlife and wildlife habitats (Cronon, 1996; Landres, Morgan, & Swanson, 1999; Mittermeier et al., 2003). It is estimated that protected areas covering 10–12% of the earth’s surface were established to address the global decline in biodiversity (Chape, Harrison, Spalding, & Lysenko, 2005; Plumptre et al., 2014). Importantly, protected areas were expected to provide the core of efforts to protect the world's threatened species and are increasingly recognized as essential providers of ecosystem services and biological resources. Protected areas, however, have been proven to be ineffective toward delivering conservation benefits, particularly if the sizes are too small and disconnected (Brooks et al., 2004), governance and ownership rights are poorly defined and contested (Borrini-Feyerabend, 2003; Lockwood, 2010), and law enforcement is weak or lacking (Plumptre et al., 2014). Furthermore, their role has expanded to include the provision of socioeconomic benefits (McNeely, 1994; (West, Igoe, & Brockington, 2006), a role that was never envisaged during the conceptualization of the parks idea.

Earlier approaches to conservation of elephants focused on managing elephant numbers (Beuchner & Dawkins H.C., 1961; Buss, 1961). Elephant management aimed to stabilize numbers at levels with the assumption that vegetation would not be degraded, thereby maintaining biodiversity (Caughley & Krebs, 1983; Gillson & Lindsay, 2003; Van

Aarde & Jackson, 2007). The concept of economic carrying capacity hugely influenced this practice (Caughley, 1994; Gillson & Lindsay, 2003) and consequently agricultural rather than ecological paradigms drove many early management decisions and choices. Wildlife managers, however, have become more aware that large herbivores have wide home ranges than the protected areas alone can provide. In the arid environments of Mali and Namibia, elephants were reported to travel enormous distances to meet their nutritional requirements, with recorded home ranges of 24, 000 km^2 and 12, 800 km^2 respectively (Blake et al., 2003; Leggett, 2006). At the park level, elephants moved 100 kilometers outside Tarangire National Park (Tanzania) boundary in search of food and water, and their home range varied from 477 km^2 to 5060 km^2 (Galanti et al. 2006). For example, active management by the addition of water and manipulation of fire may have instigated changes in the ranging behavior of herbivores and induced high local impacts (Owen-Smith, 1996). The ecological basis for the development of elephant management options demands integrating the socioeconomic and political realities of African countries and the transboundary nature of some wildlife. Landscape-scale planning approaches (Didier et al., 2011; Margules & Pressey, 2000) are now being tried to secure the elephant ranges, movement corridors (Ryan & Hartter, 2012), and to determine the spatial scope of activities that are needed both to meet the needs of species and to maintain ecological processes and ecosystem services important for people . Natural resource managers are developing several transboundary conservation programs (Plumptre, Kujirakwinja, Treves, Owiunji, & Rainer, 2007) and peace parks across the region.

The African elephant range states developed an African Elephant Action Plan whose main goal is to secure and restore where possible sustainable elephant populations throughout their present and potential range in Africa recognizing their potential to provide ecological, socio, cultural and economic benefits (CITES, 2010). The specific objectives of the plan are 1) to develop and prioritize mechanisms by which all elephant populations in Africa would be offered the recognition, protection and support needed to ensure their future survival, and 2) work together to reduce the threats facing elephants namely illegal international and domestic trade in ivory, human elephant conflict, habitat loss and fragmentation, illegal killing for ivory and meat, lack of institutional and enforcement capacity and local overabundance. The listing of African elephants on Appendix I of CITES in 1989 succeeded in reducing the scale of killing, however, it is clear that poaching and illegal trade continues to pose a very serious threat to many African elephant populations. In addition, not all African elephant range states have developed country-level elephant action plans. As such, illegal killing of elephants continues to occur unabated.

The impact, both positive and negative, of elephants on other biodiversity continues to be a topic of research and concern (Gandiwa et al., 2011; Kohi et al., 2011; Eppset al, 2011; Odadi et al., 2011). Current approaches do recognize that humans are part of the protected area system and do face major political, socioeconomic and ecological costs (Brandon & Wells, 1992; Cernea & Schmidt-Soltau, 2006; Naughton-Treves, Holland, & Brandon, 2005). Human-elephant conflict in particular, continues to pose a serious challenge across much of the range. As result of these economic consequences, wildlife managers and conservation organizations have been forced to experiment with new conservation strategies such as community conservation where communities are expected to participate in conservation, and integrated conservation development (ICD) program that includes income generating activities such as ecotourism and sport hunting alongside the core traditional conservation activities (Wells & McShane, 2004).

Some of these initiatives have already received very strong criticism such as community conservation (Berkes, 2004), collaborative natural resource management (Agrawal & Ostrom, 2001; Blaikie, 2006; Leach, Mearns, & Scoones, 1999; Shackleton & Shackleton, 2006) for falling short of expectations.

Sport hunting developed rapidly becoming one of the major sources of foreign income in developing countries targeting particularly elephants, known to be of high economic value (Lindsey, Balme, Booth, & Midlane, 2012). Sport hunting has made elephants an important source of revenue through ivory trade and trophy hunting (Leader-William et al. 2001) and ecotourism (Ogutu 2002). For example, a male elephant is worth a trophy value of US$10,000 (Murphree 2001), and tourists from America and Europe are willing to pay US$14,000–60,000 for a 10–21-day safari to hunt elephant, buffalo, lions, eland and other trophy species (Wilkie and Carpenter 1999). Trophy hunting has been estimated to generate gross revenue of at least US$201 million per year in sub-Saharan Africa from a minimum of 18,500 clients (P. Lindsey et al., 2007). Other new innovative ways of financing the management and conservation of wildlife include Payment for Ecosystem Services (PES) (Nelson et al., 2009; Nelson et al., 2010; Turpie, Marais, & Blignaut, 2008; Wendland et al., 2010; Wunder, Engel, & Pagiola, 2008), which involves compensating land owners to maintain their property under forestry, wetland, and wildlife conservation. Under the PES schemes, Reduced Emissions from Deforestation and Forest Degradation Plus (REDD+) (Burgess et al., 2010; Phelps, Webb, & Koh, 2011) supported by United Nations and the World Bank is being pursued by most countries in developing countries such as Madagascar, Tanzania, DR Congo, and Uganda. Although a number of innovative methods have emerged to help mitigate conflicts (Graham et al., 2011; King, 2011), long-term land use planning and cooperative management of elephant populations with local communities are required to provide sustainable solutions. Studies of elephant movement patterns are ongoing in many African elephant range sites (Boettiger et al., 2011; Duffy et al., 2010; von Gerhardt-Weber, 2011) and these could provide useful information for land-use planning.

4.4 Methodology

This section provides a description of the study area, including the major national parks and their global importance, detailed account of the sources of data, the conceptual and empirical models for the study. It also attempts to explain the importance of each of the state and auxiliary variables used to implement the model.

4.4.1 Study area

The Greater Virunga Landscape (GVL) straddles the borders of Uganda, Rwanda, and the Democratic Republic of Congo (DRC). It is one of the six key landscapes in the Albertine Rift and is among the most species rich of any landscape in the world (Plumptre et al. 2007). The Greater Virunga Landscape is an interconnected set of protected areas (i.e. seven national parks, three large tropic high forest reserves, and three wildlife reserves) that straddle the borders of Uganda, Rwanda and Democratic Republic of Congo (DRC) (Figure 3.1) covering an area of 15,700 km^2, of which 13,200 km^2 (88%) is protected. Bwindi Impenetrable National Park is sometimes considered together with this landscape but its connection to it was severed about 50 years ago. This landscape includes the Virunga Volcanoes, famous for their population of mountain gorillas (Gorilla beringei beringei), the savanna parks of Virunga and Queen Elizabeth, the Kibale National Park famous for its diversity and biomass of primates (Oates et al., 1990; Struhsaker, 1997) and the Rwenzori massif also known as the ‘Mountains of the Moon’. Altitude ranges from 5109m at the top of the Rwenzori massif to 600m in the Semliki Park and consequently the landscape supports a wide variety of habitats. These habitats include, alpine moorland, giant heather, bamboo,montane and submontane forest, savanna woodland and grassland (Langdale-Brown et al. 1964; Plumptre et al 2007), high and low-altitude wetlands, lakes and vegetation types specific to lava colonisation and thermal pools around the active volcanoes of Nyamluagira and Nyriagongo in Virunga Park (Mollaret, 1961; Lock, 1977; Howard, 1991). Papyrus and Carex wetlands are found around the lakes and some streams, and the lakes have their own habitat types varying from rocky and sandy edges to the pelagic zones in their depths. Virunga, Rwenzori Mountains and Bwindi Impenetrable national parks are World Heritage Sites, Queen Elizabeth Park is a Biosphere Reserve and Lake George is a Ramsar Site. Given their global importance, a brief description of each of these national parks is provided in the sections below.

4.4.1.1 Virunga National Park

Virunga National Park is located in the Albertine Rift, a part of the Great Rift Valley. It was designated as a National Park in 1925 and named as Albert National Park with an original area of 8090 km^2. The park boundary was delineated by the 1954 Ordinance covering an area of 7800 km^2 and its name was revised by Decree No. 69-041 of 22 August 1969 and changed to Virunga National Park. In 1979, it was designated as a World Heritage site and is currently managed by Congolese Institute for Nature Conservation (ICCN) (UNESCO). Virunga National Park comprises an outstanding diversity of habitats, ranging from swamps and steppes, marshlands, low altitude and afro-montane forest belts to unique afro-alpine vegetation and permanent glaciers to the snowfields of Rwenzori at an altitude of 5,109 m, and from lava plains to the savannahs on the slopes of volcanoes. It is also includes the massifs of Rwenzori and Virunga Mountains containing the two most active volcanoes of Africa: Nyamulagira (3,068 m) and Nyiragongo (3,470 m) and an exceptional biodiversity, notably mountain gorillas Gorilla beringei beringei that are globally threatened and the Eastern Lowland Gorilla (Gorilla beringei graueri).

The Park also contains important concentrations of wildlife, notably elephants, buffalo, okapi (Okapi johnstoni), red forest duiker (Cephalophus rubidus endemic to the Democratic Republic of the Congo and mountain Rwenzori respective, and the largest concentration of hippopotamuses (Hippopotamus amphibious) in Africa, with 20,000 individuals living on the banks of Lake Edward and along the Ishasha, Rwindi, Rutshuru and Semliki Rivers, and a home to palearctic migrant birds as far as Siberia. The Park contains 218 mammal species, 706 bird species, 109 reptile species and 78 amphibian species (UNESCO Plumptre et al. 2007; Igwabini et al. 2001). It also serves as refuge to 22 primate species of which three are the great ape – mountain gorilla (Gorilla beringei beringei), the eastern plain gorilla (Gorilla beringei graueri) and the eastern chimpanzee (Pan troglodytes schweinfurthi), with a third of the world population of mountain gorillas. The Park contains two highly important ecological corridors as it connects the different respective sectors: the Muaro corridor connects the Mikeno sector to the Nyamulagira sector; the west side connects the north sector to the centre sector of the Virunga massif. Queen Elizabeth National Park, a protected area contiguous with Virunga National Park from Uganda, also constitutes an ecological land corridor connecting the centre and north sectors, and Lake Edward and a network of rivers form an important aquatic corridor. These corridors facilitate the movement of wildlife, including the elephants.

On Uganda’s side, there are three major national parks that constitute part of the Greater Virunga Landscape namely Queen Elizabeth National Park (QENP), Rwenzori Mountains National Park (RMNP) and Bwindi Impenetrable National Park (BINP). All Uganda’s national parks are protected under the provisions of various national laws (The Constitution (1995), Uganda Wildlife Act Cap 200 of 2000, National Environment Act (2000), Local Government Act (1997), The Land Act (1998), the Forest and Tree Planting Act 2003 and the Uganda Wildlife Policy (1999).

4.4.1.2 Queen Elizabeth National Park

The Queen Elisabeth National Park (QENP), which covers 2080 km^2, is a key component of GVL. It is connected to Kigezi Wildlife Reserve (265 km^2), Kyambura Wildlife Reserve (154 km^2), Kibale National Park (795 km^2) in Uganda (Lamprey et al. 2006) and linked to Virunga National Park in the Democratic Republic of Congo (Figure 4.1). It also is contiguous with Uganda's central forest reserves of Kasyoha-Kitomi Forest Reserve (399 km^2), Kalinzu and Maramagambo Forest Reserves (428 km^2) in the east and borders Lake George and Edward, which are both connected by the Kazinga channel, and a range of crater lakes and a major wetland included on the Ramsar Convention's list of wetlands of international importance. QENP constitutes one out of the thirty Important Bird Area (IBA) sites in Uganda (Byaruhanga *et al*., 2001) and it contains critical wetlands that support migratory bird species, which are recognised under the Convention on Biological Diversity and the Ramsar Convention on wetlands (Pomeroy *et al.*,2004). QENP was named a UNESCO Man and Biosphere Reserve in 1979 because of the presence of 11 fishing villages within its boundaries which date to the time it was commissioned in 1952. Unfortunately, the park has suffered from heavy poaching and resource degradation. Much of this poaching was conducted by armed groups operating in the parks during and following the regime of Idi Amin in the mid1970s early 1980s. In the early 1980s, Kyambura WR and Kigezi WR suffered severe encroachment by local communities (Eltringham and Malpas 1983; Lamprey *et al*., 2004). In the late 1980s, intensive efforts were made to rehabilitate QENP. Law enforcement patrols were effective in curbing poaching. Encroaching communities in Kyambura WR were persuaded to leave, and efforts were made to contain the expansion of fishing communities within the borders of QENP. The boundaries of Kigezi WR were adjusted to exclude settled areas. However, human populations around the protected areas are increasing rapidly and there is a high demand for land, forest products, water, fish, grazing land, and game meat by the rural communities and also in local towns such as Kasese to the west of QENP. The problem is exacerbated by the general design of QENP and the adjacent wildlife reserves. With so many public roads running through it, virtually all parts of the ecosystem may be accessed with limited control by UWA. As a long and narrow protected area, core areas of the QECA ecosystem may be easily reached from all borders and many linkages between different areas of the park rely on narrow corridors of land.

4.4.1.3 Rwenzori Mountains National Park

The Rwenzori Mountains National Park is shared between Uganda and Democratic Republic of Congo and located less than a kilometer away from the equator and the third highest mountain in Africa at 5,109 m (after Kilimanjaro and Mount Kenya). The park is contiguous with the Virunga National Park in the Democratic Republic of Congo (DRC) and forms part of the Queen Elizabeth Conservation Area in Uganda covering an area of 996 km^2 of which the largest portion (70%) lies over an altitude of 2,500 m. Rwenzori mountains are well-known for their unique alpine flora which includes many species endemic to the Albertine Rift in the higher altitude zones including giant heathers and lobelias (Plumptre et al. 2007). It was recognized as as a World Heritage site in 1994 and Ramsar site in 2008 because of its importance for birds with a notable record of 217 species. It is home to the endangered Rwenzori black-fronted or red duiker (Cephalophus rubidus: Thomas, 1901), the threatened African savanna elephant, eastern chimpanzees and l'Hoest's monkey (Cercopithecus lhoesti: P. Sclater, 1899). There are several rivers and streams that originate from the mountains that serve as a source of water for industrial and domestic users in the region. The main threats to the park are agricultural encroachment resulting from human population pressure and illegal hunting of small ungulates.

In terms of management, Rwenzori Mountains National Park is managed by Uganda Wildlife Authority (formerly Uganda National Parks)) in accordance with the provisions of the National laws(The constitution (1995), Uganda Wildlife Act (2000), National Environment Management Act (2000), Forest and Tree planting Act (2003), Local Government Act (1987), The Land Act (1989) and international conventions (Convention of Biological Diversity 1992 (CBD), Convention on International Trade in Endangered Species (CITES), the RAMSAR convention 1971 and the World Heritage Convention 1972).It was gazetted in 1991 under statutory instrument number 3 in 1992 and the National Park's Act 1952. . It was designated a World Heritage Site in…and managed according to a ten-year General Management Plan

4.4.1.4 Bwindi Impenetrable National Park

Commissioned as a national park in 1991, Bwindi Impenetrable National Park (BINP) a world heritage site, is located on the eastern side of the Albertine Rift valley covering an area of 32,092 ha (331 km^2) and is one of the largest Afromontane lowland forest in East Africa. It has an altitudinal range of 1,160 to 2,706 m with phenomenon fauna and flora diversity and known for habiting nearly half of the remaining mountain gorilla population in the world and a high diversity of the species (over 200 tree species 347 species of forest birds, 202 species of butterflies). The park suffers from illegal harvesting of natural products, subsistence hunting and a lack of buffer areas and corridors for wildlife movement. The park is managed by Uganda Wildlife Authority and all management activities are implemented according to the 10-year rolling management plan. In terms of threats, BINP is situated in an area with the highest population density of 300 -400 persons per square km who are also very poor.

4.4.1.5 Volcanoes National Park

Volcanoes National Park (Parc National des Volcans) lies in northwestern Rwanda and borders Virunga National Park in the Democratic Republic of Congo and Mgahinga Gorilla National Park in Uganda. The national park is known also known for the mountain gorilla. It is home to five of the eight volcanoes of the Virunga Mountains (Karisimbi, Bisoke, Muhabura, Gahinga and Sabyinyo), which are covered in rainforest and bamboo. The park was first gazetted in 1925 as a small area bounded by Karisimbi, Visoke and Mikeno to protect the gorillas from poachers. In 1929, the borders of the park were extended further into Rwanda and into Democratic Republic of Congo to form the Albert National Park covering an area of 8090 km^2 managed by the Belgian colonial authorities. In 1958, 700 hectares of the park were cleared for a human settlement. After the Congo gained independence in 1960, the park was split into two, and upon Rwanda gaining independence from the French in 1962, the new government agreed to maintain the park as a conservation and tourist area. Due to population increase, the park was halved in area in 1969 and 1050 hectares of the park were cleared to grow pyrethrum.

The park is best known for the Mountain Gorilla (Gorilla beringei beringei). Other mammals include: golden monkey (Cercopithecus mitis kandti), black-fronted duiker (Cephalophus niger), buffalo (Syncerus caffer), Spotted Hyena (Crocuta crocuta) and bushbuck (Tragelaphus scriptus). There are also reported to be some elephants in the park, though these are now very rare. There are 178 recorded bird species, with at least 13 species and 16 subspecies endemic to the Virunga and Ruwenzori Mountains. In terms of threats, Volcanoes National Park was turned into a battlefield during the Rwandan Civil War (1990-1994), with the park headquarters being attacked in 1992. All management activities were abandoned not until 1999 when the area was deemed to be safe for business and under control by government forces.

The presence of the mountain gorillas and contiguous nature of BINP (Uganda), Volcanoes National Park (Rwanda), and Virunga National Park (DRC) created an opportunity for the conservation organizations namely Fauna & Flora International (FFI) and the World Wide Fund for Nature (WWF) to establish a partnership - International Gorilla Conservation Program (IGCP) in 1991 with support from the three national governments. The goal of the International Gorilla Conservation Programme (IGCP) is to ensure the conservation of mountain gorillas and their regional afromontane forest habitat in Rwanda, Uganda and the Democratic Republic of Congo (DRC). The partnership also incorporates the respective protected area authorities of the three countries in which IGCP works: the Rwanda Development Board (RDB), the Uganda Wildlife Authority (UWA) and the *Institut Congolais pour la Conservation de la Nature* (ICCN). The recognition that several species need to be managed at a scale larger than any individual protected area is one of the main reasons driving this collaboration (Plumptre et al. 2007; Plumptre et al. 2008). In order to address the above threats to wildlife conservation, a number of initiatives have been undertaken in the last two decades at a regional and protected area level. First, a Ten – Year Transboundary Strategic Plan (2006-2016) for the Central Albertine Rift Transboundary Protected Area Network (also referred to as the Greater Virunga Landscape Plan) was developed by the three Protected Areas Authorities of the Democratic Republic of Congo, Rwanda and Uganda directed by the Transboundary Core Secretariat (Transboundary Core Secretariat, 2006). The transboundary strategic plan is being implemented under a Memorandum of Understanding signed between the three Protected Area Authorities (ICCN, UWA, and ORTPN).

4.4.2 Vegetation changes in the GVL from early 1950s to 2006

Wildlife Conservation Society assessed the vegetation changes in GVL from 1950s to 2006 using aerial photography taken in the 1950s (1954 –QENP and 1959 – VNP) and 1990 (QENP), and the recent vegetation mapping based on recent aerial photo imagery taken in 2006 by the Wildlife Conservation Society (WCS) Flight Program (Plumptre et al. 2010a). The imagery was from before and after periods of major changes in the large mammal faunas in the landscape. Aerial photo coverage of Uganda was taken in 1954 and used to create the East Africa 1:50,000 maps of Uganda that are still used today. Copies of the photos were obtained for QENP and scanned into a computer. The computer software package ENSO Mosaic was used to create photo mosaics of the park which are orthorectified. Photo Mosaics from 1958 of the southern section of VNP was provided by WWF to WCS. Imagery from 2006 was taken with a digital camera and ENSO Mosaic used to create similar photo mosaics for both QENP and VNP. The vegetation type of 250 x 250 meter cells was identified by the same technician for both sets of photo mosaics and specific descriptions of the 29 habitat classes were defined to allow accurate classification. The resolution of the imagery was less than 1 metre making identification of the habitat classes relatively simple. The overall impression is that the vegetation of the landscape has become more wooded/forested (Figure 4.2). The area of grassland has diminished and been replaced by woodland/Bush shrub and there has been some expansion of the Maramagambo Forest and Kisali Mounts forests. Another clear difference is the isolation of the landscape in a sea of agriculture.

Whereas in the 1950s there was very little agriculture around the parks now farms are cultivated up to the boundary along most of the limit of the parks. This landscape contains more terrestrial vertebrate species and more endemic and threatened species than any other site in Africa (Plumptre et al. 2007). The Greater Virunga landscape is typified by the coexistence of woody plants and grasses, with the relative (and wide-ranging) proportions of each being influenced predominantly by water availability, fire, and people (Scholes and Archer 1997; Baxter and Geitz 2005). Currently, there are at least two horrendous droughts in the region with negative effects on elephants perhaps related to widespread climate change, but very much exacerbated by habitat degradation caused due to overgrazing, deforestation, agricultural expansion and extractive industry disturbances. Since 1996, protected areas in the GVL have suffered from high pressure due to high population density and the effects of civil war in the DRC. Displaced people have moved into Virunga National Park as a result of the insecurity and others have taken advantage of the situation to encroach and grab land and resources. Arguably, however, a more urgent and immediate problem facing the elephants in Africa is the increasing trend in habitat loss and fragmentation (Gordon et al. 2004; Plumptre et al. 2008), apparently fueled by human population growth on the continent, and the eminent climate change.

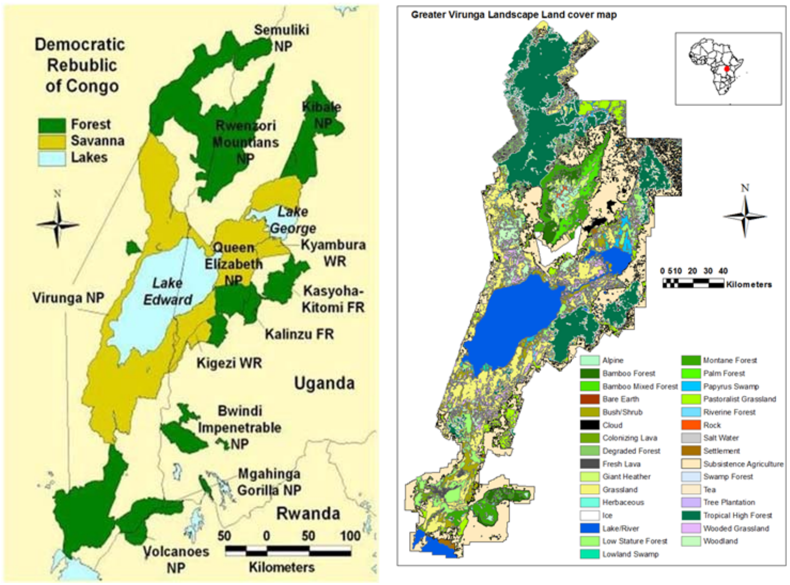


Figure 4.4 Map of Greater Virunga Landscape vegetation map developed in 2006 by WCS from aerial photographs taken in the 1950s coupled with satellite imagery acquired in 2006 (Plumptre et al. 2006)

4.4.3 Conceptual model

There is a concern that elephant populations in Greater Virunga landscape, which had started to bounce back to the numbers that they were in the early 1960s could easily slip back due to an overall reduction in suitable habitats caused by land use changes, and a decline in water supply. We present a deterministic-based model of elephant population–habitat dynamics by accounting for water supply and climate change effects (Figure 4.3). By modeling vegetation cover demographics, elephant–habitat interactions, stochastic environmental variables such as precipitation and temperature, and spatial variability of water supplies, including glacier melt contribution on top of Mt. Rwenzori and runoff, the model is used to analyze changes in elephant population acting via natality and mortality over the last 55 years. In the conceptual model, elephant population is a number of factors or stressors. First, the density-dependant factors mainly food, space and water resources, which limit population increase beyond the area's carrying capacity. Second, war and poaching affect the elephant population by increasing the death risk to all age groups, but more especially to calves. As the elephants free the war zones, the calves lose track of their mothers or social groups and easily get killed. War indirectly contributes to the breakdown in law enforcement and elephants become more susceptible to poaching. Third, poaching has similar effects on elephant population through increased mortality and the removal of adult males and females affects the reproduction rate. Forth, climate change affects the elephant population both directly and indirectly. In the case of high temperature, the direct consequence is physiological stress resulting in diminished reproduction rate and increased mortality risk among the calves and juveNile elephants. Climate change is also considered to after the elephant population by reducing the water resources and biomass production due to increased evapotranspiration and loss of suitable habitat respectively. Habitat change is also driven human activities, particularly expansion of agricultural land, settlement in corridor areas, and burning of suitable habitat. Elephants depend mainly on savanna and forest vegetation, occasionally relying on wetlands/swamps for water, salt, body temperature regulation through wallowing and for navigation to other patches in the landscape.

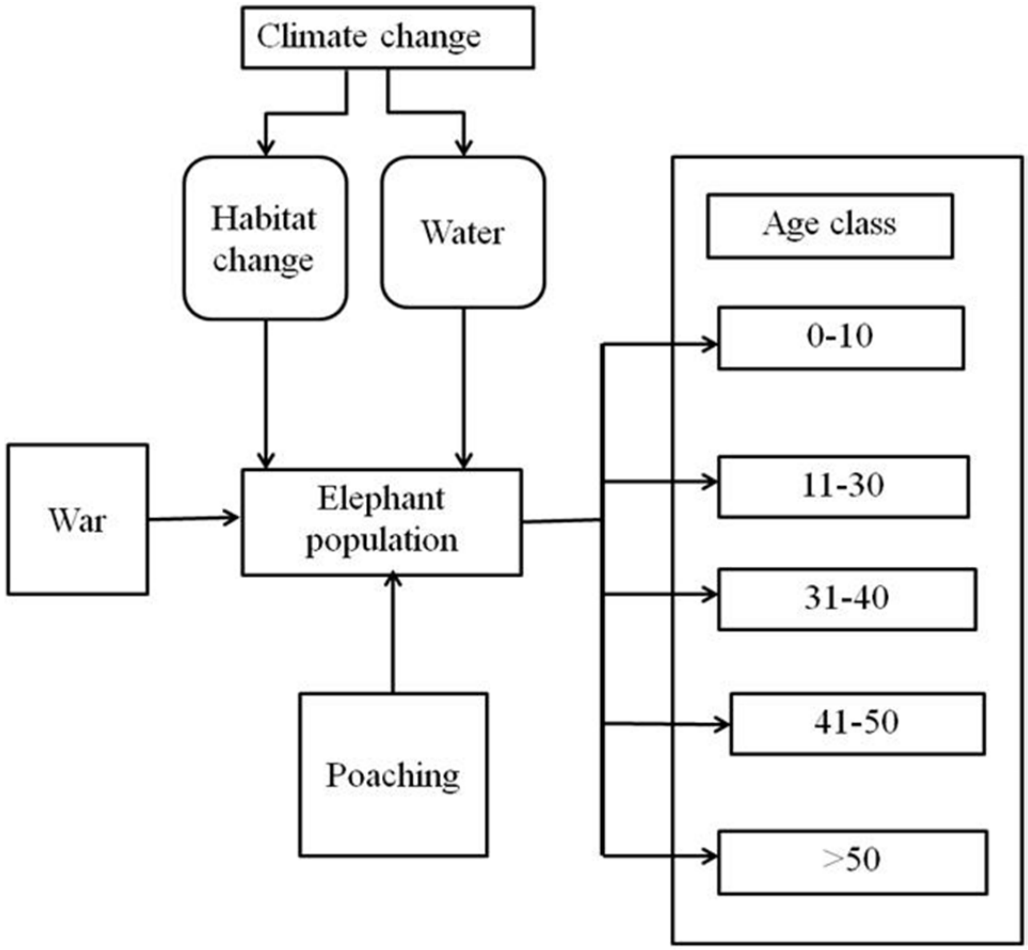


Figure 4.5 Conceptual model for the age-class specific population dynamics influenced by climate change, water resources, habitat, war and poaching in GVL

The habitat component of the model produces long-term vegetation conversions, and associated with emergent declines in precipitation and Mount Rwenzori glacial stock. Including water supply and climatic changes in the elephant age specific population-land cover change model had the expected effect of increasing elephant natality and reducing mortality, mainly via increased reproductive success, although at a vegetation cover change of nearly one percent, and an elephant population density of 1.0 elephant/km^2, savanna woodlands turned out to be the most suitable habitat for elephant survival.

4.5 Data

This section provides a detailed account of the sources of information and field data used in the calibration and validation of the model. The information collected include observed elephant numbers, habitat change, hydroclimatology, wars, poaching and water resources.

4.5.1 Elephant population

Elephant population data was obtained from the large mammal census databases and associated government reports of protected area authorities of Uganda (Uganda Wildlife Authority), DRC (ICCN), and Rwanda (RDB), and conservation organizations such as WCS and WWF that have worked in the region and supported conservation programs in the region for a long time (Plumptre et al.2006, 2008, 2010a). As a way of cross-checking the information, intensive reviews of published survey reports in peer-reviewed articles was also done. In the early 1950s till the late 1970s, most of the census of large mammals was conducted by foot (also known as ground survey) by walking along transects systematically designed for this purpose and identifying all scats such as dung, footmarks, and physical count of live animals. After the 1970s, most of the large mammal counts is done by aerial means, specifically the use of a four-man carrier light aircraft and animal count is done following The methods used are the standard methods that were proposed by Mike Norton Griffiths (1978) and Craig (undated).

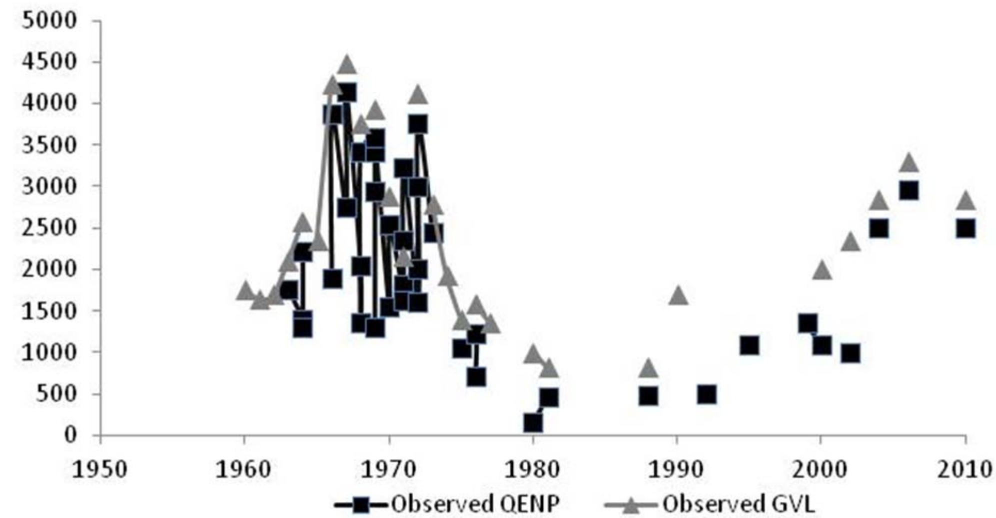


Figure 4.6 Elephant population in the Greater Virunga Landscape from 1960-2010

Survivorship data for ten-year intervals is based on elephant counts by wildlife authorities in the region, that is, Uganda Wildlife Authority (UWA), and *Institut Congolais pour la Conservation de la Nature* (ICCN) of Democratic Republic of Congo. According to Laws (1969b), the values of the first age class are modified for both males and females from value of 0.388, which is based on an estimate of natality and extrapolation from adult mortality rates but excluded from the age-classes that contribute to natality (Armbruster and Lande 1993). For tracking purposes, however, this consideration is important if we are to account for a sudden influx in births associated with the second age-class 11 to 30. This value suggests that 37.5% of the population would be comprised of the 0-10-year age class graduates.

Table 4.2 Elephant number by age group in the year 1960 and 2006

Age structure	Elephant Numbers by age group 2006	Proportion by age group in 2006	Proportion by age group in 1960
0-10	1240	0.375	660
11-30	998	0.302	531
31-40	599	0.181	319
41-50	398	0.120	212
>50	72	0.022	38
Total population	3307		1760

Individual birth and death during the twenty-year interval are simulated in accordance with Leslie (1945) as follows. We use the Random Built-in function to generate uniform random numbers. The reproductive probabilities are evaluated based on the size of the total population from the last time step using the counter and delay built-in functions. Individual females give birth to zero or two offspring with density-dependent probabilities as discussed earlier in the report. A random number then determines the state of the environmental parameters, and the corresponding survivorship values generated from the population dynamic submodel. Individuals survive to the next age class or die based on the mortality rate index and driven by the habitat quality factors (habitat quality index). This yields the state of the population at the end of one-ten year step. The relationship between habitat quality and the probability of elephant extinction will be examined in simulations both with and without human driven killings (poaching, and retaliation or defensive assult). A greater number of simulations will be conducted in order to attain acceptable statistical confidence levels.

4.5.2 Habitat change

As part of the long term vegetation change mapping for Virunga National Park and Queen Elizabeth National Park, Wildlife Conservation Society (Plumptre et al. 2010b) calculated the woody cover change (Table 4.1) and shows that there have been increases and decreases in woody cover in different parts of the GVL between the 1950s and 2006. In general, Plumptre et al. 2011 noted an increase in woody cover of 1,579 km^2 compared to a decrease of 334 km^2. They attributed the net gain in woody vegetation cover (1245 km^2) to a decrease in large mammals in the landscape from the 1970s as demonstrated by the observed elephant population trend, continued recovery of the vegetation from the human resettlement away from the landscape in 1880s (Spinage 1970), increasing rainfall, climatic variability, and changes in fire frequency. In Virunga National park alone, 98.2 km^2 were encroached upon by humans and the new settlement within 2 km of QENP was recorded as 179,200 people (Plumptre et al. 2006).

Table 4.3 Woody vegetation cover change in QENP and Virunga NP in GVL (1950s – 2006)

National Park	Area (km^2)	increase in woody cover (km^2) in some areas	Change (%)	decrease in woody cover (km^2) in some areas	Change (%)	Net change in woody cover (km2) from
QENP	2080	1021	28%	260	7%	761
Virunga	8090	558	40.5%	74	5%	484
Total	10,170	1579		334	1245	1245

Source: Plumptre et al. 2006, 2010b

The land cover map produced by WCS in 2006 (Plumptre et al. 2010b) was reclassified into four major land uses namely forest, savanna woodland and grasslands, wetlands and water, and human settlement and agriculture. The reason for this broad classification was that elephants depend a lot on savanna woodland and grasslands, and forest habitats for food, but need water for drinking and body temperature regulation. Elephants, however, use less dense human settlement and agricultural areas during migration and occasionally feed on the crops in the course of their movement to other patches within the landscape.

Table 4.4 Greater Virunga Landscape land cover by proportion in year 2006 (WCS 2006)

Land cover type 2006	Proportion of land cover type
Forest (FO1) in (Sq km)	7721.95
Savanna (S1) in (Sq km)	8,859.56
Wetland (W) in (Sq km)	3,539.22
Human settlement/agric (H) in (Sq km)	5,569.98
Total area (TA) in (Sq km)	25,690.71
Total woody cover (TWC) = (FO1+S1) in (Sq km)	16,581.51
Percentage of TWC (PTWC) in GVL	0.65
FO1proportion of TWC (%), FO1P= (FO1/TWC)	0.53
S1 proportion of TWC (%), S1P = (S1/TWC)	0.47
Net gain in woody cover (NWC) from 1950s to 2006 (Sq km)	1245
Percentage net gain in woody vegetation (PNWC) 1950s to 2006	0.08
1950s	
Woody vegetation (WV) in 1950s = (TWC-NWC) in Sq km	15,336.51
Percentage woody vegetation (PWV) =(PTWC-PNWC)	0.56
Annual net woody cover gain (ANWC) in 55 years = (PNWC/55)	0.00148
Forest (FO2) in Sq km = (FOIP*WV)	7142.16
Savanna (S2) in Sq km = (SIP*WV)	8194.35

Calculated values based on land cover map produced by WCS in 2006 (Plumptre et al., 2006, 2010b)

In STELLA, habitat at time t (H_t) is empirical represented as

$\boldsymbol{H_t = H(t-dt) + (HCinc - HCDEC) * dt}$ where HCinc is habitat increase in km2, HCDEC is habitat decrease in km2, but $HCinc = H * FP$, and $HCDEC = H * SP$ where FP and SP are proportions of forest, and savanna grassland and woodlands in the entire landscape respectively.

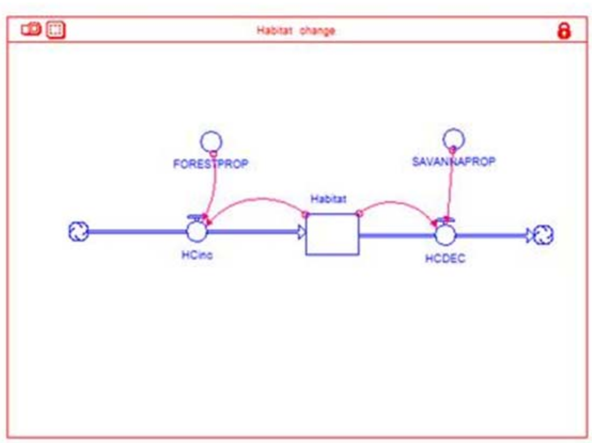


Figure 4.7 Habitat change sub model in STELLA

4.5.3 Hydroclimatology

According to the study conducted by Wildlife Conservation Society scientists (Plumptre et al. 2009 Chap 6: p89), rainfall in and around the Greater Virunga Landscape has not changed greatly since the early 1900s. Climate variability and change is simulated with the prior knowledge that there are two drought cycles in a year, that is, December to February, and June to August. Analysis of climatologic data from nine different sites with at least 20 years of continuous data showed an increasing trend in annual rainfall (Plumptre et al. 2010b). According to Plumptre et al. (2011), the study results revealed significant ($P<0.05$) increases in total annual rainfall in Beni (1974-2007), Mweya (1958-2007), Kabale (1918-1996), Kiamara (1982-2007) and Ruhengeri airstrip in Rwanda (1928-1986) over time but the rest of the stations did not show any significant trends. At local scale, rainfall varies greatly across the landscape. The driest parts of the landscape occur in the savanna areas to the north and south of Lake Edward recording an average monthly rainfall of 30-40 mm. The Albertine Rift climatological assessment study conducted by Seimon and Picton-Phillipps (2009) revealed that both precipitation and temperature for the region is going to increase over the next 100 years. According to this study, the mean annual temperature in the base year (1990) was recorded as 22.7 ^{o}C (max 26.0 and min 15.0 ^{o}C) and the modeled temperature for 2030, 2060, and 2090 was 23.6, 24.7, and 26.3 degrees celsius respectively. Similarly, the mean annual precipitation in 1990 was noted to be 1199 mm, 1233 mm (2030), 1287 mm (2060), and 1406 mm in 2090 (Seimon and Picton-Phillipps, 2009). Across the entire modeling period, the variation in precipitation ranges from 821 mm – 2098 mm.

For this study, weather data from Lwiro weather station provided by the *Observatoire Volcanoligue de,* Goma was selected for use in the computation of ET. Data from other weather stations such as Beni, Butembo, and Mweya were available, but had several missing data for some years. Other sources of hydroclimatological data are the Climate Research Unit (CRU) at the University of East Anglia, UK, and NOAA CIRES Twentieth Century Global Reanalysis Version 2, Research Data Archive at the National Center for Atmospheric Research, Computational and Information Systems Laboratory (Compo, et al. 2009). The CRU data was summarized to historical monthly averages for the years and variability at an annual scale was lacking while the NOAA data was good, but with similar concerns and learnt about this source rather too late and needed more computational time to convert it into a usable format.

This study uses 51 years of historical climate observations of daily temperature and precipitation from two out of the eight weather stations – Lwiro and Butembo in DRC located within the study region. These climate data from these two weather stations were selected for use because they had the highest consistent record of observations from 1960 to 2010 compared to the rest of the stations in the region. Following the description of the IPCC AR5 climate change results based on the four RCP scenarios earlier discussed, East Africa was one of the five regions identified for analysis of regional climate change by IPCC (2013). Of the four scenarios, future climate change values based on RCP2.6, RCP6.0, and RCP8.5 were selected for implementation of the climate change effect on elephant population dynamics. Prior to integrating future climate change prediction, the annual historical precipitation data was detrended to incorporate climate change induced by temperature increase. This was achieved by performing a regression analysis to estimate the temporal trend in the time-series. For example, if P is precipitation and *t* is the time in years, the fitted regression $\acute{P} = \alpha + \beta t$ (Figure 4.8). The time scale in this case was from 1960 to 2010. The fitted regression equation $\acute{\mathrm{P}} = 1.4024x - 1493.9$ was used to estimate an average historical precipitation (β) of 1254.80 mm for annual precipitation, and an average historical annual precipitation increase of 25.86 mm. The detrended equation is given as:

$P - \beta(t) + RCP_{\acute{P}}/100)$, where P is the historical precipitation, t is the year and $RCP_{\acute{P}}$ is the future annual precipitation change in percentage. The historical precipitation trends calculated from the regression analysis were removed from the observed temperature data to isolate climate change trend from natural variability.

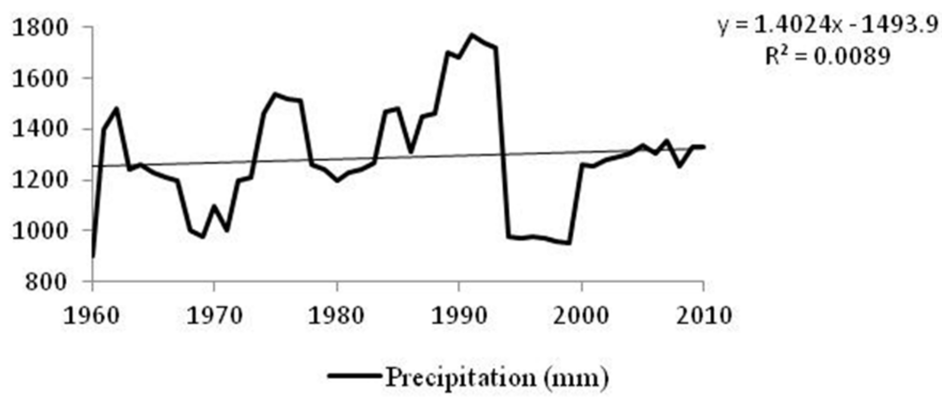


Figure 4.8 Baseline precipitation from Lwiro weather station in Democratic Republic of Congo

This was done by subtracting the trend component derived from the regression model from actual observation. In so doing, it became easy to incorporate various climate change scenarios into the model. The baseline assessment (identified as BL scenario) is a 51 year scenario without the influence of historic warming and is useful in evaluating exclusive effects of climate change scenarios. The detrended baseline data from 1960 to 2010 was then recalculated to reflect the three precipitation change scenarios identified as RCP2.6, which is the lowest, RCP6.0 (medium with a 10% precipitation increase), and RCP8.5, the highest with 18% precipitation increase (Figure 4.9).

The warming trends predicted by IPCC represent long term increases over the current temperature data over the next 100 years. To incorporate these predictions, the IPCC warming predictions were added to the baseline data as a linear trend for mean annual precipitation.

Future temperature change over the next 100 years was modeled as an absolute value for each of the RCPs (IPCC, 2013). The future temperature change values for East Africa selected were RCP2.6 = 1.6 ^{o}C, RCP6.0 = 2.8 ^{o}C, and RCP8.5 = 4.3 ^{o}C (IPCC, 2013). As such, incorporation of temperature change did not require detrending rather a straight forward calculation of the annual increment to the historical values was done as per the equation below: $T = T_h + \left(\frac{T_f}{100}\right) \times t$ where, T is the new calculated temperature for the elephant modeling scenario, T_h is the historical temperature, T_f/100 is the RCP scenario temperature per year, and t is year (Figure 4.9).

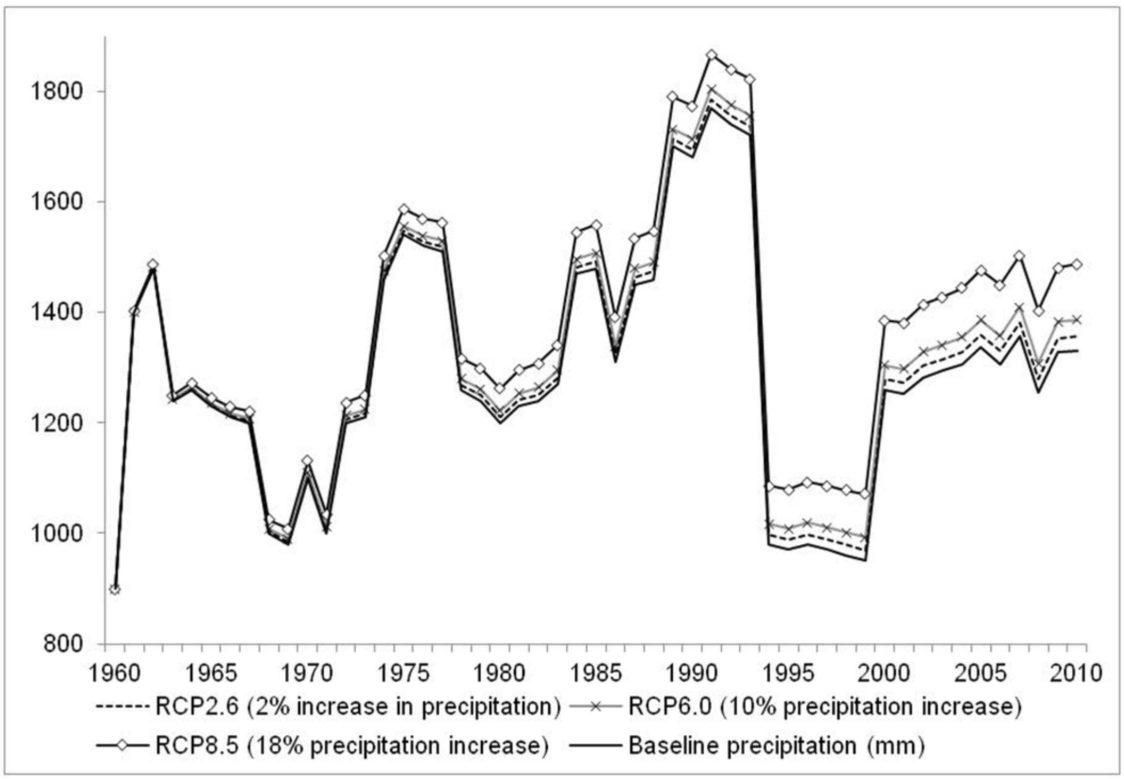


Figure 4.9 Baseline and calculated future precipitation increase based upon selected IPCC Representative Concentration Pathways results for the study area.

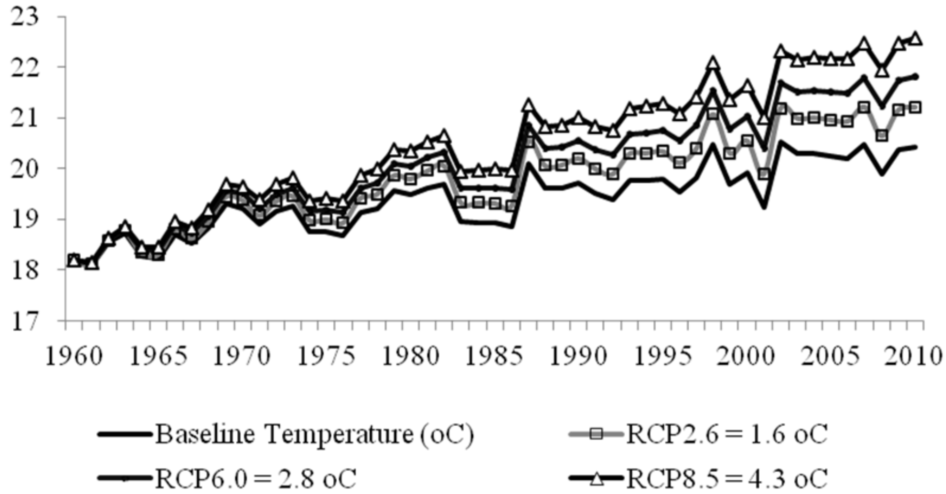


Figure 4.10 Baseline and calculated future climate change temperature based on selected IPCC RCP results for the study area

In order to account for the available water resources in the landscape, surface runoff, glacier melt contribution to river discharge and water in reservoirs mainly lakes and river was considered. Surface runoff refers to the loss of water from an area by flow over the land surface. Runoff flow is composed of two main elements: base flow, which has its origin in ground water, and surface runoff, which is the accumulation of rainfall that drains to the stream. The Soil Conservation Service (SCS) Curve Number Method was used to compute the runoff in ESRI's ArcGIS ver 10.2.1 with the help of the arc hydrology tools. The Soil Conservation Service (SCS) Curve Number Method is a versatile and widely used procedure for runoff estimation because it gives consistently usable results (Rao et al. 1996; Sharma et al. 2001; Gumbo et al. 2002; Senay and Verdin 2004). The curve Number (CN) measures the relationship between initial abstraction and potential maximum retention of an area after a storm event. It *CN* method is based on the relationships between rainfall depth, P in (inches), runoff depth, the stormflow, Q in inches, and storage factor, St (USDoA, 1986; Pilgrim and Cordery 1993; Schulze et al., 1992; Gumbo et al. 2002; Brooks et al. 2003; Senay and Verdin 2004) represented as:

Q is given as $Q = \frac{(\mathrm{P}-0.2\mathrm{St})^2}{(\mathrm{P}+0.8\mathrm{St})}$ and $S_{\mathrm{t}} = \frac{1000}{\mathrm{CN}} - 10$

The maximum potential retention (*S*) is related to the soil and land cover conditions of the watershed through the curve number equation above and *S* is a dimensionless watershed parameter ranging from 0 to 100. The USDoA SCS developed tables of runoff curve numbers corresponding to various land use, land cover types that are available in the SCS-SA User Manual (USDoA, 1986). A *CN* of 100 represents a limiting condition of a perfectly impermeable watershed with zero retention and thus all the rainfall becoming runoff. A *CN* of zero conceptually represents the other extreme, with the watershed abstracting all rainfall with no runoff regardless of the rainfall amount. In order to compute the water balance, Thornthwaite-Mather approach (Thornthwaite 1948, Thornthwaite & Mather 1955) was used. Thornthwaite's method underestimates potential evapotranspiration during the summer when the solar radiation received at the surface is at its annual maximum (Rosenberg et al., 1983). In addition, this method does not capture local soil moisture patterns that vary with slope and aspect and which are important state variables in capturing the ecological differences in high altitude and forest dominated areas. Penman-Monteith equation (Penman, 1981) or Hargreaves method (Hargreaves, 1981; Hargreaves and Samani, 2003) are the best alternative methods widely used, but they are data-intensive (Vorosmarty et al. 1993; Federer et al. 1998).

The Penman-Monteith equation is a combined equation that considers both energy supply and mass transfer of water vapor from the evaporating surface derived from the leaf energy balance equation (Rosenberg et al., 1983; Bonan 1989). The Penman-Monteith method is very rigorous, however, it was noted to have limitations when applied to highly forested areas because the canopy resistance term cannot be easily parameterized (Rosenberg et al. 1983; Bonan 1989; Landsberg and Waring, 1997). It was also developed to predict evaporation from open water, bare soil, and grass (Penman, 1948). Furthermore, since it is derived from the energy balance of a leaf, the Penman-Monteith equation ignores fluxes of water vapor to and from the soil (Landsberg and Waring, 1997). Despite its shortcomings, Thornwaite method was selected for compilation of evapotranspiration (ET) because of several reasons namely: 1) scarcity of climatic and adequate land-atmospheric data required for compilation of ET; 2) Thornwaite method has been used extensively used in North America and is proved to produce reasonable results regardless of its shortcomings (Hulme et al. 1996). In Africa, Feddema (1998) used this method to evaluate the effects of soil water holding capacity assumptions on estimates of African evapotranspiration rates, moisture deficit, and moisture surplus conditions.

Thornthwaite's method is as follows:

$$\boldsymbol{E}'_p = \begin{cases} 0, & \boldsymbol{T} < \boldsymbol{0^o}\, C \\ 16\left(\frac{10\boldsymbol{T}}{\boldsymbol{I}}\right)^a, & 0 \le \boldsymbol{T} < 26.5\ ^o C \\ -415.85 + 32.24\boldsymbol{T} - \boldsymbol{0.43}\boldsymbol{T}^2, & \boldsymbol{T} \ge \boldsymbol{26.5^o C} \end{cases}$$

Where $\boldsymbol{E}'_p$ is monthly unadjusted potential evapotranspiration in mm, ***T*** is mean monthly surface air temperature (^{o}C), and ***I*,** the annual heat index, is given by the equation:

$$\boldsymbol{I} = \sum_{\boldsymbol{i}=1}^{\boldsymbol{12}} \left(\frac{\boldsymbol{T_i}}{5.0}\right)^{1.514}$$

and **a** is: $\boldsymbol{a} = 6.75 * 10^{-7}\boldsymbol{I}^3 - \boldsymbol{7.7} * \boldsymbol{10^{-5}}\boldsymbol{I}^2 + \boldsymbol{1.79} * 10^{-\boldsymbol{2}}\boldsymbol{I} + 0.49$

In case the air temperature is measured on a monthly scale, then potential evapotranspiration is adjusted for variable day (h) and month (Θ) lengths as follows:

$$\boldsymbol{E}_p = \boldsymbol{E}'_p \frac{\Theta}{\boldsymbol{30}}\ \frac{\boldsymbol{h}}{\boldsymbol{12}}$$

In this study, the temperature data used was already compiled on an annual basis and no adjustment of ET was needed. After manipulating the data in Microsoft Excel, the calculated ET was then used in the STELLA model to implement the water balance computation (Figure 4.11).

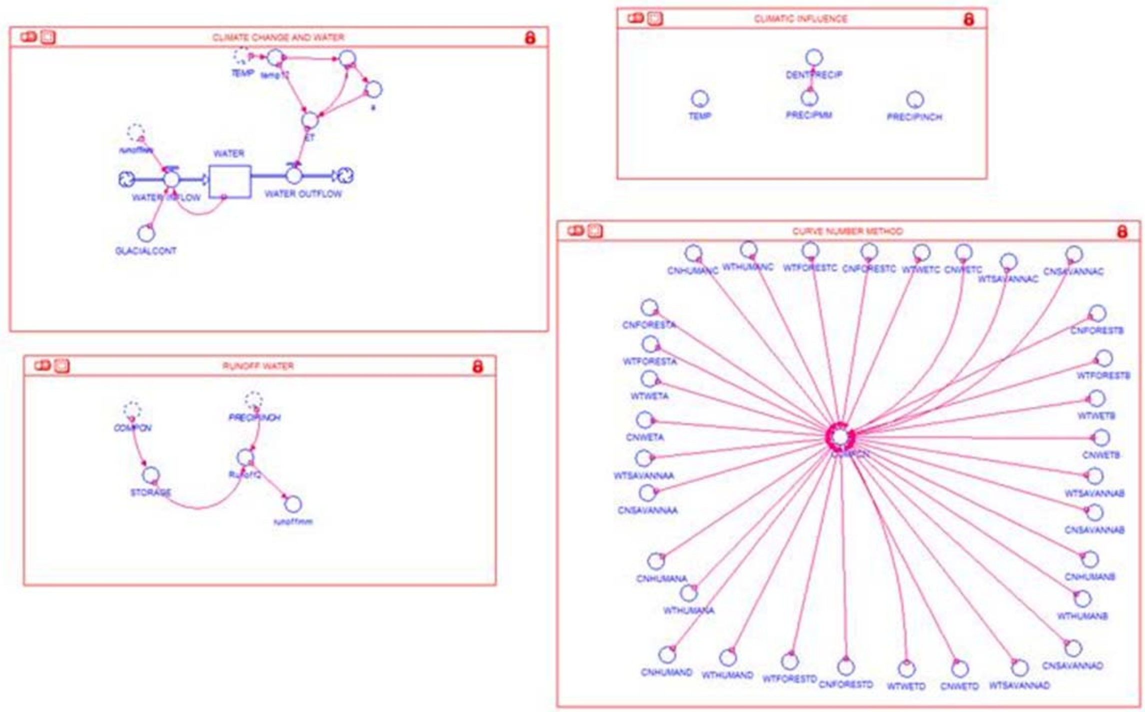


Figure 4.11 Water resources and climate change sub models

The main source of water in this region is precipitation which ends up as runoff, groundwater, and in reservoirs through river discharge. The other source of water is the glacier melt contributing less than 1% toward river discharge. In STELLA, water stock was computed as the difference between sources of water (i.e. runoff, glacier contribution) minus the loss through evapotranspiration (ET) and expressed in form of an equation: $Y = Y(t - dt) + (Y_i - Y_o) * dt$ where, Y_o is water loss in form of ET, Y_i is water available in the landscape given as: $Y_i = R + (Y * gm)$ where, R is the runoff in mm, *gm* is the contribution from glacial melt from mountain Rwenzori ice fields. Runoff was compiled following the Curve Number (CN) methodology already described in the methodology section implemented both ArcGIS, specifically to assign the curve numbers to the land cover matched with the corresponding soil hydrologic group, and in STELLA where a composite curve number (COMPCN) was calculated as per the equation below:

$$COMPCN = (CNFO_A * wFO_A) + (CNFO_B * wFO_B) + (CNFO_C * wFO_C) + CNFO_D * wFO_D + (CNS_B * wS_B) + (CNS_C * wS_C) + (CNS_D * wS_D) + (CNWET_A * wWET_A) + (CNWET_B * wWET_B) + +(CNWET_C * wWET_C) + +(CNWET_D * wWET_D) + +(CNH_A * wH_A) + (CNH_B * wH_B) + (CNH_C * wH_C) + (CNH_D * wH_D)$$

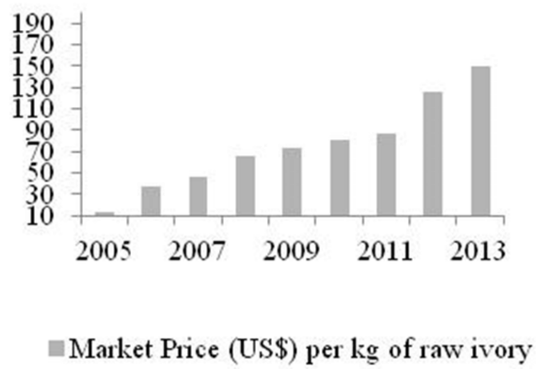
Where, *CNFO, CNS, CNWET, CNH* are curve numbers of forest, savanna, wetland, and human settlement and agriculture land cover classes associated with the soil hydrologic group represented by the subscripts A to D.

4.5.4 Anthropogenic data

A detailed description of the impacts of anthropogenic factors mainly climate change, poaching, poverty and wars have been provided in the previous sections of this chapter.

4.5.4.1 Poaching

Data on poaching was acquired from secondary sources specifically CITES, Inernational Fund for Animal Welfare and peer reviewed journal articles. According to both CITES (2013), and the International Fund for Animal Welfare (IFAW, 2012) both the price for raw ivory at the source and international market have been increasing (Figure 4.12) and the largest international market being Asia. The demand for ivory was considered to be on a steady increase if not much effort is done to limit the supply through poaching, stop civil wars, human population control, and reduce poverty. In the model, poaching was modeled as a function of price of ivory (*pi*) changing with time, *t*, human population (*H*), and law enforcement (*l*). Poaching was treated as having major impacts on the total reproduction, mortality, and natality rates.



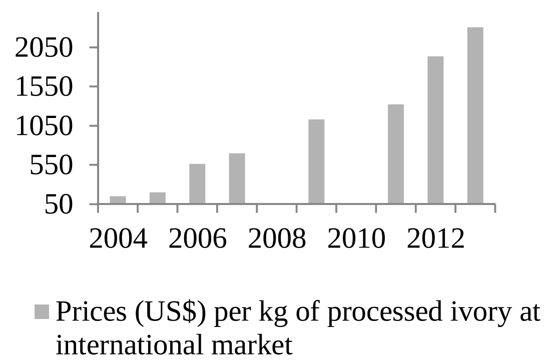


Figure 4.12 Prices of raw ivory and international prices

(Source: http://comtrade.un.org, IFAW, 2012; CITES 2014)

Poaching was empirically defined by the equation: $Z = Z(t - dt) + (Z_{inc} - Z_{dec}) * dt$ and $Z_{inc} = Z * k$, and $Z_{dec} = Z * d$ where Z is poaching at time t, Z_{inc}, and Z_{dec} are poaching increase and decrease respectively, *k* and *d* are poaching increase and decrease indices respectively. Poaching increase is precipitated by human population increase, poverty, and price of ivory while poaching decrease is driven by deliberate management interventions such as law enforcement, monitoring and surveillance, and integrated conservation and development program that enhance incomes of protected area frontline households. Given the factor that poaching accelerating and reducing factors are very dynamic, *k* and *d* were represented in graphic form expressed by the following equations: $k = graph(hp * w_{hp} + q * w_q + x * w_x)$ where hp is human population, q is poverty, x is price of ivory per kg, whp, wq, and wx are weighting factor of human population, poverty, and price respectively. In the Figure 4.13, these factors are represented as HumanPop, price, and poverty while the weighting indices are denoted as WPOP, WPRICE, and WPOVERTY. On the other hand, the indirect factors that help to reduce *d* are income, monitoring, and law enforcement which vary with time. Thus, *d* is also expressed as

$d = Graph(I * Q_i + L * Q_l + M * Q_m)$ where, *I* is he income, *L* is law enforcement, *M* is monitoring and surveillance, *Qi*, *Ql*, and *Qm* are the weighting factors of income, law enforcement, and monitoring and surveillance respectively.

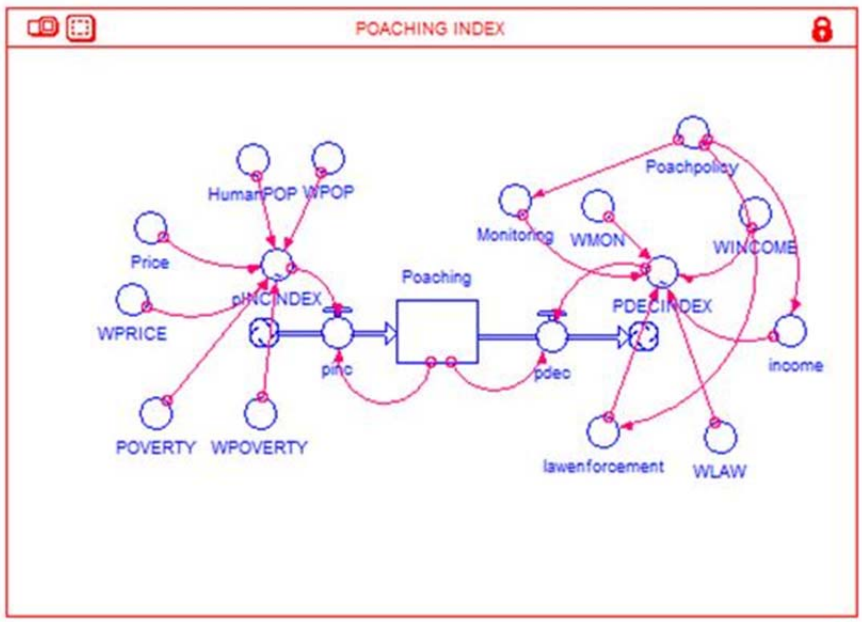


Figure 4.13 Poaching sub model in STELLA

4.5.4.2 Poverty

In modeling the impacts of poverty on elephant population dynamics, a number of considerations were made. First, poverty measured in terms of household incomes, increases the risk of households getting involved in poaching of elephants for monetary purposes; and second, poverty is related to food insecurity and households that are food insecure either due to loss of crops and livestock to wild animal raids and predation or limited economic opportunities are more likely to kill elephants. Available records in all three GVL shared countries, poverty has been increasing over the last ten years as earlier discussed under section 4.4.3 above. In this study, poverty (P) in the 1960 was considered to be less (10%) and 75% in 2006, but varying with time as described by the equation, $P = 10 + (0.75) * t$ where, p is poverty and t is time.

4.5.4.3 Wars

The frequency and magnitude war constructed from Table 4.1 under literature section were used to implement the model simulations. Under the baseline scenario, war impact was represented in a graphical format (Figure 4.14) so as to adequately capture the time lag, frequency and intensity or magnitude of the war on elephant natality and mortality in GVL. The impact of the magnitude or intensity of war on elephants was developed based on the period the war lasted, the associated negative impacts particularly on the habitat, the number of elephants killed and attributed to the war impacts, and the human displacement into or near protected areas reported in published journal articles profiled in Table 4.1. The relative percentages were then assigned to those years when the war actually occurred and extended to 1-2 year in order to account for the residual effects of the war. The understanding was that if the war was very intense and lasted for a short time (e.g. 1-2 years), the impact is likely to be high and elephant losses due to the war were also high. On the other hand, if the war was very mild and lasted a slightly longer period (3-5 years), then the impacts were slow acting and the residual effects could be spread over to the next two years. It is also known that ecological impacts on a system or wildlife species acting external to the system take time to be detected.

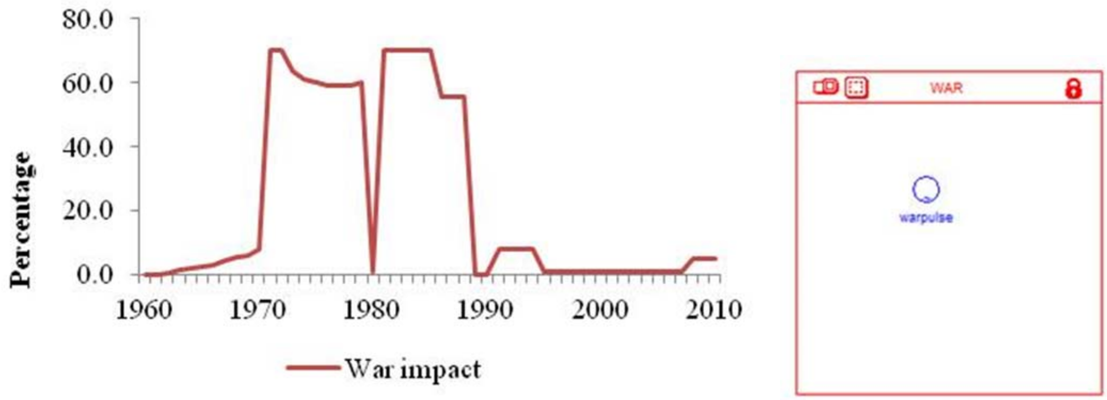


Figure 4.14 Trend, magnitude and frequency of war profiled from 1960-2010 (left) and the war submodel in STELLA (right)

4.6 Empirical model

Elephant population dynamics modeling was implemented using Structural Thinking, Experimental Learning Laboratory with Animation (STELLA) software. The STELLA (High Performance Systems, Inc., Lebanon, NH) dynamic systems software package is an icon-based simulation tool that uses differential equations represented as stocks and flows. This software has been highly used for understanding population dynamics and economic fluxes (Fitzharris 1998; Costanza and Voinov, 2001; Seppelt and Richta, 2005; Rizzo et al.2006). Stocks represent a balance unit that changes with each time step while flows represent a positive or negative change of flux. The converters represent input parameters and arrows represent mathematical relationships between the elements. STELLA consists of three numerical integration methods namely: Euler (default method), Runge-Kutta 2-order, and Runge-Kutta 4-order. In this study, a Forth Order Runge-Kutta method was used to perform the integration because it estimates changes in stocks by making more forecasts of flows unlike the other two methods. Runge-Kutta 4-order is also known to provide the best results with fairly large tolerances when functional evaluations are simple (Butcher 1964; Shampine & Watts 1971; Hull et al.1972).

4.6.1 Description of the age class- specific elephant model

The model consists of five submodels namely: 1) the elephant population dynamics representing the five subpopulations within Greater Virunga Landscape (Figure 4.3). The elephant population dynamics represents the five subpopulations within Greater Virunga Landscape. The elephant population dynamics submodel is impacted by all the variables in all submodels and act directly by lowering or increasing the mortality and natality of each age class, and total reproduction of the entire population. Reproduction starts in age class 11-30 all the way through 50 years and above. The observed (census) population of 2006 (UWA, ICCN, WCS, & WWF, 2006) was used to assign the number of elephants per age-class in 1960 based on protected area management agency (UWA, ICCN, and ORTPN) reports and peer-reviewed publication (Laws 1970; Eltringham and Malpas, 1973; Hanks and Mctonsh 1973; Moss, 2001). The proportions of each age-class were generated based on scholarly work by savanna elephant specialized scientists as shown in (Table 4.2) (Hanks and Mctonsh 1973; Moss, 2001). Of the total population in 2006, 37% of the elephants observed were considered to be in the age class 0-10, and the rest of the distribution was as follows: 11-30 (30%), 31-40 (18%), 41-50 (12%) and ≥50 years (2%). The year 2006 was treated as the base year and used the observed population in 1960 for model calibration (Table 4.5). Mortality, however, starts from 0-10 years all through the entire age structure (Table 4.5).

Table 4.5 Observed elephant population for the years 1960 and 2006 by age class proportions used in model calibration

Age structure	Population of 2006	Proportion of 1960	Population in 1960
0-10	1240	0.37	660
11-30	998	0.30	531
31-40	599	0.18	319
41-50	398	0.12	212
>50	72	0.02	38
Total	3307		1760

Natality and mortality are density-dependent, but natality is also influenced by precipitation, water, and habitat quality (Figure 4.12). In the empirical mode, A1, A2……A5 represent the age classes shown in Table 4.3). The initial elephant population under A1 to A5 is provided in Table 4.6. The number of elephants in each particular age class for the entire landscape at given time (t) was calculated according to the following equations: $\mathrm{A1(t)} = \mathrm{A1(t-dt)} + (\mathrm{NAT} - \mathrm{A1S} - \mathrm{MOA1}) * \mathrm{dt}$ ………………(eq1)

$\mathrm{A2(t)} = \mathrm{A2(t-dt)} + (\mathrm{SA2} + \mathrm{NIMMA2} - \mathrm{SA2} - \mathrm{M0A2}) * \mathrm{dt}$ ……………………………(eq2)

$\mathrm{A3(t)} = \mathrm{A3(t-dt)} + (\mathrm{SA3} + \mathrm{NIMMA3} - \mathrm{SA3} - \mathrm{M0A3}) * \mathrm{dt}$ ……………………………(eq3)

$\mathrm{A4(t)} = \mathrm{A4(t-dt)} + (\mathrm{SA4} + \mathrm{NIMMA4} - \mathrm{SA4} - \mathrm{M0A4}) * \mathrm{dt}$ ……………………………(eq4)

$\mathrm{A5(t)} = \mathrm{A5(t-dt)} + (\mathrm{SA5} + \mathrm{NIMMA5} - \mathrm{SA5} - \mathrm{M0A5}) * \mathrm{dt}$ ……………………………(eq5)

where, t = time, NAT = Natality (elephants/km^2/year), SA1….SA5 is surviving animals in each age classes, NIMMA2, NIMMA3,…..NIMMA5 is number of elephants immigrating in each age class; MOA1, MOA2…..MOA5 is mortality (elephants/km^2/year),

The total annual reproduction (TOPR) for the entire population was 40 percent and the natality rate index was modeled as a graphic function of the entire observed population and its values were in the range of 1- 2.5 for the period 1960-2010. The TOPR was adjusted (adj) according to the maximum natality rate (MNR). $MNR = TOPR/MNR$…………………………………(eq6)

and MNR was derived as $MNR = MNRA2 + \mathrm{MNRA3} + \mathrm{MNRA4}$………………………………(eq7)

where, MNRA2, MNRA3, MNRA4 are the maximum natality rates for the respective age classes.

$$\text{adjMTR} = \text{adjMNRA2} + \text{adjMNRA3} + \text{adjMNRA4} \quad \text{...(eq8)}$$

where, adjMTR = Adjusted maximum total reproduction, adjMNRA2, adjMNRA3, and adjMNRA4 are the adjusted maximum reproduction rate for each age class.

Survival rate of elephants in each age class (SA1, SA2.....SA5) were compiled as follow:

$$SA1 = (A1 - MOA1) * 0.1 \text{..(eq9)}$$

$$SA2 = (A2 - MOA2) * 0.05 \text{...(eq10)}$$

$$SA3 = (A3 - MOA3) * 0.05 \text{...(eq11)}$$

$$SA4 = (A4 - MOA4) * 0.1 \text{...(eq12)}$$

$$SA5 = (A5 - MOA5) * 0.1 \text{...(eq13)}$$

Age class A2 and A3 were assigned a slightly a lower survival rate because the elephants in this age group are more susceptible to poaching and kills by humans in reiteration for lost human lives or physical injury and crop loss. This is also the age bracket when sub-adult males abandon their social groups in search of mates (Moss 2001). It was reported that the survival probabilities for females were slightly higher (89%) than for the males (82%) (Moss, 2001). According to Wittemyer et al. (2013), the annual population growth, including immigration and emigration averaged 0.17% and 2.8% when migration is excluded, over a period of 14 years for the Samburu National park (Kenya) population. The mean annual mortality for this population recorded was 4.71% and a maximum of 14.1 percent while the mean annual natality was 7.21% (maximum 14.4% and minimum 2.1%).

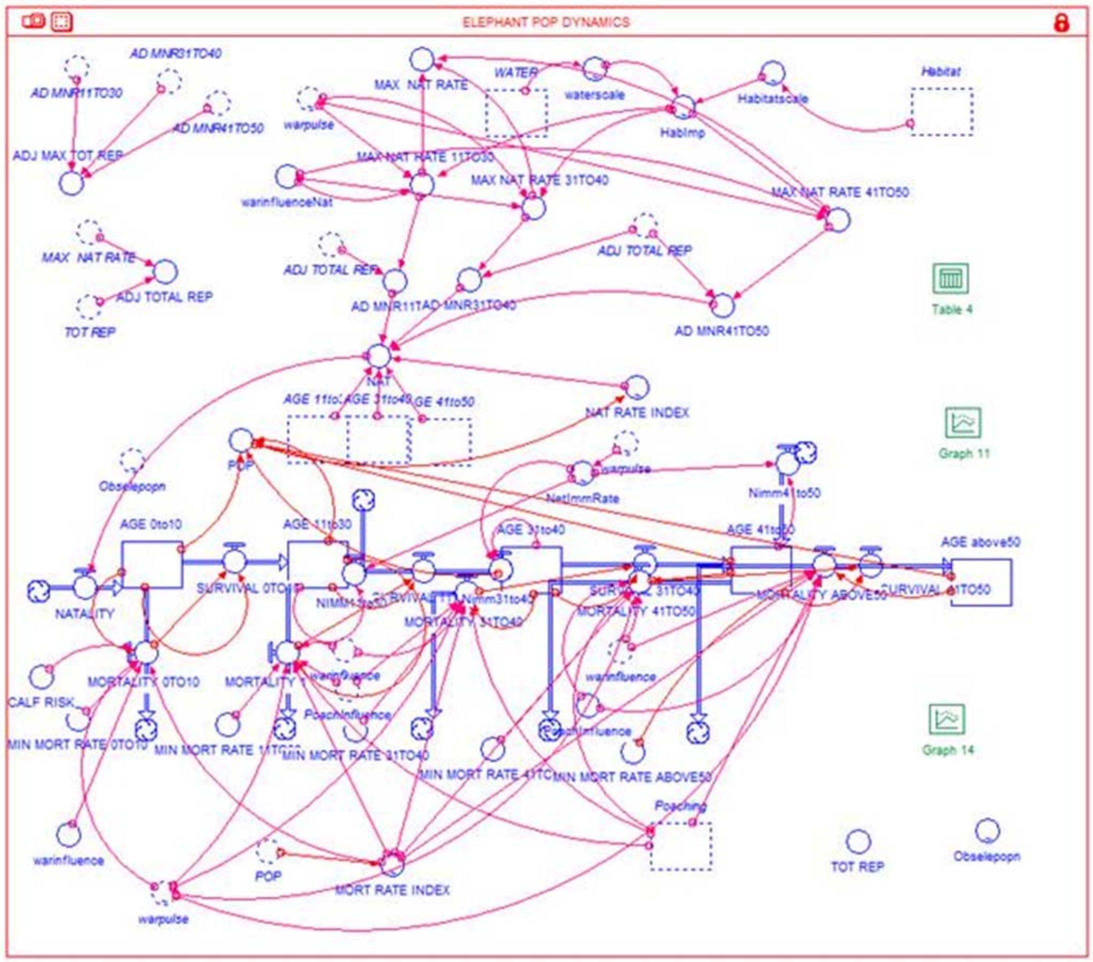


Figure 4.15 The STELLA model showing the elephant Age structure, habitat and water interaction, and climate change effects on their entire ecosystem

Mortality rates for each age class were empirically calculated as per the equations below:

$MOA1 = (MMRA1 * A1 * MRI * Z) * (1 + WP * WI)$……………………………………..eq14

$\mathrm{MOA2} = (\mathrm{MMRA2} * \mathrm{A2} * \mathrm{MRI}) + (1 + \mathrm{P} * \mathrm{PI})) + (1 + \mathrm{WP} * \mathrm{WI}))$……………………….(eq15)

$\mathrm{MOA3} = (\mathrm{MMRA3} * \mathrm{A3} * \mathrm{MRI}) + (1 + \mathrm{P} * \mathrm{PI})) + (1 + \mathrm{WP} * \mathrm{WI}))$……………………….(eq16)

$\mathrm{MOA4} = (\mathrm{MMRA4} * \mathrm{A4} * \mathrm{MRI}) + (1 + \mathrm{P} * \mathrm{PI})) + (1 + \mathrm{WP} * \mathrm{WI}))$……………………….(eq15)

$\mathrm{MOA5} = (\mathrm{MMRA5} * \mathrm{A5} * \mathrm{MRI}) + (1 + \mathrm{P} * \mathrm{PI})) + (1 + \mathrm{WP} * \mathrm{WI}))$……………………….(eq15) where, MMRA1, MMRA2,……MMRA5 are the Minimum mortality rates for the respective age classes, MRI is the Mortality rate index, *C* is the calf risk, *WP* is the war impulse, and *WI* is the war influence. The war impulse is modeled as a function of time and intensity (Figure 4.xxxx) while war influence was assigned a factor value of 16 in relation to the impact it exerts on elephant population.

4.6.2 Model calibration and sensitivity analysis

Proper model calibration is important in ecological modeling studies to reduce uncertainty in model simulations (Moriasi et al., 2007) and involves a sensitivity analysis followed by a manual or automatic calibration. The most ideal calibration techniques are heavily discussed by many scholars (Boyle, Gupta, & Sorooshian, 2000; Legates & McCabe, 1999). The model was calibrated using the 1960 elephant population census data and then simulated to the observed elephant population data from 1960-2010 through manually changing model input parameter values to produce simulated values that are within a certain range of observed elephant population. In order to allow for balancing the trade-offs in the ability of the model to simulate various aspects of the observed population, each submodel parameters in STELLA were being tracked while adjustments were made, recognizing potential errors associated with the observed data itself. For example, errors associated with low detection during census were highly thought about, but no effort was made to consider this issue in our model. An assumption was made that for each census year, the detection error was uniform and not a critical factor in explaining the observed data variability. In addition, no attempt was made to address the stochastic behavior of the elephant population because of the complexity that is involved in doing so. Automatic calibration methods that involve the computation of the prediction error using an equation (objective function) and an automatic optimization procedure using an algorithm to search for the perimeter values that optimize the value of the objective function are highly recommended (Gupta et al.199). In this case, a regression equation of the 4-order polynomial type, an equivalent to automation procedure was only used in filling up the elephant census data gaps for 18 years without census data (accounting for 35% of no data) (Figure 4.16).

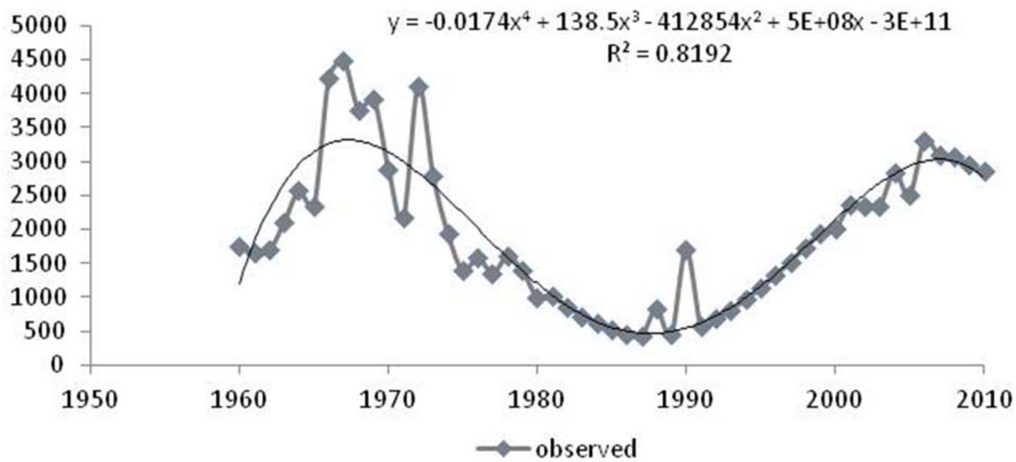


Figure 4.16 Observe elephant population fitted with a 4-order polynomial regression equation

The simulated population was plotted and compared with the observed population (Figure 4.17) to visually inspect the performance of the model toward simulating the observed population. Furthermore, a scatter plot fitted with a linear regression and the coefficient of determination (R^2) was used to assess how well the model explained the observed data (Figure 4.18). A high R^2 = 0.78 showed that the model explained very well the observed data, providing a high degree of confidence in the model.

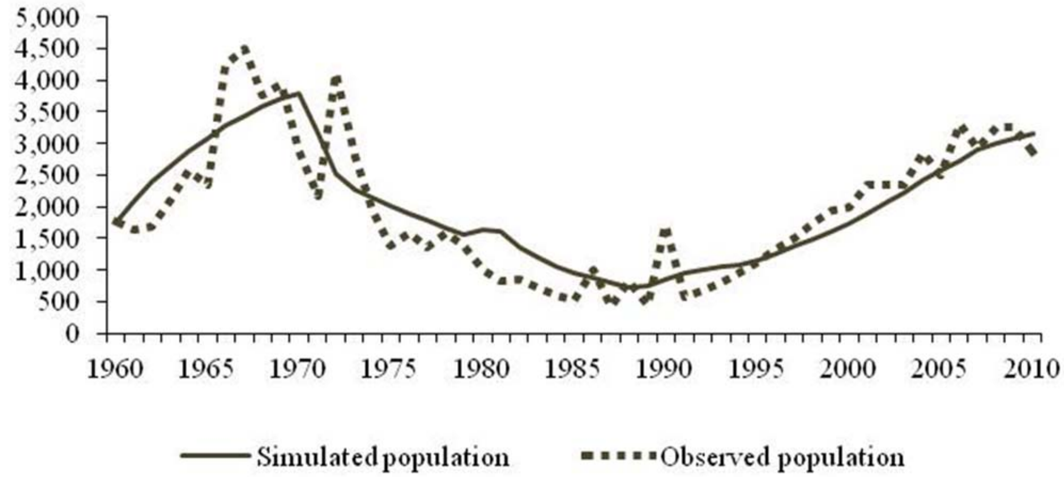


Figure 4.17 Observed and simulated elephant population trend over 51 years in GVL

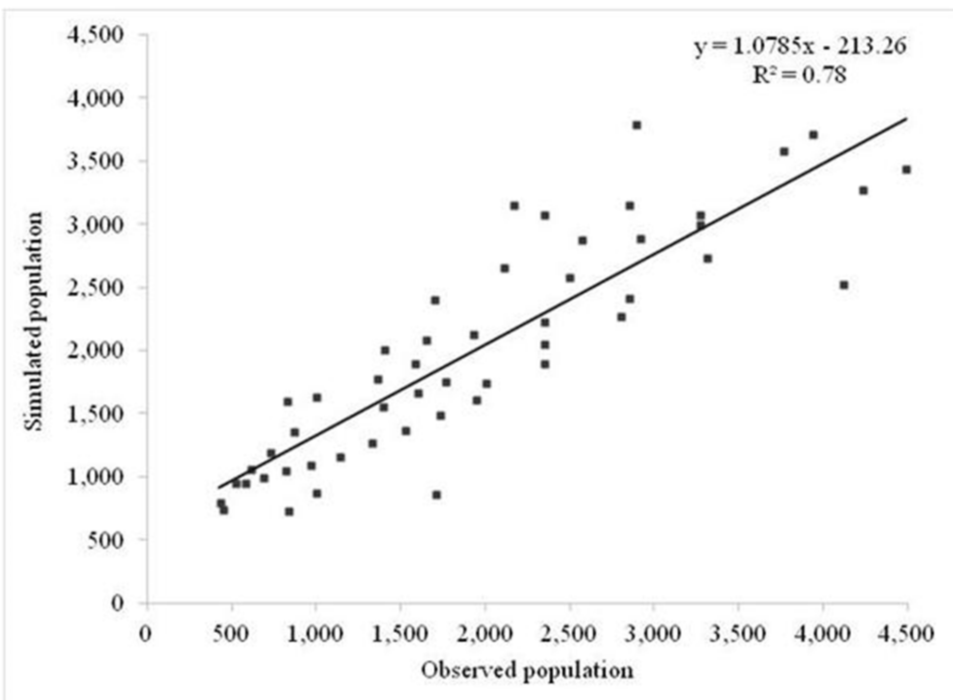


Figure 4.18 Fitted linear regression on simulated against observed elephant population for GVL elephant population after model calibration

4.6.3 Model validation

The performance of the model was validation was done using observed data from QENP (Uganda), a savanna park located within the GVL (Figure 4.19). Forty nine years of census data for this park was accessed from Uganda Wildlife Authority and thoroughly cross-checked with the reports from Uganda National Parks currently housed by the Ministry of Tourism and Antiquities, and corroborated with census figures published in peer reviewed journal mainly the African Journal of Ecology. The observed and simulated populations are well explained by the model (R^2 = 0.68) and the trend was well represented by the line graph (Figure 4.20).

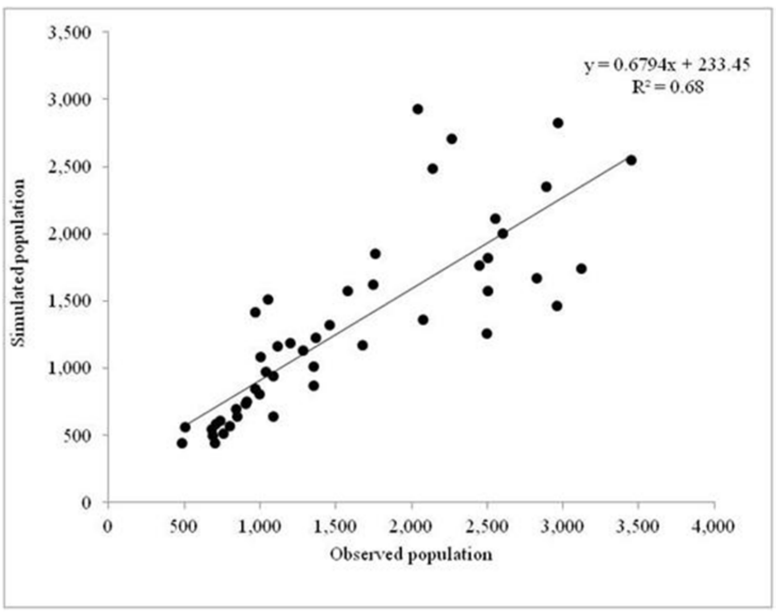


Figure 4.19 Scatter plot of observed and simulated elephant population for QENP

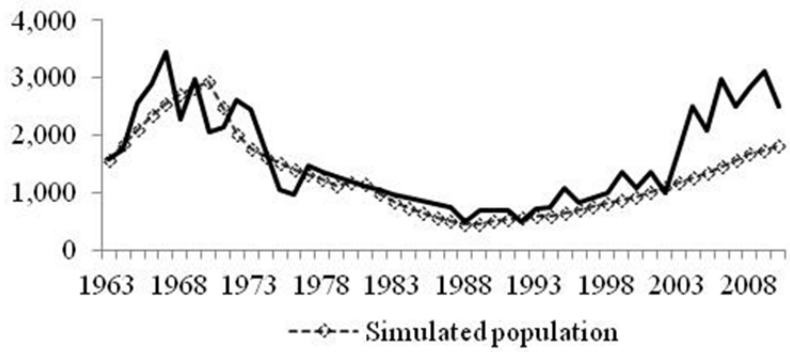


Figure 4.20 QENP Observed and simulated elephant population based on calibrated GVL model

Model performance validation was further assessed by computing the Nash-Sutcliffe efficiency (NSE). Nash-Sutcliffe efficiency (NSE) defined by (Nash & Sutcliffe, 1970) and mean squared error (MSE) are the most commonly used criteria for calibration and evaluation of hydrological models with observed data (Gupta, Kling, Yilmaz, & Martinez, 2009; Moriasi et al., 2007; Van Liew, Veith, Bosch, & Arnold, 2007). The Nash-Sutcliffe efficiency (NSE) is a normalized statistic that determines the relative magnitude of the residual variance ("noise") compared to the measured data variance ("information") (Moriasi et al., 2007; Nash & Sutcliffe, 1970). NSE indicates how well the plot of observed versus simulated data fits the 1:1 line.
The Nash and Sutcliffe model efficiency coefficient has been used to assess hydrological model performance in Africa's watershed (Kileshye Onema, Taigbenu, & Ndiritu, 2012). In particular, (Kileshye Onema et al., 2012) used Nash Sutcliffe Efficiency (NSE), Percent bias (PBIAS) and root mean square error to the standard deviation ratio (RSR) predict the long-term monthly mean discharges for the two groups of subcatchments in Semliki River, a part of the upper Nile drainage basin located in the study area. The Nash and Sutcliffe model efficiency coefficient was used to assess the predictive power of the model.

It is defined as:

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n}(Qobs - Qsim)2}{\sum_{i=1}^{n}(Qobs - Qm)\,2}\right]$$

where Q_{obs} is the ith observation for the constituent being evaluated, Q_{sim} is the ith simulated value for the constituent being evaluated, Q_m is the mean of observed data for the constituent being evaluated, and n is the total number of observations. NSE ranges between $-\infty$ and 1.0 (1 inclusive), with NSE = 1, corresponds to a perfect match of modeled to the observed data and a NSE = 0, indicates that the model predictions are as accurate as the mean of the observed data, and Inf < NSE < 0, indicates that the observed mean is better predictor than the model. NSE was selected for this test because of two important reasons: 1) it is widely used in hydrologic modeling and recommended (Gupta et al., 2009; Legates & McCabe, 1999; Singh, Knapp, Arnold, & Demissie, 2005). In particular, Legates & McCabe (1999) noted that a modified NSE is less sensitive to high extreme values due to the squared differences. The modified version, however, was not selected because of its limited use. In this study, the Nash-Sutcliffe efficiency (NSE) value for the simulation model is 0.769162 and for the model performance validation, the NSE is 0.457571. Given the magnitude of the NSE values for the observed elephant population and the QENP elephant population datasets that were used for model calibration and evaluation, the model predicted well the observed population changes in GVL over the last forty decades.

4.6.4 Model evaluation statistics

Several error indices are commonly used in model evaluation and these include mean absolute error (MAE), mean square error (MSE), and root mean square error (RMSE). These indices are valuable because they indicate error in the units (or squared units) of the constituent of interest, which aids in analysis of the results. RMSE, MAE, and MSE values of 0 indicate a perfect fit. Singh et al. (2004) state that RMSE and MAE values less than half the standard deviation of the measured data may be considered low and that either is appropriate for model evaluation. A standardized version of the RMSE was selected for recommendation and is described later in this section. Percent bias (PBIAS) measures the average tendency of the simulated data to be larger or smaller than their observed counterparts (Gupta et al., 1999). The optimal value of PBIAS is 0.0, with low-magnitude values indicating accurate model simulation. Positive values indicate model underestimation bias, and negative values indicate model overestimation bias (Gupta et al., 1999). PBIAS was calculated following the equation below:

$$PBIAS = \left[\frac{\sum_{i=1}^{n}(Qobs - Qsim)}{\sum_{i=1}^{n}(Qobs)} \times (100)\right]$$

where pBIAS is the deviation of data being evaluated, expressed as a percentage. Percent population error (PPE) (Singh et al., 2004), prediction error (Fernandez et al., 2005), and percent deviation of constituents are calculated in a similar manner as pBIAS. The deviation term (D_V) is used to evaluate the accumulation of differences in population between simulated and measured data for a particular period of analysis. pBIAS was selected for use because it is easy to compute, but it has been widely used (ASCE 1993) and pBIAS has the ability to clearly indicate poor model performance (Gupta et al., 2009).

RMSE-observations standard deviation ratio (RSR), a model evaluation statistic is an improvement to the most commonly used RMSE error index statistics (Chu & Shirmohammadi, 2004; Kileshye Onema et al., 2012; Moriasi et al., 2007). RSR standardizes RMSE using the observations standard deviation and it combines both an error index and the additional information recommended by (Legates & McCabe, 1999). RSR is calculated as the ratio of the

RMSE and standard deviation of measured data and is known to incorporate the benefits of error index statistics and includes a scaling/normalization factor so that the resulting statistic and reported values can apply to various constituents (Gupta et al., 2009; Moriasi et al., 2007). RSR varies from the optimal value of 0, which indicates zero RMSE or residual variation and therefore perfect model simulation, to a large positive value. Lower values of RSR and RMSE signify a better performance for the model to simulate the phenomenon under investigation.

RSR was calculated as shown in the equation below:

$$RSR = \frac{RMSE}{STDEV_{obs}} = \left[\frac{\left[\sqrt{\sum_i^n (Q_{obs} - Q_{sim})2}\right]}{\left[\sqrt{\sum_i^n (Q_{obs} - Q_{mean})2}\right]}\right]$$

Following the above equation, the computed PBIAS for the simulation model was: PBIAS= -2.86437; and the Root Mean Square Error-observations standard deviation ratio (RSR) = 0.480456, and for the validation model was PBIAS=17.2055(0.0172); RSR=0.736498

Four different scenarios with respect to the anthropogenic factors were assessed namely: 1) the impact of war when frequency and magnitude is varied as per the changes provided in (Table 4.6); 2) a reduction in poaching under three policy options, that is, a uniform increase in law enforcement, monitoring and household incomes by ten percent while maintaining all the other stressors;3) the impact climate changes, specifically precipitation and temperature under three different RCP scenarios changing water volume and habitat; and 4) and habitat change

Table 4.6 Scenarios and parameters selected for model simulation over a period of 51 years

Scenarios	Description	Parameter values		
		Low	Medium	High
Change in wars	Magnitude	25%		50%
	Frequency	25%		50%
Influence of Poaching	Law enforcement	5% increase		10% increase
	Monitoring	5% increase		10% increase
	Income	5% increase		10% increase
Climate change according to IPCC AR5 RCP2.6, RCP6.0, & RCP8.5 scenarios	Temperature	1.6 ^{o}C	2.8 ^{o}C	4.3 ^{o}C
	Precipitation	2% increase	10%	18%
Habitat change	Increase in forest & savannas	50%		
Water resources	Increase	Determined by the 3 RCP scenarios		

To assess the effect of habitat quality on age-specific elephant population dynamics, three different scenarios of habitat loss (i.e. low, medium, and high), particularly savanna grassland, and forests/woodlands conducted over a period of 20 years using a monthly simulation scale for each year. Similar simulation experiments were conducted for war, poaching, climate change and water resources, and poverty scenario while assessing the changes in age-specific dynamics associated with each experiment (Table 4.6). Quantitative models of the kind presented here provide a valuable tool for exploring the consequences of management decisions involving the manipulation of habitats and watershed ecosystems to achieve viable elephant population densities.

4.7 Results

This section provides the results of the various analyses performed coupled with an attempt to explain them. In terms of structure, results of the baseline conditions are presented first, followed by different scenario analyses done.

4.7.1 Baseline results

The baseline results from the calibrated model (Figure 4. 21) shows that the total elephant population was at its peak in the early 1960s, declined tremendously in the 1990s steadily rising in the 2000s. In terms of individual age specific classes, baseline results indicate that age classes 0-10, 11-30, and 31-40 years are recovering from the dramatic decline that occurred in the mi 1990s while the age classes 41-50 and >50 years completed phased out during the same catastrophic period. The declining trend that started in the 1970s all the way to early 1990s is attributed to mainly anthropogenic factors such as war, and poaching as opposed to density dependent factors. As discussed earlier in this chapter under anthropogenic impacts section, poachers selectively harvest elephants with large tasks, which are also the adult elephants. Baseline results also show that the number of elephants in age classes 0-10, and 11-30 are the highest or greatest of across the entire modeling period (1960-2010). For the sake's argument, if these age classes were equally selected by poachers, then the trend in their numbers would have behaved in the same way as the seniors. On the contrary, it is reasonable to suggest that the declining trend would not be expect to behave the same way given that a large proportion of the elephants at the baseline level is high among these classes compared to the other classes. This argument is weak because adult elephants are expected to avoid death risk much better than the juveNiles and calves. In fact, the calf risk associated with war, fires, and poaching in some African populations have been reported to be very high.

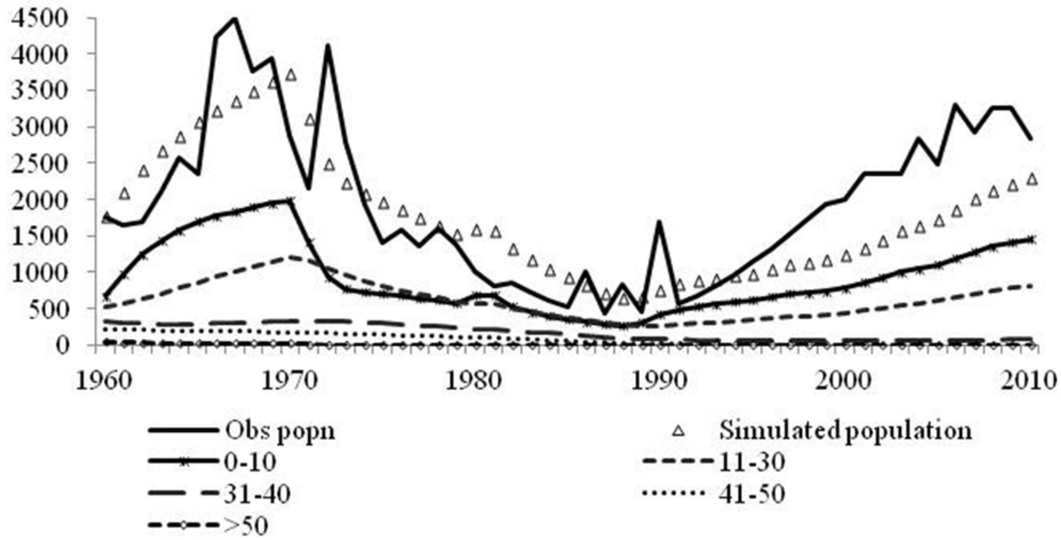


Figure 4.21 Baseline results for the elephant population dynamics

4.7.2 Scenario modeling results

Results from different simulation analyses implemented are presented in this section starting with the climate change effects on total elephant population and age class-specific population dynamics,

4.7.2.1 Climate change impacts

The results of the climate change analyses based on the IPCC RCP2.6, RCP6.0, and RCP8.5 scenario showed that elephants in the age classes 41-50, and >50 years would have been eliminated from the population (Figures 4.22, Figure 4.23, and Figure 4.24). A small number of elephants in the age groups 31-40, and 41-50 would survive under these climatic conditions.

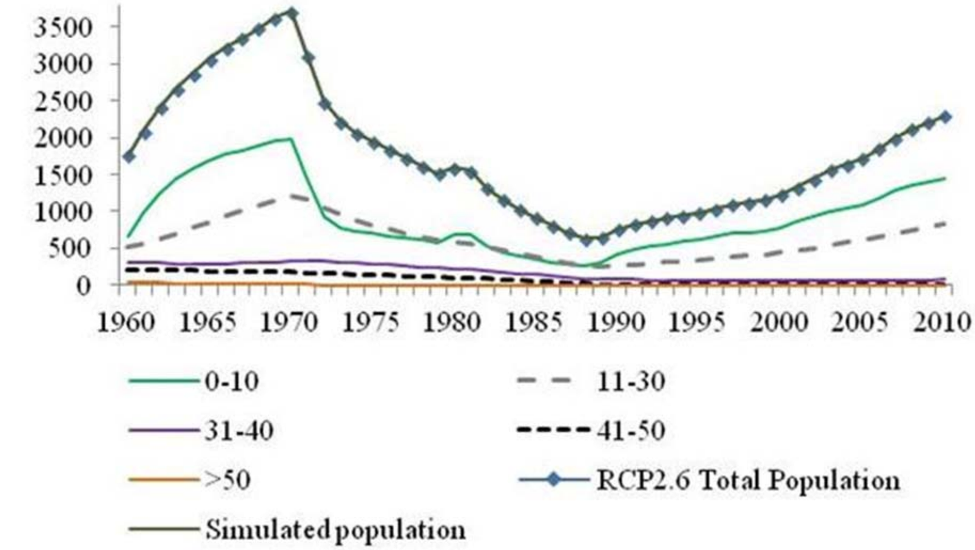


Figure 4.22 Age class specific elephant population dynamics under RCP2.6 future climate change scenario

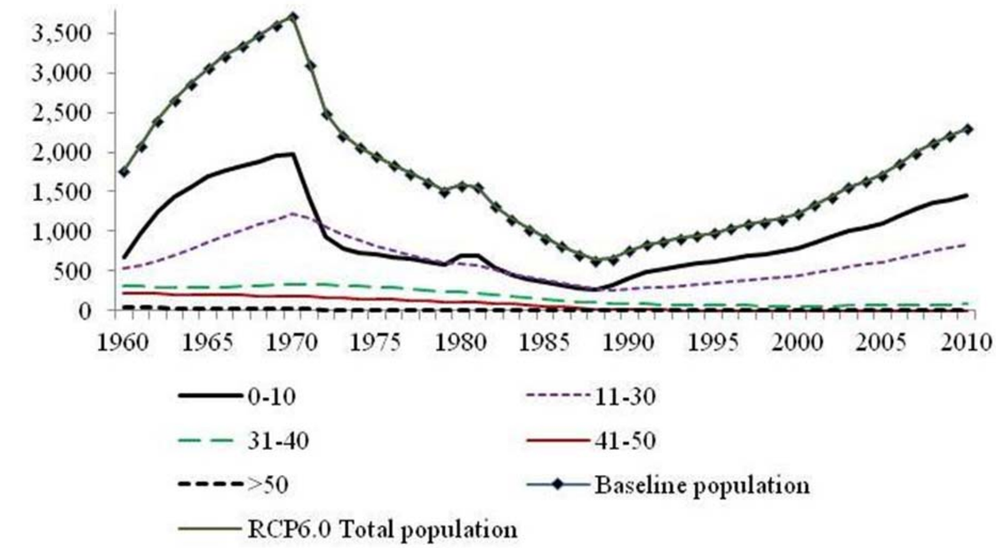


Figure 4.23 Age class specific elephant population dynamics under RCP6.0 future climate change scenario

The numbers of elephants in the age group 0-10, and 11-30 showed an initial increase followed by a sharp decline from late 1970s down to 1990s and in the late 1980s, the numbers in these classes begin to rise again. This increase in the numbers could be attributed to immigrations elephants with a reasonable number of calves. The total population of elephants resulting from the different RCP scenarios was the same (Figure 4.25). Climate change is a slow acting environmental process whose effects take years to exert their influence on the species or system.

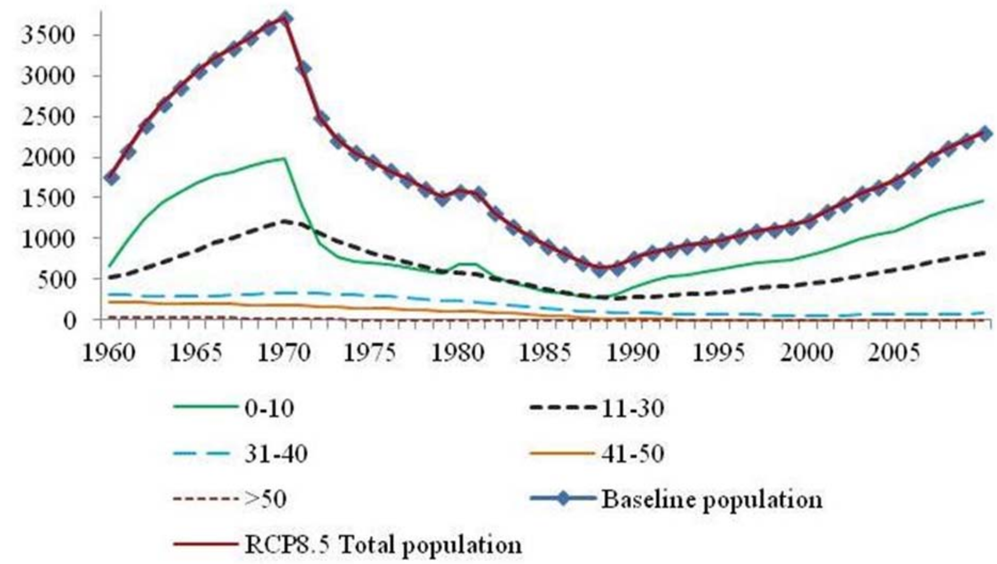


Figure 4.24 Age class specific elephant population dynamics under RCP8.5 future climate change scenario

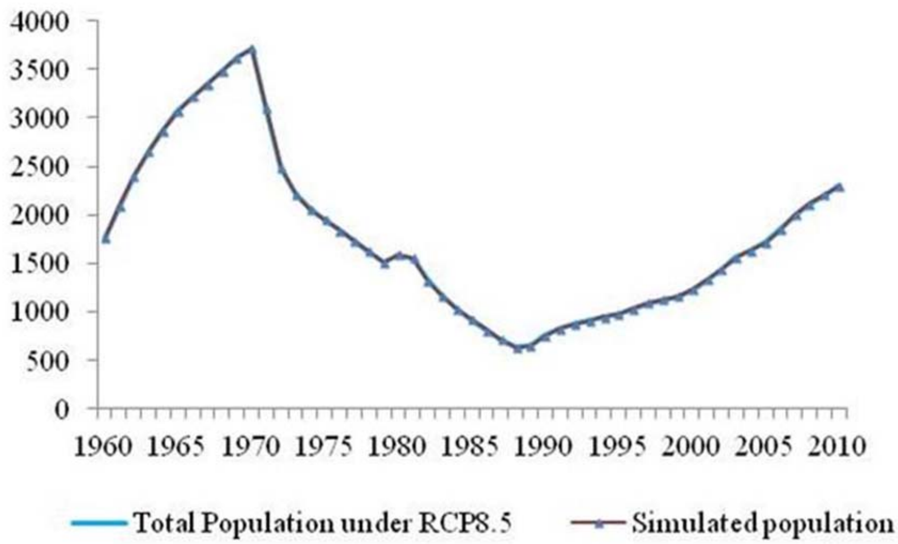


Figure 4.25 Impact of climate change on elephant population dynamics

Once the impacts of climate change become eminent, they cause long term effects, which conservationists must plan for early to avoid disastrous event. This, however, demands that the scales of conservation be aligned with the scales of climate-change projections to develop management strategies that enhance the resilience of the ecosystems. Greater Virunga landscape is characterized by spatially varied climatic conditions attributed to the diversity in of ecosystem types, vegetation, and topography. These conditions enhance the elephants to deal with local-level climatic changes through switching of habitats from savanna to forests and wetlands during extreme drought conditions and move back to savannas during the wet seasons. According to Wiens and Bachelet (2009), local scale variations mentioned above, often override the projections of broad-scale climate models resulting high uncertainty.

Mann–Whitney U test (also called the Mann–Whitney–Wilcoxon (MWW), Wilcoxon rank-sum test, or Wilcoxon–Mann–Whitney test) is a nonparametric test of the null hypothesis that two populations are the same against an alternative hypothesis (Wilcoxon 1945; Mann and

Whitney, 1947; Cheung et al. 1997; Fay and Proschan 2010). The same test was applied to the test the impact of climate change under the extreme representative concentration pathway scenario 8.50 and the results did not show a significantly result in greater population dynamics compared to the baseline simulated population response (RCP8.5>B0, $p<0.5$; $W = 1300.5$, $P = 0.5$). This confirm the results observed by examing the graphic population trend presented under the RCP8.5 scenario with the baseline, that actually climate change impact did not show such dramatic change in the elephant population dynamics.

4.7.2.2 Habitat change

Habitat change in this landscape is mainly driven by regular fires inside the savanna parks, agricultural expansion, particularly commercial plantation crops such as tea, sugarcane, tobacco, palm oil, and cocoa. Elephants spend nearly 80% of their time in savanna woodlands and grasslands, only moving to forest areas during the dry season to feed on tree leaves and in search of water and salt leaks (Laws, 1970; Wing and Buss, 1970; Laws and Parker, 1975). They also frequently move to the wetlands in search of water. In order to investigate the impact of a direct increase in the suitable habitat for elephants, a fifty percent increase in the proportion of woody vegetation (i.e. forest and savannas) was implemented. The results of this experiment resulted in an increase in the populations of all age classes (Figure 4.26). The increase in population for age classes 41-50, and >50 years was small, declining to zero in the year 1990.

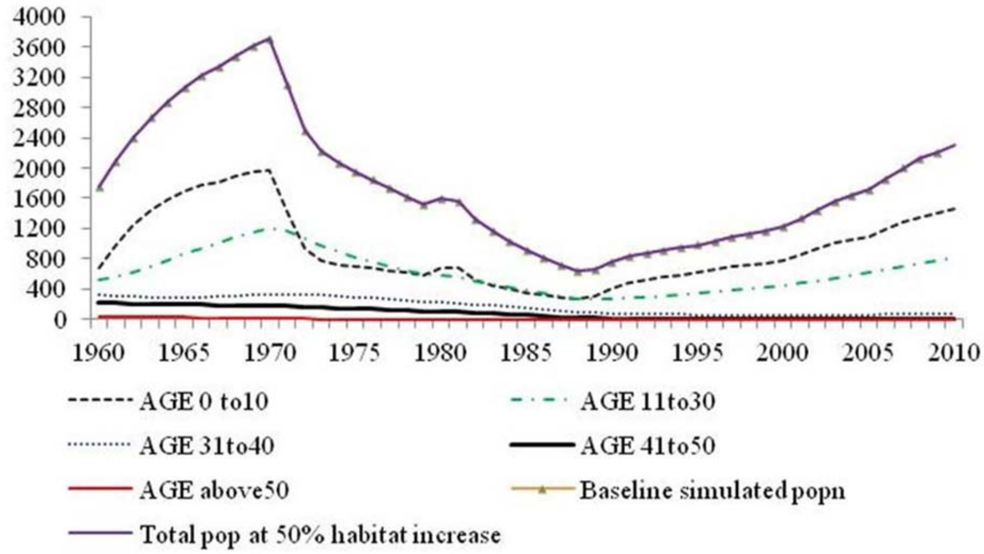
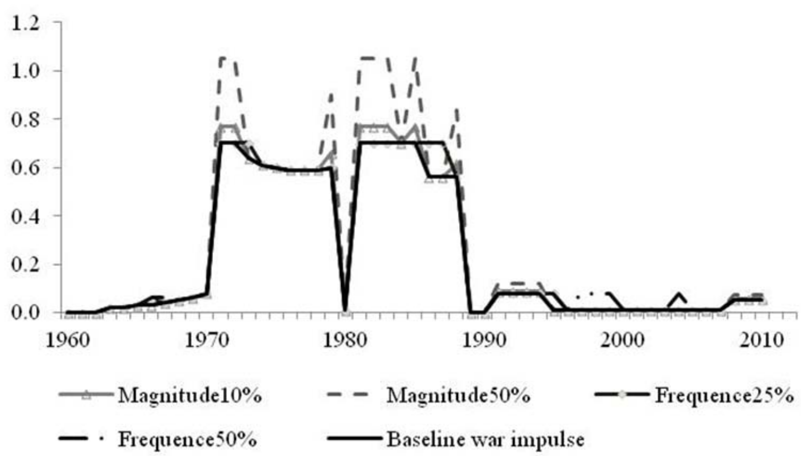


Figure 4.26 Impact of doubling habitat on elephant population by age class

4.7.2.3 War effect

The impact of war was examined at two different levels, that is, the frequency and magnitude of wars. Both frequency and magnitude of war were varied from the baseline by 25%, and 50%. Time lag impact of war was considered, but this analysis would only result in a shift in population demographics to the while the left, which didn't add any value to the



storyline.

Figure 4.27 Baseline war impulse varied by severity and frequency of 25% and 50%

The impact of war on elephant populations when the frequency of occurrence is increased by 25%, and 50% did not show much differences in terms of total population dynamics, but exhibited some reasonable differences in the age class specific populations, particularly 41-50, and >50 age classes (Figure 4.28). Similar pattern of results was shown at age class specific level when war frequency was increased by 50% (Figure 4.29), an increase in magnitude by 25% above the baseline was applied and the results showed that elephants in the age groups 41-50, and > 50 year would decline tremendously, and by the year end of the 1980s, they would be completely purged out of the population. Similarly, when the magnitude of war was increased by 50% above the baseline, the response in terms of age class specific population was the same.

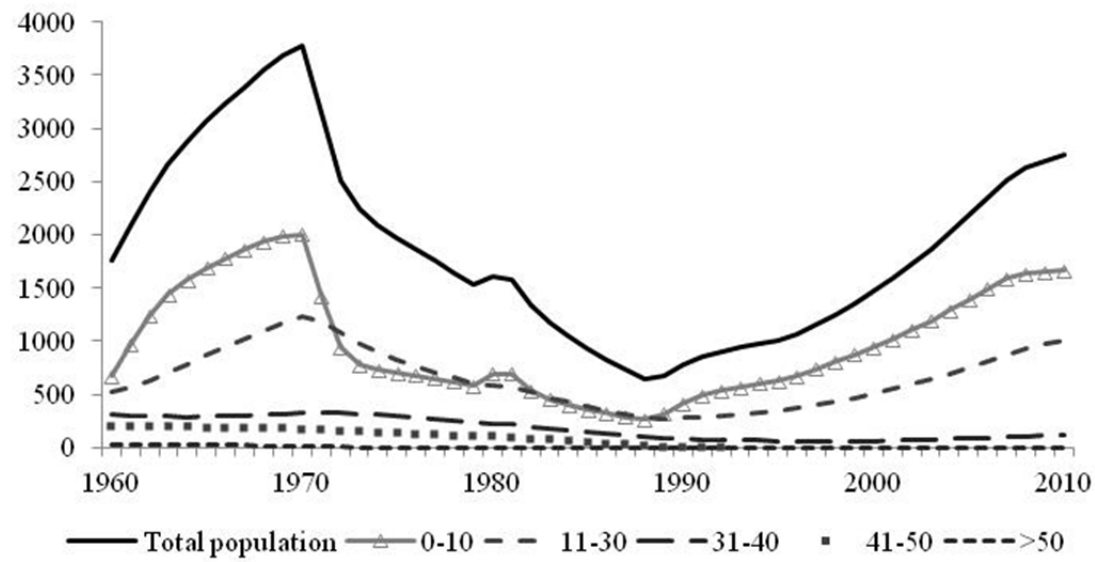


Figure 4.28 Impact of war on elephant population dynamics when frequency of occurrence is increased by 25%

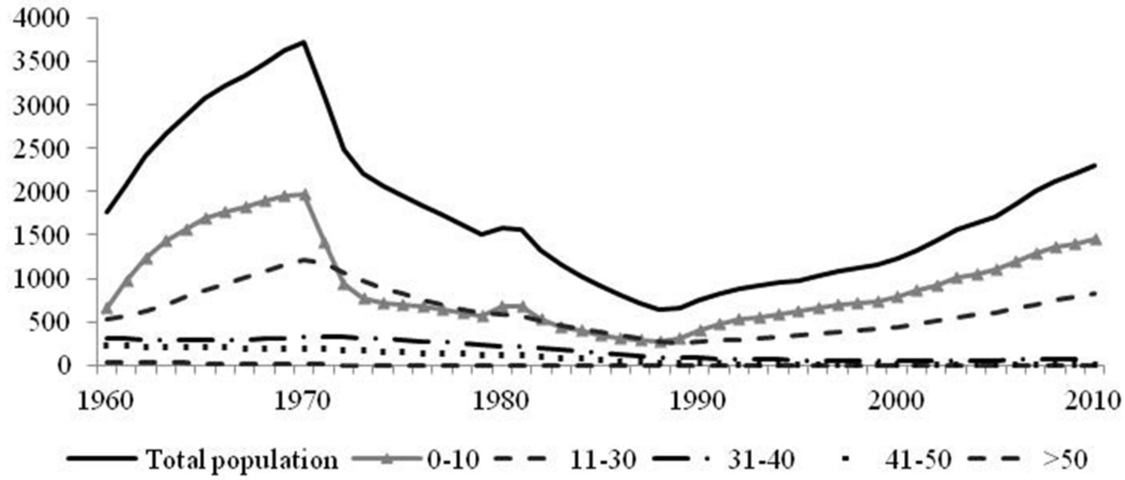


Figure 4.29 Impact of war on elephant population dynamics when frequency of war occurrence is increased by 50%

War could occur at high or low frequency, but its magnitude may impact the ecological system or species population differently. In order to understand the differences in frequency and magnitude of war impact on elephant populations, the magnitude of war, a measure of intensity or severity on the population was assessed and the results are presented in figures 4.30, and 4.31.

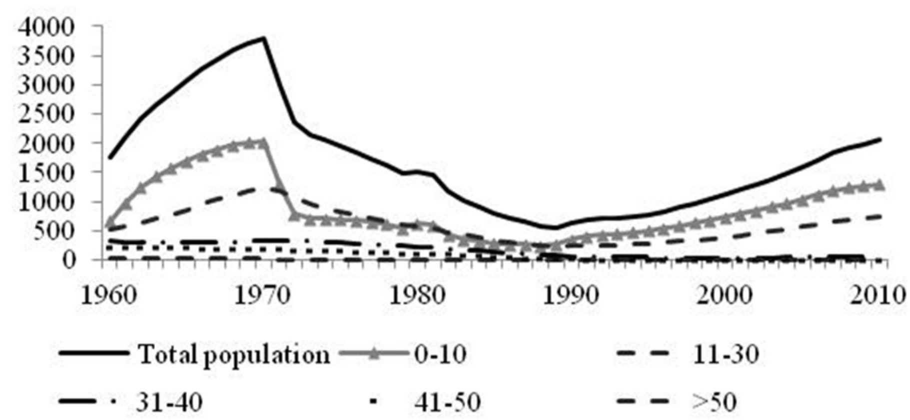


Figure 4.30 Impact of increasing the magnitude of war by 25% on elephant population dynamics

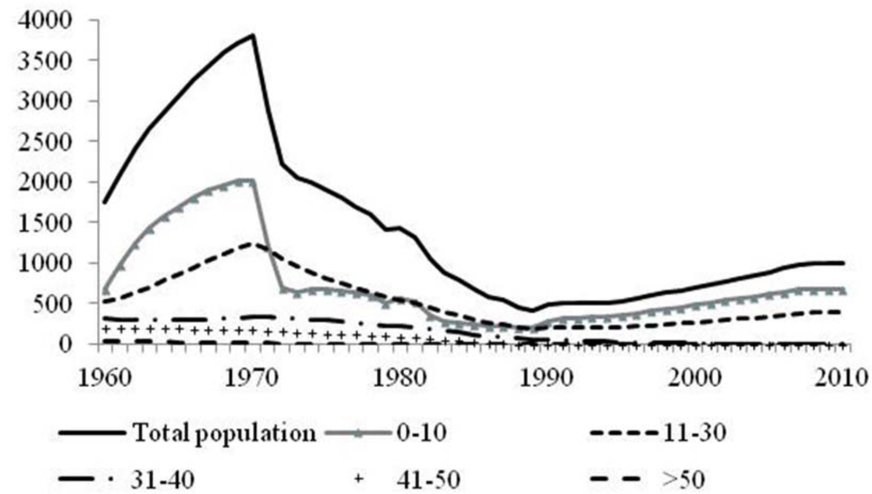


Figure 4.31 Impact of increasing the magnitude of war by 25% on elephant population dynamics

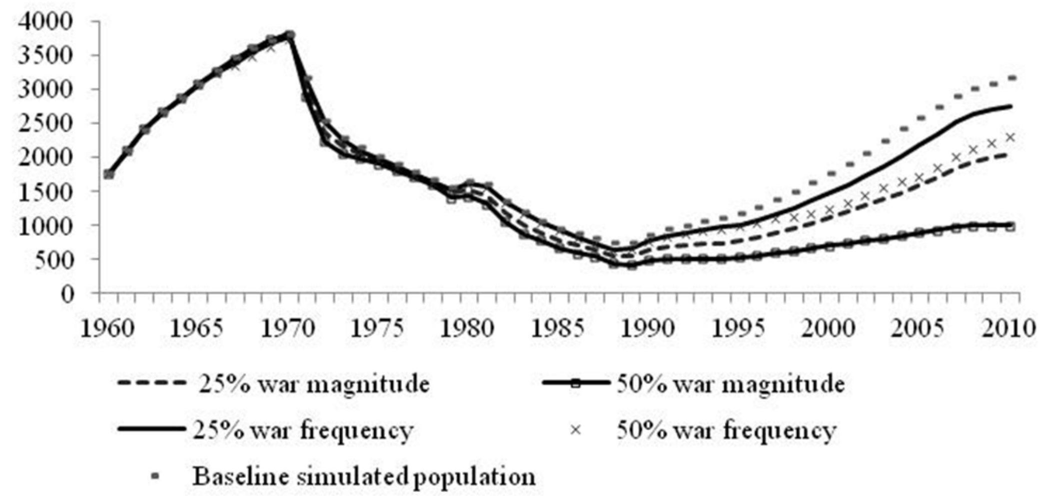


Figure 4.32 Total elephant population dynamics under varied war frequency and magnitude

The Mann Whitney Wilcoxon rank sum statistic was used to test if there was a significant difference s between the population response to a 25%, and 50%increase in frequency of war. Using R software, the Mann Whitney Wilcoxon rank sum test was computed. Before conducting the test, the null hypothesis was stated that the baseline elephant population was less than the simulated population when frequency of war increased by 25%. Mathematically represented as

H0: B0<B25, p<0.05, and the alternative, H1: B0>B25. The results of the Wilcoxon-Mann-Whitney showed that there was no significant difference in elephant population of the baseline and population when war frequency was increased by 25% (W = 1419, p-value = 0.7861). Similarly, an increase in war frequency by 50% didn't result in a significant difference in the baseline and the simulated population (W = 1510, p-value = 0.9201). The increase in the severity of war by 25% and 50%, however, resulted in a significant differences between the baseline population and the simulated population (W = 1605.5, p-value = 0.0261) and (W = 1825.5, p-value = 0.0002) respectively.

4.7.2.4 Poaching impact

Poaching directly removes an individual animal from the population resulting in severe population structure distortion, particularly when the species numbers are very low, and the species's reproduction rate is very low. Poachers normally target mature individuals with well-developed tasks. By removing the adult elephants, poaching affects the age structure and sex ration of the population. Two policy experiments were performed: 1) targeting to reduce the poaching by increasing law enforcement, monitoring and research, and increasing incomes of households living adjacent to conservation by five percent respectively; 2) the second policy option was to reduce poaching by increasing all the above mentioned intervention by 10% each (Figure 4.33). Monitoring several wildlife species across a wide area can be very costly in terms of time, human and financial resources (Field et al.,2005; Gompper et al., 2006; Long et al., 2007; Treves et al. 2010). Without monitoring and research, however, wildlife managers are unable to make the right management strategies. In this study we envisaged that increasing monitoring and research, law enforcement, and raising incomes of rural communities who are hired to poach because of economic vulnerability, can adequately result in an increase in elephant populations (Figures 4.34 to 4.36). Under the two policy options, the age class specific elephant populations, the numbers of elephants in age classes increased drastically except for two very mature age classes. The observed increase was also true for the total population.

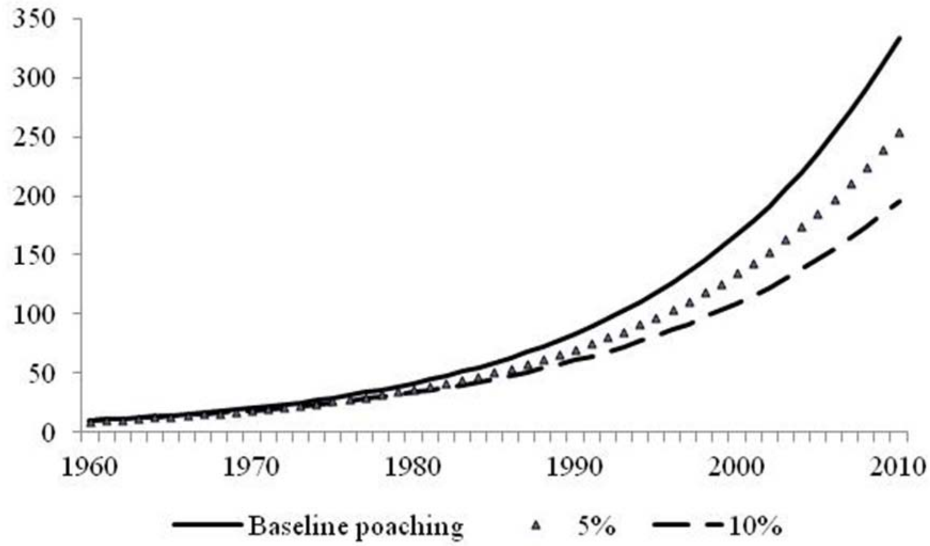


Figure 4.33 Poaching reduction policies compared with baseline poaching level

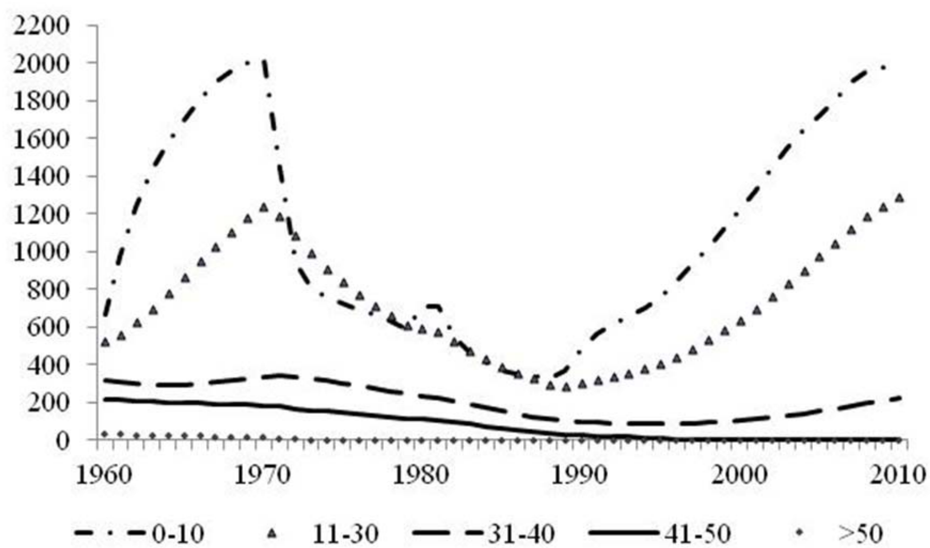


Figure 4.34 Age specific population dynamics under a five percent reduction policy where law enforcement, income, and monitoring interventions are increased by 5% each above the baseline

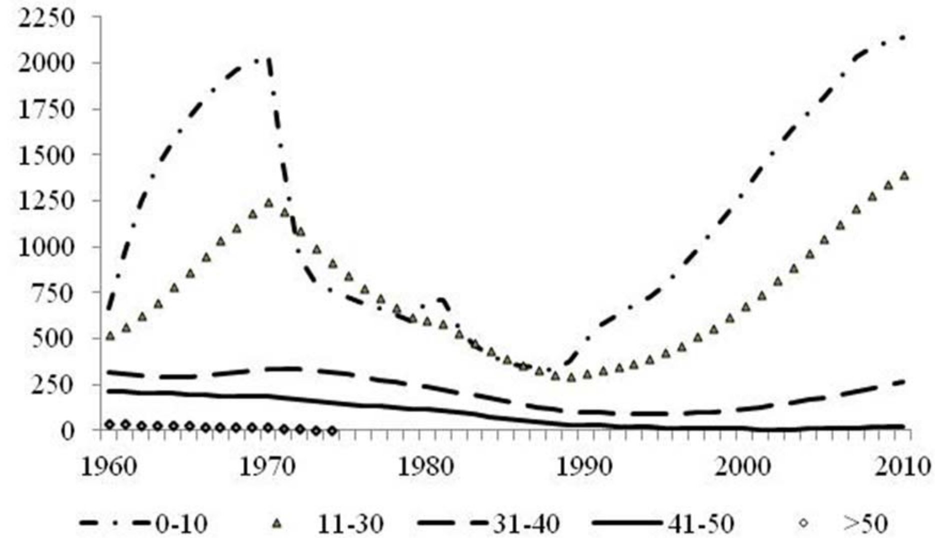


Figure 4.35 Age specific population dynamics under a five percent reduction policy where law enforcement, income, and monitoring interventions are increased by 10% each above the baseline

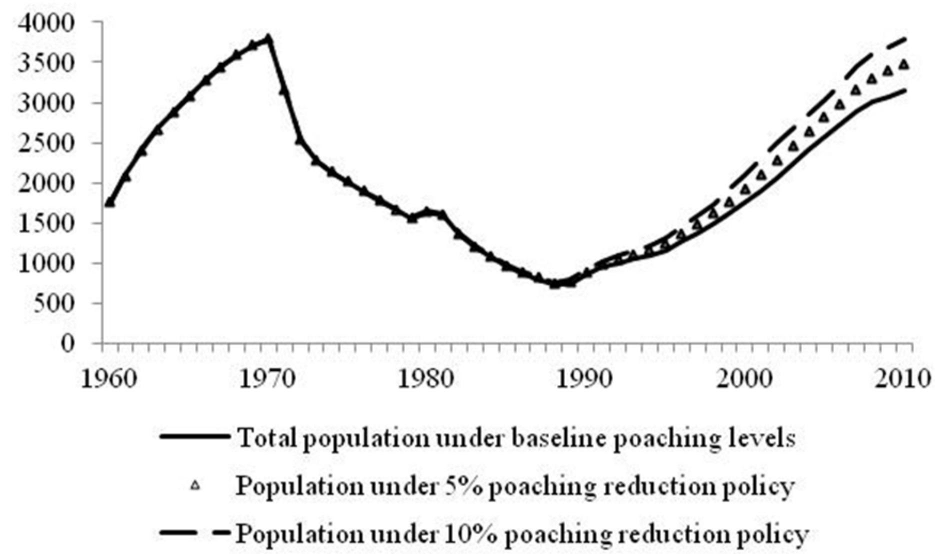


Figure 4.36 Total elephant population under the 5% and 10% poaching reduction policies

Reduction in poaching by increasing law enforcement, monitoring, and household incomes by 5% and 10% would result in a greater population dynamics. In order to assess whether or not there were any significant differences between the baseline population and the simulated population and these two different scenarios, a Wilcoxon test was performed. Again, the null hypothesis was that H0: B0<B5 at $p<0.5$, and alternative, H1: B0>B5 measured at a 95 % significance level. The results of this test showed that there was a significant difference ($W = 1036$, p-value = 0.038) in population when a five percent increase in law enforcement, monitoring and household incomes was made. The population showed a steady increase under the five percent increase in the poaching reducing interventions compared to the population under the baseline conditions. Similarly, an increase in law enforcement, monitoring, and household incomes by 10% resulted in a significant difference in population compared to the baseline ($W = 996$, p-value = 0.0208).

This analysis was extended to examine the actual cost involved in implementing these policy options. For example, based on the cost estimates of efficiently targeting law enforcement for the eight protected areas in GVL calculated by Plumptre et al. (2014), increasing law enforcement by 5% and 10% respectively the total cost of law enforcement alone would total up to US$6.16 and US$6.45 million dollars per year respectively, a cost way above the current budget cost of 5.86 million dollars (Plumptre et al. 2014) (Table 4.7).

Table 4.7 The current unit and total cost of law enforcement under the five and ten percent law enforcement policy for the selected National parks of Greater Virunga Landscape

NP	Area (km^2)	Annual budget**	Unit Cost US$/$km^2$	5% Unit Cost US$/$km^2$	10% Unit Cost US$/$km^2$
QECA	2040	757,617	371	390	408.1
Virunga	7844	3,500,000	446	468	491
Volcanoes	160	470,000	2938	3,085	3232.0

** Plumptre et al. 2014

The unit cost of law enforcement in each of the National parks under the five and ten percent policy scenarios increases way above the current cost. This, however, results in a mean population of 2080 (StDev ±947) elephants under the five percent policy, mean of 2148 (StDev ±910) elephants for a 10 percent policy compared to the baseline population mean of 1770 (StDev ±848) elephants. Under the ten percent poaching strategy reduction policy, 378 elephants are saved compared to 310 elephants under the five percent policy in relation to the baseline mean population. According to United Nations Development Programme (UNDP), "in Kenya, wildlife attracts over one million tourists per year, generates over 12% of the national revenue, and directly employs over 230,000 people." In the same country, a single elephant's economic value through tourism is worth $1 million in its lifetime, and the elephant's tusks are worth about US$18,000 wholesale in Asia. Using these figures, the marginal benefits and cost were computed when the lifespan of the elephant is considered to be 60 years (Table 4.8).

Table 4.8 Marginal benefits and costs under the two poaching reduction strategic policy options

Scenario	Mean elephants	Elephants saved	Marginal benefit over 60 yrs (Tourism)	Ivory Benefit	Marginal cost	unit cost/yr	Total cost over 60 yrs
Baseline	1770		1,000,000		5,860,000	3310.73	198,644.1
5 percent	2080	310	310,000,000	5,580,000	6,160,000	2961.54	177,692.3
10 percent	2148	378	378,000,000	6,804,000	6,450,000	3002.89	180,167.6

The benefits of conserving the elephants for tourism very much outweigh the economic benefits from one-off sale of elephant tasks.

The cost associated with implementing a compensation strategy for communities that undertake to monitor and enforce community policing against illegal killing of elephants was also computed. Using the economic value derived by households living adjacent to forests/woodlands, wetlands, and fisheries calculated from the socioeconomic survey data of 2006, the cost of incentivizing the local communities to participate in conservation of elephants and protected are resources in general was even much higher than for the law enforcement (Table 4.9). The maximum economic benefits derived by the households were used in the calculation because the average income per household of US$672 per year, at the time was not high enough to trigger off collective action in terms of reporting and policing illegal activities in the protected areas. Thus, the maximum household economic benefits from each of the three ecosystem systems were used. Under both the five and ten percent policy options, the cost of rewarding communities for participating in the conservation of elephants through community policing and monitoring is very high. Increasing household incomes was envisaged as one way of the viable strategies for reducing poaching, particularly poor households are more easily convinced to engage in poaching or bushmeat hunting because they are assured of a one-off income in their life time associated with the sale of ivory. To reduce this risk, providing economic incentives with clearly defined reward and penalty system is considered a good strategy. The households are expected to offer support to conservation of elephants and return receive entrepreneurial skills training, money to pay for direct upfront costs of creating alternative income generating activities.

Table 4.9 The incentivization cost for rewarding community participation in conserving elephants

	Number of Households	Max HH annual Income (US$)	5% Total Income subsidy (US$)	10% Income subsidy (US$)
Forests/ woodland	69565	1600	116,869,565	122,434,783
Wetlands	21739	1673	38,188,043	40,006,522
Fisheries	8696	1237	11,294,348	11,832,174
Total	100,000	1503.3	157,850,000	165,366,667

4.8 Policy implications

The challenges to elephant conservation and management in the face of massive poaching, prolonged civil wars, and climate change is daunting (Hulme 2005; Kostyack & Rohlf 2008). Fortunately, considerable attention is being given to climate change impacts through conservation planning programs, policy adjustments, increased awareness and education to mitigate potential climate change impacts, and developing new management (McLachlan et al. 2007) and financing strategies. In this study, we considered these challenges as nested, multiscaled, and multidisciplinary demanding a holistic approach. It was also recognized that the hierarchical conservation planning, though useful toward prioritization of the places with the greatest conservation need, and a focus on species population management is rooted in the system of managing wildlife, wildlife managers and research examine these inextricably linked problems in total isolation. As such, an integrated approach was necessary to examine these challenges collectively. Under this approach, policy options and conservation efforts can be evaluated and strategic decisions made to effectively allocate scarce resources and key players persuaded to intervene at a broader scale. A number of policy experiments were conducted in this study aimed at sustaining or improving the elephant populations in GVL namely: 1) suitable habitat increase through the reduction of forest and savanna vegetation conversion to human settlement and agriculture; 2) reduction in poaching through increased law enforcement, monitoring and research, and increasing household incomes. Results of the first policy experiment, suggests that increasing the current suitable habitat by 50% would significantly lead to an improvement in elephant population numbers. This could be achieved mainly through reduction of forest and savanna vegetation loss and conversion to human settlement and agriculture, proper management of fires, and invasive species. The second policy focused more on reducing poaching of elephants. It was demonstrated that by increasing law enforcement, monitoring and research, and household incomes for the protected area adjacent communities by 10%, poaching would reduce significantly. This was based on the understanding that the conditions that encourage poaching such as the price of ivory, human population, and poverty are either maintained as per the baseline conditions or reduced. It was recognized that a lucrative international market for ivory is a big incentive for poachers to take the risk and kill elephants for ivory. Unfortunately, dealing with this clandestine market is very complicated and destroying the international market is way beyond the jurisdiction of the wildlife managers, local and national governments, but rather requires a collaborative effort with the international community. Strategies to reduce human population increase and the associated poverty, demand national effort through allocation of resources in capacity building, development of entrepreneurial skills, creation of alternative incomes, and family planning strategies. Therefore, an attempt to experiment with these policies was made, but it did not offer any meaningful and attainable solutions because the price of ivory is highly sensitive and would require eliminating the market completely by reducing the price to zero in the model. Furthermore, poverty and human population had to be reduced by 60 percent, which is nearly impossible in the short and mid-term.

4.9 Conclusions and recommendations

Elephants in Africa are disappearing at an alarming rate mainly due to poaching, habitat degradation and loss, and retaliation killings due to human-wildlife conflicts. The situation is expected to become worse with the advent of climate change impacts resulting in high occurrence of prolonged droughts in both arid and semiarid regions. Understanding how animal populations will respond to such dramatic events is very crucial to the design and implementation of mitigation strategies, and development of adaptive capacity of wildlife managers to adequately respond to these challenges. We explored how GVL age specific elephant populations are likely to respond to habitat change, war, poaching, water resources and climate change. The senior or old elephants did not respond very well to all the anthropogenic stressors that were investigated. For example, an increase is war frequency and magnitude, whatever the scale, affected the senior elephants more than the juveNiles and sub adults who constitute the age classes 0-10, 11-30, and 31-40. In addition, climate change affected the old age elephants more than young ones in terms of survival abilities, but this could be due to immigrations from other areas. It is also likely that the undetected direct climate change impact on elephant population is due to the presence of reasonable area of suitable habitat particularly forests and wetlands used for thermal regulation. This, however, does not rule out the fact that indirect impacts such as thermal and latent flux impacts are occurring already. On the other hand, an improvement in the habitat type and availability of water resources resulted in a slight increase in all age class populations. In all the analyses, the results seem to suggest that if the environmental and anthropogenic stressors are not mitigated, GVL will have only a very young population of elephants. Studies elsewhere have showed that elephants are particularly sensitive to the drought and calf mortality was higher among young mothers than the more experienced mothers (Foley et al.2009). A very dramatic improvement in elephant population was observed when both a 5% and 10% poaching reduction policy is implemented. This, however, requires a ten percent matched increase in law enforcement, monitoring, and household incomes. Furthermore, regional governments together with conservation organizations and donor community must be willing to invest in poverty reduction programs, scale up the effort to reduce the international demand for ivory through CITES, treat ivory business as a national and global security threat, and invest in human population reduction strategies such as family planning, and education.

Elephants in GVL are a transboundary resource which requires transboundary management approach, cooperation between conservation agencies, and development of effective partnerships with all relevant stakeholders such as local and regional governments, wildlife protected area authorities of DRC, Rwanda, and Uganda, law enforcement agencies such as the military, police, judiciary, customs and border control authorities. Collaboration is fundamental to effective conservation (Wondolleck & Yaffee 2000; Plumptre et al. 2009) and regional governments must be willing to develop institutional frameworks to allow it to happen. Law enforcement helps to reduce illegal activities inside protected areas and is the only way the park authorities can identify and reprimand those involved in illegal activities. Law enforcement, however, requires a lot of man power, finances, and equipment in form of camping gears, patrol vehicles, and training in handling of incriminating evidence in the field to be presented to courts during litigation.

Plumptre et al. (2014) conducted a study in the Greater Virunga Landscape to assess how illegal activities in protected areas can be reduced by efficiently targeting resources and estimated the cost of effective law enforcement to meet all the conservation targets in the landscape by 63% to be US 2.2-3.0 million dollars. Currently, US$ 5.9 million is spent on law enforcement, including transboundary monitoring in GVL (Plumptre et al. 2014). Increasing rural household's incomes through establishment of alternative income-generating activities that are not entirely nature-based require huge funding and commitment from national governments as well as the donor community.

In itself, stopping bushmeat hunting is an economic war that could slide into civil unrest due to lost revenue as well as regional markets. For example, across the continent, rural and urban populations still depends heavily on bushmeat for protein, and the pressure will surely increase as the human population rises (Wilkie 2001; Pearce, 2005). Bushmeat hunting provides an important source of revenue to local communities and has been estimated to contribute 33% of total annual village income (Infield, 1988). Bushmeat trade also makes a significant contribution to national economies in Africa, particularly West Africa. A study by Fa et al. (2000) revealed that Ivory Coast and Liberia earned an estimated annual income of US$117 million and $42 million respectively from bushmeat trade. Based on these shocking economic benefits, it is obvious to the readers of this thesis that provision of alternative income sources is inevitable, and wildlife management authorities can not afford to implement this intervention on their own, hence the need for collective participatiom of the private, governments and donor community in order to create alternative livelihoods for the rural poor.

4.10 References

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CHAPTER 5

ECOSYSTEM SERVICES

5.1 Introduction

Balancing the trade-offs between satisfying immediate human needs and maintaining ecosystem functions and services requires quantitative knowledge about the nature and flow of ecosystem services both spatially and temporally. Tropical forests play a critical role as repositories of biological diversity and in regulating global biogeochemical and hydrologic cycles. They also provide the biological underpinnings essential to sustain life, including water supplies, water and air purification, flood control, nutrient cycling, pest control, pollination, carbon sequestration and wildlife protection. Typically, provisioning services such as timber, minerals, fish and Non Timber Forest Products (NTFPs) contribute greatly to the regional economies. Equally important is the contribution of forests and wetlands to regulation of climate change mainly through carbon sequestration, and hydrological processes.

5.1.1 Problem

The Congo river basin is facing eminent challenges from increasing human population, civil strife, extractive industry developments, particularly logging companies, oil and gas, and climate variability and change. These have led to large-scale impacts on the region and zones of conflict for resource use have emerged within and outside protected areas. Studies quantifying the responses of wildlife behavior to industrial disturbances have shown that the construction of roads, trails and buildings and the increased presence of human activities beyond a certain threshold resulted in a direct loss of wildlife, their habitats and indirect habitat loss following avoidance behavior of affected wildlife species (Wilkie et al. 2000; Dyer et al. 2001). In addition to influencing patterns of habitat use, avoidance behavior may result in cumulative social and physiological consequences that may lead to implications on population productivity (Johnson et al. 2005). The impact of oil and gas exploration activities on ecosystem services may be direct and obvious such as replacement of the native habitats with well pads, access roads, and pipelines but the physical footprints may be insignificant relative to the functional habitat loss due to avoidance (Dyer et al. 2001; Sawyer et al. 2009). The planned location of an early production system in Uganda's portion of the Albertine Rift, which is contiguous with the Congo basin with a targeted production capacity of 40,000 barrels per day of heavy crude oil processed into diesel, kerosene and naptha is a major cause for concern with regard to direct and indirect impacts on the ecosystem services. It is anticipated that once the oil and gas production refinery and gas pipeline construction starts, this will attract a lot of migrants looking for employment opportunities. As a result, the demand housing land, forest products and agricultural land to feed the growing population in the region will become immense. Unfortunately, the lack of capital resources to achieve effective intensification on currently cultivated land and loss of cropland due to degradation suggest a large probability of future forests and wetlands loss.

The other important issue concerns the new global shift toward conserving forests by implementing carbon sequestration initiatives through REDD+ financing options in the future. The governments of Uganda, Democratic Republic of Congo and Tanzania have made commitments to the UN REDD + program to invest in REDD+ activities as a way of addressing forest loss and degradation in the region. Furthermore, the promotion of tree planting to secure carbon credits may result in unavailability of land for agriculture leading to food insecurity, creating a poverty trap and dependence on protected areas for livelihood Bell, & Callan, 2011). Similarly, the planned displacement of crop farmlands with commercial plantation crops such as tea, palm trees and sugarcane, present major livelihood and environmental challenges. The regional governments, including Rwanda and Burundi are committed to the development of a vibrant tourism sector coalescing around Mountain Gorillas that occur nowhere else in the world, but move regularly across international boundaries and are a shared resource. A huge investment in infrastructure development, specifically paved roads, railways, telecommunication and power transmission lines will make remote parts of the Congo basin accessible from Ugandan side. Studies quantifying landscape patterns and dynamics in East and Central Africa have shown that the main drivers of change in land use and land cover namely agriculture and extraction of minerals and timber (Laporte et al. 2004; Duveiller et al. 2008; Van Vliet et al. 2012) also happen reduce ecosystem services, particularly biodiversity values (Copeland et al. 2009; Sawyer et al. 2009).

5.1.2 Current approaches

A number of planning tools have been developed and are being applied in conservation planning. For example, Marxan (developed by the University of Queensland http://www.uq.edu.au/marxan), a stochastic optimization tool that has been used extensively in conservation planning (Watts et al. 2009a; Watts et al. 2009b; Ball et al., 2009) to examine trade-offs in land appropriation aimed at minimizing conflicts as well as identifying options for offsets. Although Marxan is a good planning tool, it requires extensive consultations and negotiations with all stakeholders to agree on drivers of landscape systems and processes change, define objectives and set realistic targets, which is very costly and time consuming. Computable General Equilibrium (CGE) Models and Biophysical accounting have been used to provide an analytical framework for understanding the relationship between trade and population by measuring the impact of trade in a specific commodity on aggregate production and land use changes (Mendelsohn, Nordhaus, and Shaw 1994). The CGE model relay heavily on its ability to rank welfare effects of different policy instruments (Stenberg and Siriwardana 2005) as it performs the analysis at a disaggregated level. CGE models have their limitations such as the reliance on neo-classical assumptions of perfect competition and production functions characterized by constant returns to scale (Bandara 1991). In addition, the results from CGE are difficult to explain to policy makers, hence considered as black box models. Aware of these limitations and the limited knowledge of overcoming those challenges, CGE models were not the best choice for this study. Other relevant models developed by NASA such as Terrestrial Observation and Prediction System (TOPS) (Nemani et al. 2009) and Spatially Explicit Regional Growth Model (SER-GOM) (Theobald 2005) were also explored. These models have been applied in Ecological Forecasting (EF) (Nemani et al. 2009) focusing mainly on land use change and climate change effects on water yield and quality. Spatially Explicit Regional Growth Model (SER-GOM) has been used to assess infrastructure-related land use developments on water quality (Theobald 2005) were reviewed and considered to be good candidates for this study. Most of these methods are very data-intensive and need more time to learn how to generate alternative data suitable for use.

5.1.3 Justification of the study

In order to maintain the resources that local people use as well as those that conservationists and other key stakeholders value, appropriate strategies need to be developed in which multiple and often conflicting uses or values of natural resources can be coupled or negotiated (Kaimowitz and Sheil 2007; McShane et al. 2011; Sayer et al. 2013). Proponents of Integrated Conservation and Development (ICD) in the management and conservation of forest resources have failed to account for the complexity and dynamics of underlying contexts and drivers of land use change and how decisions made at varied scales (e.g. household, community, national and region) impact the stocks and flow of ecosystem services and goods. It has been rightly argued by scholars (Schweik 2000;Lynam et al. 2007; Norgrove and Hulme 2006) that real and perceived ecosystem benefits weighed against cost incurred to secure the integrity and proper functioning of ecosystems largely determine people's use of, and impact on that resource. Protected areas in the Congo Basin cover approximately 6% of the landscape, and several international NGOs are proposing substantial additions to the present network of parks and reserves (Wilkie et al. 2001; Zhang et al. 2006).

In this study, an alternative approach based on an integrated spatial modeling, which includes the main sectors competing for land mainly agriculture, forestry, bioenergy and wildlife protection to indentify the ecosystem services hotspots and evaluate different policy options was done. This approach had several advantages in that the ecosystem values are explicitly represented through the multi-sectoral approach and the watershed level analytical framework ensured that external drivers of ecosystem services mainly human-driven changes are taken into account. The best available data on land use, soils, carbon sequestration, biodiversity survey data, and observation data from remote sensing platforms were used. The work also benefited from the feedback of scientists and researchers working in the region.

5.1.4 Uniqueness of study

This study depended heavily on ecological knowledge and the application of Geographic Information System (GIS) to map the ecosystem services of the Congo basin. The main emphasis here was to propose an improvement to the existing environmental accounting techniques to identify and map areas of the basi with varied ecosystem services from purely an ecological perspective. In general, valuation data provide prima facie support for the hypothesis that net ecosystem service value diminishes with biodiversity and ecosystem loss (Balmford et al. 2002).

Remarkably, where attempts have been made to analyze values of ecosystem goods and services, biological capacity, and ecological footprints, such analyzes have treated these three fundamental elements independently yet they are inextricably linked. In addition, the framework for such studies sought to capture the state of environment 'before and after changes have taken place as opposed to "with" and "without" economic analysis. Aggregate estimates of ecosystems services at a regional or global scale are problematic, given the fact that only 'marginal' values are consistent with conventional decision-aiding tools such as economic cost/benefit analysis. Importantly, reliable measures of the supply of, and human demand on, natural capital are critical towards the design of natural resource management strategies, setting targets, and drive policies aimed at achieving economic efficiency. In this study, an integrated approach to ecosystem services valuation was used where different ecosystem services namely carbon stocks, water quality and quantity, and biodiversity values were mapped later aggregated using Geographical Information Systems (GIS) to develop a multiple attribute ecosystem index.

5.1.5 Objectives

The study aim was to quantify and develop a spatial decision support tool that can be used by resource managers to strengthen habitat and biodiversity conservation through PES and REDD schemes. Specific objectives of the study are to: i) assess the extent of ecosystem services at a coarse spatial scale (1km x 1km) by focusing on carbon stock, water quantity and quality, and habitat potential for multiple species; ii) quantify and integrate multiple ecosystem services into a multi-attribute ecosystem service index (MESI); iii) examine the impact of single policy strategies on the value and spatial distribution of other ecosystem values under consideration.

5.1.6 Hypothesis

The hypotheses are: i) Spatial variability in ecosystem services exists in the Congo watershed; ii) ecosystem benefits can be maximized under multiple policy objectives implemented at a watershed scale.

5.2 Literature review

Ecosystems provide a wide range of direct consumptive (e.g. timber, fish, minerals, and Non Timber Forest Products), and non consumptive benefits (e.g. tourism and recreation, research, cultural and worship values) that are enjoyed locally and globally. In the Congo Basin, the value of African rattan trade on the local market was estimated at US$287 505 in 2001 (Sunderland 2001), and at the international level, annual imports of medicinal plants was approximately US$3.5 million (CARPE, 2001; Ndoye and Tieguhong, 2006). The largest growing market segment of nature-based economic activity with minimal environmental impact, ecotourism, is presently valued at over US$3 trillion per year (WWF 2000). A global net carbon emission from land-use change in the tropical regions is estimated at 1.1±0.3 Gt C yr-1 (Achard et al., 2004). Land use change, primarily through forest loss and degradation in the tropics is estimated to contribute 20% of all anthropogenic carbon emissions (IPCC 2007; Kapos et al., 2009). In Africa, deforestation alone accounts for nearly 70% of total emissions (FAO, 2005). FAO (2006) estimates that between 2000 and 2005, gross deforestation was 12.9 million hectares per year, while the most recent studies taking into account afforestation and natural expansion of forest estimate annual net loss of forest at 7.3 million hectares per year globally (Nabuurs et al. 2007). Tropical deforestation is estimated to have released of the order of 1–2 billion tonnes of carbon per year during the 1990s, roughly 15–25% of annual global greenhouse gas emissions (Malhi and Grace 2000; Fearnside and Laurance, 2004; Gibbs et al., 2007). Moreover, clearing tropical forests also destroys globally important carbon sinks that are currently sequestering CO_2 from the atmosphere and are critical to future climate stabilization (Stephens et al 2007).

More recently the importance of including emissions reductions from tropical deforestation in future climate change policy has grown. During the 15th Conference of the Parties (COP) on Climate Change held in Copenhagen, Denmark, the United Nations Framework Convention on Climate Change (UNFCCC) was mandated to spearhead and coordinate the implementation of mechanisms for “Reducing Emissions from Deforestation and forest Degradation” (REDD) as a means of reducing global carbon emissions. With this intention, numerous forest-based pilot projects have been implemented, and many of these projects have accurately measured the carbon benefits to high degrees of precision at a very high cost (Brown 2002; Brown et al., 2005). A study by Turner et al. (2004) has demonstrated the complementary nature of remote sensing and ecosystem modeling to adequately measure terrestrial carbon cycling. As a result of this, a number of studies have been conducted to estimate carbon stocks using remote sensing techniques and a combination of datasets from different platforms (Baccini et al., 2008; Goetz et al., 2009).

Valuing ecosystem services, however, requires the successful integration of ecology and economics and presents several challenges especially in making explicit descriptions and adequate assessment of linkages between structures and functions of natural systems, the benefits derived by humans, and their subsequent values, including non-use values (NRC, 2005, p2). The value of specific ecosystem functions and services is entirely relative. As such, spatial and temporal scales of analysis are critical determinants of the potential value estimates. Although a variety of valuation methods are currently available (Pressey 1998; NRC, 2005; van Jaarsveld et al., 2005), no single method can be considered best at all times and for all types of ecosystem applications. For example, Costanza et al (1997) estimated the annual values for 17 ecosystem services from 16 ecosystem types including forests, wetlands, grasslands, estuaries, and other marine and terrestrial ecosystems to range from US$16 trillion to $54 trillion, with a mean estimate of $33 trillion. Several authors (vatn and Bromely, 1994; Shriver and Randhir, 2006; Randhir and Shriver, 2009) have challenged the presumption that environmental choices made without pricing mechanisms are inferior to those using hypothetical valuation. In practice, majority of natural resources and ecosystem-related decisions have relied on attribute-based methods without explicit use of money-metric prices to prioritize management objectives (Lamy et al. 2002) and efficiently allocate resources (Antunes et al., 2006; Howarth and Wilson, 2006; Hajkowicz and Collins, 2007; Randhir and Shriver, 2009). Therefore, total ecosystem services valuation that also accounts for cumulative impacts due to alterations in the ecosystem's disconnectedness to the entire landscape, requires assessment of changes in ecosystem services that may play out at different spatial and temporal scales. This, however, does not require compressing and commoditizing this complexity into simple metrics of monetary values, which result in further loss of information (vatn and Bromely 1994).

5.3 Methodology

5.3.1 Study area

The Congo Basin covers approximately three million square kilometers and includes the world's second largest tropical forest of approximately two million square kilometers (Laporte et al., 1998; Wilkie et al., 2001; Hansen et al., 2009; Bwangoy et al., 2010). The Basin's forest zone includes parts of six central African countries: Cameroon, Central African Republic, Congo, Equatorial Guinea, Gabon, and the Democratic Republic of the Congo. The study area includes the core Congo River basin, located within the forest zone from 5° north to 6° south and from

13° to 26° east. The Congo is the second largest river in the world by water discharge and surface area, and estimated to be 4700 km long (Gaillardet et al., 1995). The climate of the study area is warm and humid with two wet and two dry seasons (Samba et al., 2008). The mean temperature is approximately 25 °C, and the difference between the temperature of the warmest month (March) and the coldest month (July) is only 2 °C (Bwangoy et al., 2010). The average rainfall is about 1800 mm per year in 115 days (Bwangoy et al., 2010). The basin is largely bisected by the Equator with little seasonal variation within 1° north or south of the Equator. The climax ecosystem is tropical evergreen forest with little seasonal variation (Mayaux et al., 2000) typified by irregular and very dense (70–100%) crown cover that often preclude the development of understory shrubs and grasses (Mayaux et al., 2000; Mayaux et al., 2004). Congo basin is one of the world's important biodiversity areas highly recognized for habiting the lowland gorillas, primates (Brooks et al., 2004) and elephants (Blanc et al., 2007). It is also contiguous with the Albertine Rift, a biodiversity hotspot (Brooks et al., 2004; Plumptre, et al. 2007).

The total human population in the Congo Basin region is estimated at over 73 million people, with an annual growth rate of over 2.5%. The average population density is about 15 people km^2 (Ndoye and Tieguhong, 2006). Although the Democratic Republic of Congo's (DRC) forests and biodiversity have the potential to contribute to national, local, and global welfare, it is hampered by threats from subsistence agriculture, loss of forest lands and protected areas; unsustainable and illegal logging practices; the expansion of mining into biodiversity reserves; commercial trade in bushmeat and protected species, erosion of traditional property rights, and the continued presence of armed troops in many protected areas. These threats pose a major challenge to sustained provision of ecosystem goods and services in the region.

Figure 5.1 Map showing the Congo basin and overlapping countries

5.3.2 Conceptual model

The idea behind this study was that in order to fully understand how ecosystem services are spatially distributed in the Congo watershed, it was important that ecosystem values that are dependent on each other being examined together. The main reason was that carbon sequestration, water resources, and biodiversity are all depend on environmental conditions, land use land cover, interaction with human activities, and any changes in one ecosystem service alters the other. The drivers of land use land cover change, particularly deforestation, agriculture, resource mining, and human settlements and development are driven by humans who place different values to how land and its products are used. For example, it has been documented that increases in fertilizer application (Sharpley, 1994) and rapid urbanization result in degradation of water quality (Ren et al. 2003). Ruesch and Gibbs (2008) overlaid biodiversity data from Important Bird Areas and Key Biodiversity Areas together with land cover to map conservation priority areas. Ewing et al. (2008) developed methods linking land use types to their bioproductive capacity to supply ecosystem services and goods. In the model framework shown in Figure 5.2, land cover, soils, and precipitation were used to compute water supply and quality, soil loss, topographic characteristics, and biodiversity geospatial data were used to compute and aggregate the ecosystem services into a multiattribute ecosystem services index.

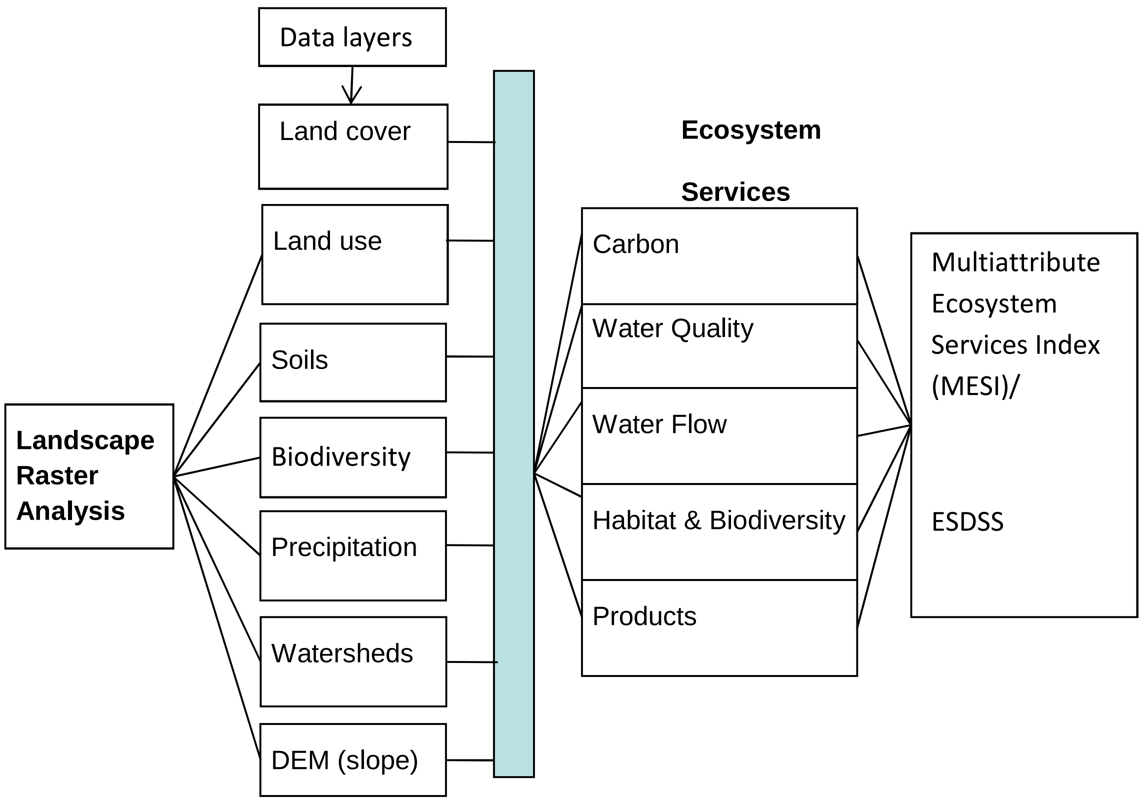

Figure 5.2 MESI and ESDSS Framework

5.3.3 Empirical model

Following methods of Shriver and Randhir (2009), a Multiattribute Ecosystem Service index (MESI) was developed from raster information on individual services. For each service, an ordinal scale of ecosystem service values under each category of ecosystem was developed. The rasters were then rank-ordered by the value of each ecosystem service on a scale of 1 to 5, (i.e. low to high value). MESI was a sum of the product of individual weights and indices represented as ($MESI = \sum_i w_i * x_i$) in order to calculate multiple ecosystem services from each raster.

Ecosystem services in a grid cell were given equal weights (w_i =1). The MESI estimates were displayed as maps and integrated in GIS that coulde be used as Ecosystem Services Decision Support tool (ESDSS). Technically, comparisons at subbasin and watershed scales were made to highlight areas with low and high ecosystem values. A resource manager can use this tool to maximize ecosystem services benefits through conservation investments in areas of high potential and implement restoration programs in areas of ecosystem impairment. The tool also empowers the manager or decision maker to identify stakeholders, resource needs, implementation and institutional challenges by zooming in to the individual subbasin.

5.3.4 Biodiversity, and watershed ecosystem values

A detailed description of how biodiversity, and watershed values were derived is provided in chapter two and three respectively. However, a summary of the methods used is provided here. Species distribution models were developed based on presence and pseudo absence data. Th field inventory data together with a number of selected environmental, biophysical and socioeconomic variables selected and assessed for multicollinearity, were included in the generalized model. The GLM model was then used to compute the probability distribution map showing areas with realized and potential species distribution. Runoff was compiled using the GIS based Curve Number (CN) (USDA/SCS, 1972; Renard et al. 1991, 1994, 1997) and soil loss map was developed using the Revised Universal Soil Loss equation (RUSLE). The results were validated for water accuracy using empirical monitoring data collected by government agencies responsible for water resources monitoring in the region from, both tributaries of Congo basin,

5.3.5 Carbon values

The third ecosystem service that was used in the study is the Above- and- Below ground Carbon. In 2008, the Woods Hole Research Center produced a first map of the distribution of above-ground biomass covering the tropical region of Africa by utilizing images from the Moderate Resolution Imaging Spectrometer (MODIS) satellite (1-km resolution) along with data from recent forest inventories covering the period from 2000 to 2003 (Figure 5.3). Another carbon map for Africa was produced by NASA-Jet Propulsion Laboratory (JPL) (Saachi, et al. 2011), which they accepted to share for use in this particular study (Figure 5.3). In both cases, Baccini et al. (2008) and the Saatchi et al. (2011) datasets are based on remotely sensed data from the early 2000's. Mitchard et al. (2011) argued that the carbon map produced by Baccinin et al. (2008) underestimated the AGB of forests and woodlands, while overestimating the AGB of savannas and grasslands mainly due to less field data used.

The recent pantropical biomass map produced by Baccini et al. (2012), however, used some field plot data from the Democratic Republic of Congo (DRC) and Uganda in 2009 coupled with satellite observation Lidar (Light Detection and Ranging) data acquired by the Geoscience Laser Altimeter System (GLAS) satellite to characterize above-ground biomass for Africa. Baccini et al. (2012) estimate the aboveground biomass carbon in the DRC at about 22 Gt. United Nations Environment Programme (UNEP)' World Conservation Monitoring Centre (WCMC) conducted an assessment of carbon stocks and biodiversity benefits in DRC in 2011 in partnership with DRC partners *Observatoire Satellital des Forêts d'Afrique Centrale (OSFAC) and Ministère de l'Environnement, Conservation de la Nature et Tourisme* (MECNT)/ Direction *des Inventaires et Aménagement Forestiers* (DIAF), and funded by the UN-REDD Global Programme (Musampa Kamungandu et al. 2010). WCMC considered both Saatchi et al. 2011, and Baccini et al. 2008 carbon maps in producing the carbon and biodiversity map for DRC. This study, did note make use of this data because it was too late to request for it and use it in the analysis. The average carbon density per ha derived by WCMC was used to compare and improve the estimates provided by Jet Propulsion Laboratory (Saatchi et al.,2008) and Woods Hole Research Center (Baccini et al. 2008; 2012) this data to improve the carbon estimates.

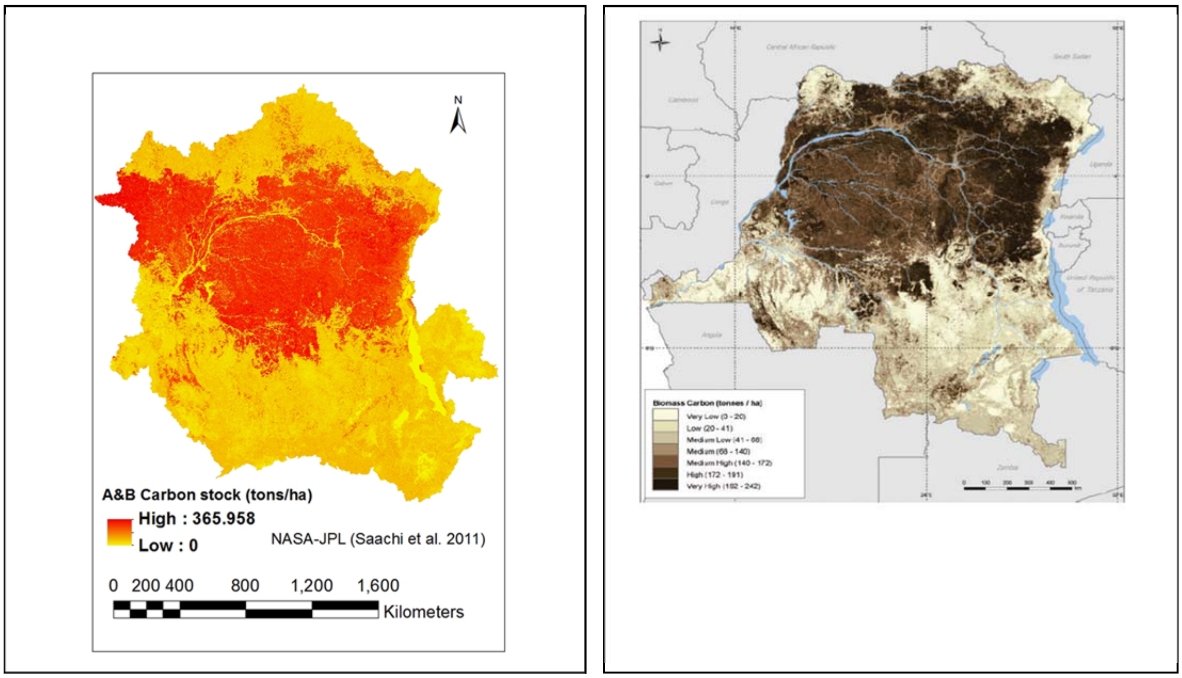

Figure 5. 3 Carbon map produced by NASA/JPL (Saatchi et al. 2008) left, and Woods Hole Research Center carbon map for Congo (Baccini et al.,2008)

The species distribution models for selected species discussed in chapter two of this thesis namely biodiversity values, watershed, and carbon sequestration values were aggregated to produce a multiattribute ecosystem services map. First, an equal weight (on a scale of 1-5, where five is high) was assigned to all the three ecosystems purely for scaling purposes. Thereafter, relative weights were applied based on expert knowledge. Fifteen expert with reasonable experience of working in the region were asked to rank the values using the stated scale proposed by Saaty (1999) and a more systematic technique was used to ensure that the scoring the three services doesn't exceed a specified value in order to avoid inflation bias. The total scores were averaged to produce scaling factors for individual ecosystem service and these were multiplied by the corresponding raster layer to prouce a surface map of that particular ecosystem service. The results from the ecosystem services mapping were converted into an index using the following procedures. 1) Identify the maximum and minimum values in each ecosystem service raster layer; 2) Calculate maximum – minimum values; 3) For each pixel, calculate (pixel value/max-min) to convert all values to a range of 0 to 1. Integration of the ecosystem services of all ecosystem services (i.e. carbon, biodiversity, runoff, and soil loss was done using a raster calculator. The results of this study are presented in the next section.

5.4 Results

The maps developed by aggregation of the individual species maps in each of the taxa are prented here. Map labeled A represents the amphibian values for the modeled species, B represents the avifauna values for selected endangered ans threatened spceices, and map C represents the fish species mmodeled (Figure 5.4).

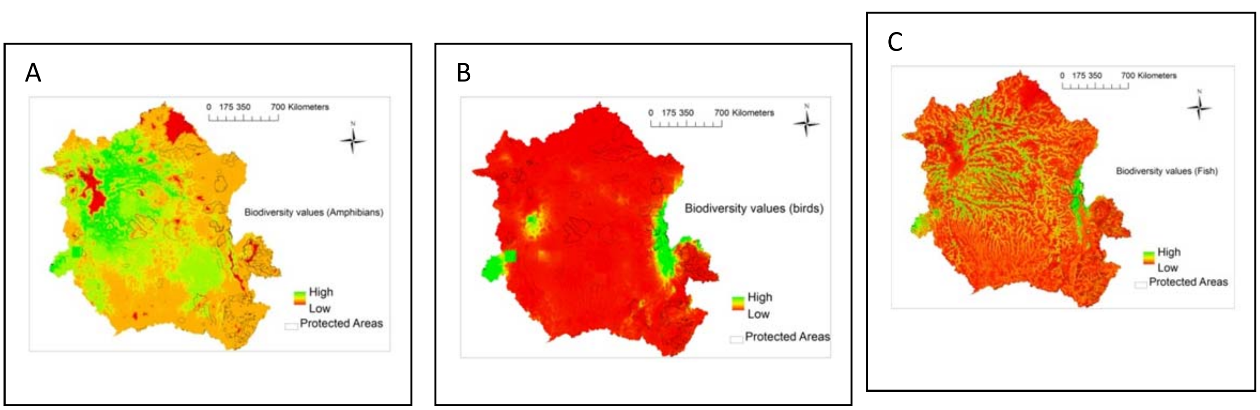

Figure 5.4 Biodiversity values associated with amphibians, birds, and fish in the Congo basin

All large mammals species, and reptiles that were selected and modeled, were agrggregated together to form a a large mammal value, and fish value map (Fgiure 5.5). The green color is used to represent the region of high value and the red color represented low ecosystems values.

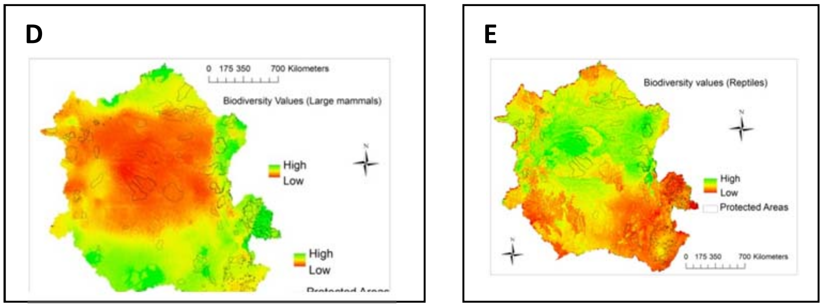

Figure 5.5 Biodiversity values associated with modeled large mammals (D) and reptiles (E)

As indicated earlier in the methods section, the ecosystem services maps were produced with and without expert scoring done. The idea was to show the ecosystem services based on the existing ecological attributes and then introduce the human-based valuation of the services. By scoring low a particular ecosystem, the expert is revealing his individual values and low preference for that service. On the other hand, scoring an ecosystem value implies that individual highly values the importance of that service and should be willing to pay to protect it. In Figure 5.6, area in red indicates low ecosystem values as identifying during the mapping exercise and argumented by the expert opion. It is very clear that the southeastern part and the norther tip of the basin have relatively low ecosystem values compared to the central part. This is not surprising because nearly 60% of the protected areas are concentrated in this part of the basin and is also linked to the Albertine Rift region, a rich known for its high biodiversity (Plumptre et al. 2007). It is possible that species receive reasonable protection and can easily move from one area to another. Furthermore, this central part of the basin is served the main tributary of the Congo River, making it relatively difficult for people to access.

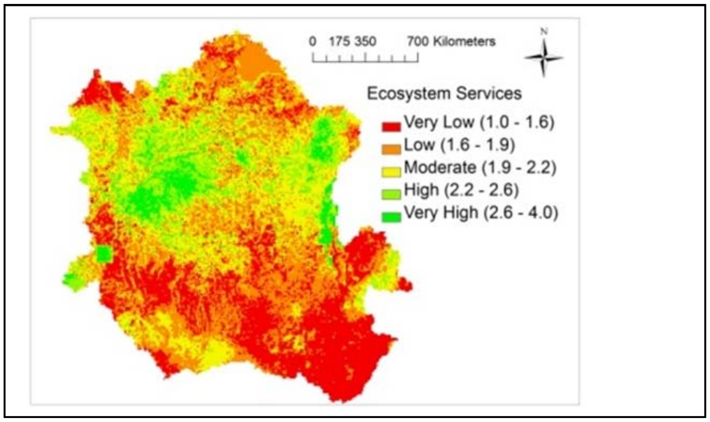

Figure 5.6 Esystem services map without scoring

When a score of five was multiplied by each of the ecosystem services to test for the likely impacts that occur when some level of bias is introduced, the ecosystem values distribution assuned a mosaic pattern with more values in the mid central part extending all the way to the western part of the basin (Figure 5.7). The southeastern portion of the basin still exhibited very low ecosystem values.

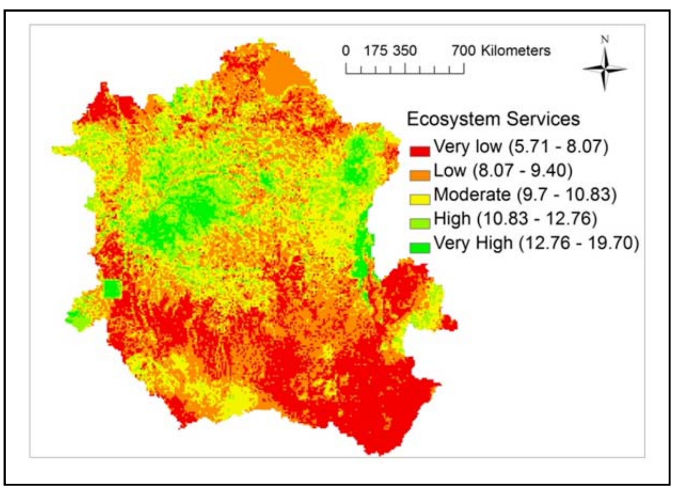

Figure 5.7 Ecosystem services map (ecosystems equally weighted at a scale of 5, where scoring scale ranges from 1-5, and 5 is the highest score)

The second part of the study results involved using the experts scores of the ecosystem services based a purely restrictive pairwise ranking and used the pairwise raking matrix to implement a weighted ecosystem services analysis (Figure 5.8). The values presented on the ecosystem services maps are entirely entirely used for identifying regions of high value at rough scale and not absolute quantities of ecosystem service values.

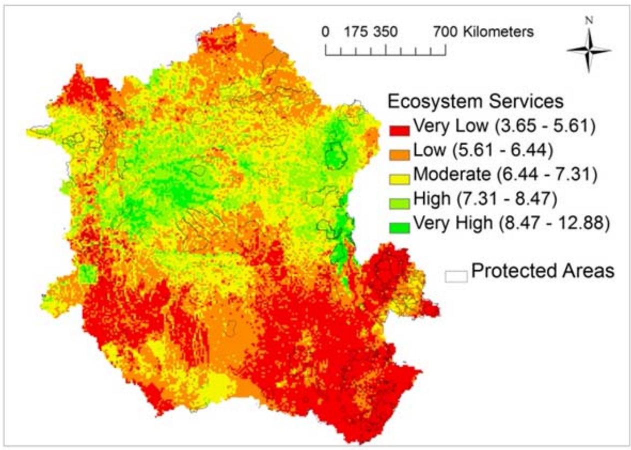

Figure 5.8 Ecosystem services spatial distribution with expert scores

Very high ES values were recorded inside protected areas and more the values are concentrated in the mid central portion for the basin that is relatively intact. The results reinforce the idea that lessening the human footprint mainly through road construction and construction of social service centres benefits conservation. Areas close to urban centres have very low ecosystem values and are more likely to be very expensive to restore, if the objective is to manage those areas for protection of nature. Out of the 10 experts contacted, on a scale of 1-5, where 5 is the highest score, on average, carbon got a score of 2.5, biodiversity was scored according to individual taxa (i.e. Amphibians = 2.0, Birds = 3.0, Fish = 3.0, Large mammals = 3.0), water quantity (runoff) = 0.5, and water quality (soil loss) = 0.5.

5.5 Policy options and their implications

A number of policy experiments were conducted by assigning the highest score (5) to the ecosystem service selected under a specified management regime such as a water quality and quantity policy (Figure 5.9), REDD (carbon sequestration) policy (Figure 5.10), and biodiversity policy (Figure 5.11). Under the water policy experiment, regions that already showed water scarcity problems still continue to prevail in the southeastern part of the basin.

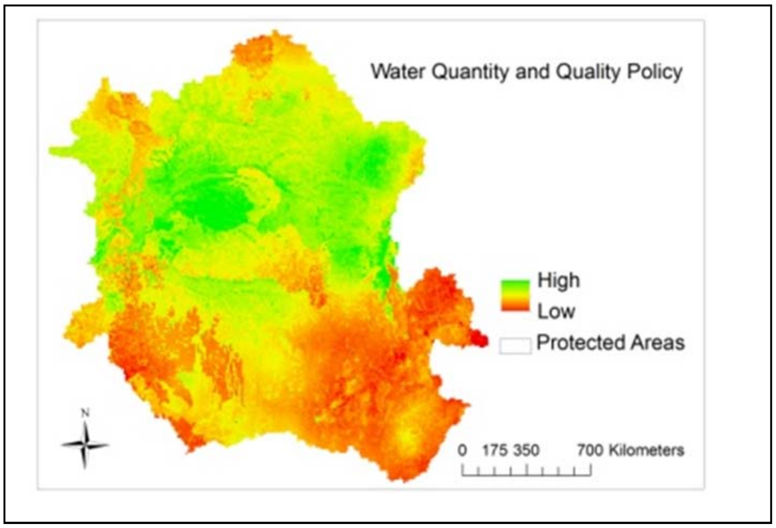

Figure 5.9 Single objective policy option, the water policy

These are areas with large population density and with relative good road density and facing a challenge of expanding commercial crop plantations such as tea, palm oil, sugarcane growing and to a limited extent livestock ranching. Such areas are likely to trigger massive movement to other remote areas by the rural poor in such for survival. Due to the diminishing sources of funding for conservation, conservationists and the donor community start rethinking alternative ways of sustainably financing conservation. Of such strategies, Reduced Emissions from Deforestation and Forest Degradation Plus (REDD+) was lunched nearly ten years ago by the

United Nations in partnership with the World Bank. The Global Environment Facility (GEF) provided a grat of US$13 million to Congo basin countries to develop REDD+ scheme. The national governments together with conservation organizations are focusing more on avoided deforestation, afforestation through strengthening of forest management, and afforestation. This is complimented by clean development mechanisms such the use of efficient energy saving stoves, biogas. Currently, many countries in the tropical regions with large chuncks of forest on both state and private land enrolled into the scheme to try and reduce or avoid forest loss and degradation.

In this study, it was note that there are on-going initiatives by conservation organizations such as the UNEP/WCMC, WWF, WCS, Woods Hole Research Center, and Jane Goodall Institute (JGI) with funding support from USAID, NORAD, and other philonthrapists to pilot small-scale, landscape level REDD+ projects. It is estimated that REDD+ projects could potentially generate up to US$900 million in revenue per year for DRC from 2010-2030 (UNEP, 2011). In order to contribute to this discussion, whether or not REDD+ is the next silver bullet for conservation, a REDD+ policy experiment was conducted where carbon sequestration was ranked highly relative to the other two ecosystem services. The results of this experiment showed that a purely REDD+ strategy could diminish the biodiversity values although it will boost the water supply in the region. Again, area with very high forest cover featured prominently as fourable candidates for sequestering carbon.

The greatest concern is that once management emphasis is geared toward carbon, the management objectives and practices will focused on enhancing tree growth, which may lead to manipulation of natural forests and replacement of natural shrubland with exotic trees such as Eucalyptus and pine. Monocrop plantations in Africa are not good habitats for wildlife, particularly elephants and primates. From Figure 5.10, it is clear that the entire southern part of the basin will diminish in terms of other ecosystem services.

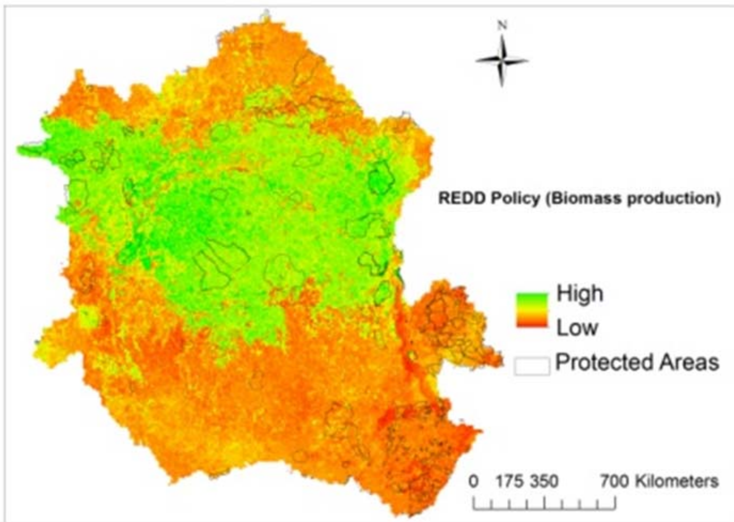

Figure 5.10 Single objective policy option experiments, REDD + Policy

Other initiatives include the Forest Law Enforcement Governance and Trade (FLEGT) program which aims to develop a system to improve the traceability and ensure legality of wood being exported to Europe. WWF and Mercy Corps are leading this initiative to build the capacity of forest and law enforcement agencies to conduct resource extraction monitoring in te region.

The third policy experiment was about biodiversity conservation, in which case, biodiversity values were rated highly. From figure 5.10, it is vividly clear that a biodiversity conservation policy would result in enormous gains in the ecosystem benefits throughout the entire basin. The central region, however, would register some minimal losses in terms of providing water and carbon values. It is possible to speculate that such areas would be recolonized by browsing and foraging species such as forest elephants, buffaloes, and several ungulates. Water quality and quantity would decline due to reduced land cover associated with increasing bumbers of wildlife and loss of large parcels of forests to savanna conversion would result in increased evapotranspiration as well as soil moisture.

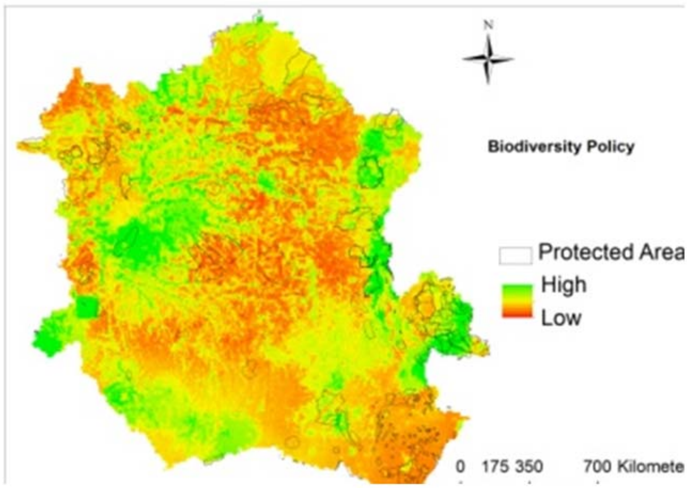

Figure 5.11 Single objective policy option experiment with biodiversity conservation

The results of the carbon sequestration policy showed no effect on water quality and quantity. The underlying assumption is that natural forest cover is maintained and the right mix of tree species is used for planting. Obviously, increase in forest cover is expected to result in reduced soil erosion, conservation of soil moisture, and high infiltration rate after a storm event. As such, stable stream flows and less runoff is expected. Intuitively, the reduction in overland flow is offset by the increase in subsurface flow that ends up as groundwater and water in reservoirs. The key to successful evaluation of policy measures is what decision-makers value rather than merely using what is easily measured. The results of the ecosystems services model is a key component in an integrated decision support system that allows the decision makers to evaluate alternative policies from an ecological point of view. Further analysis is needed, but with clearly defined land use plans, management objectives and realistic targets set. This process is made possible with the advent of Marxan and InVest tools to incorporate variables in the modeling environment that are mainly qualitative in nature and cannot be captured at a spatial scale.

5.6 Conclusions

Runoff in the Congo basin showed a high spatial variability. Areas of that experienced high runoff occur in subwatersheds with low vegetation cover, steep slopes and dense human settlement. Similarly, soil loss due to erosion occurs in areas with high slope (>=25%) that are also under cultivation without much soil and water conservation practices. Ecosystem services are high in the central part of the basin with relatively intact forests, including areas under protection. From this study, it is very clear that effort should be placed on areas already protected as they house high biodiversity values. Generally, water quality and quantity is compatible with carbon sequestration, but the latter reduces biodiversity values, particularly in agricultural and savanna dominated areas. Biodiversity values are relatively high inside protected areas and relatively remote and undisturbed areas. It is important to note that biodiversity values are derived from an ecosystem's suitability to support a diversity of species with respect to their ecological and environmental requirements. The entire study, particularly the chapter on elephant population dynamics attempted to account for declines in biodiversity due to legal or illegal harvesting, an aspect that is very important to consider.

In the analysis, only the removal of a suitable habitat due to land conversion, fires, flooding among others stressors were considered very important that did not constitute part of the main study. As such, it is not surprising to note that protected areas, which are expected to suffer less or gradual conversion and experience relatively less wildlife harvesting, showed high biodiversity values. In particular, maximizing rates of carbon sequestration could lead to potentially perverse conservation outcomes when silvicultural interventions that are good for timber and carbon are detrimental to biodiversity. In this case, forest types and tree species that are good for biomass production, but less favorable as habitats for wildlife such as birds, large mammals could easily be targeted for protection and use in restoration of impaired areas. It is also possible that when the management objective for the basin is entirely carbon sequestration, wildlife species that do not heavily depend on forests are reduced. The most affected species also happen to contribute a high proportion of wildlife biomass resulting in considerable decline in biodiversity values. On the other hand, if the plant species selected for restoration of degraded areas and for planting in farmed areas is carefully done under clear and enforced regulation, it could enhance biodiversity in those landscapes. Clearly, in the absence of regulatory constraints and maximization of carbon stocks becomes the sole goal of forest management, then the intensity of carbon farming management driven by markets of carbon credits could heavily reduce biodiversity values.

Ecosystem-based approaches can contribute to climate change mitigation and adaptation and to sustainable development more broadly. Spatial planning for ecosystem services at international, national and local levels will be an important component of ecosystem-based approaches. However, because not all elements of biodiversity are critical for ecosystem services, it is important to also target critical areas for protecting biodiversity

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CHAPTER 6

EXTRACTION OF COMMON POOL RESOURCES

6.1 Introduction

Managing natural resources for sustainability, that is, balancing current human wise use with opportunities for the resource use by future generations) is a key priority for managers and policy makers (Klare 2001; Palmer et al., 2004). Common-pool resources (CPR), however, have proved to be a challenge for managers and resource users to manage sustainably. The commons refers to institutional commons that entail government abstention from designating anyone as having primary decision-making power over use of a resource (Hess and Ostrom, 2001). Most common-pool resources have been simply defined as resources available to anyone, making them difficult to protect and easy to deplete (McKean, 2000). Common-pool resources systems include lakes, rivers, irrigation systems, groundwater basins, forests, fishery stocks and grazing areas which provide resource units such as water, timber, medicinal plants, fish and fodder. Resource degradation is a severe problem for common-pool resources in the developing world (Ruddle and Manshard, 1981; Randhir and Lee, 1996, Randhir & Thomas, 2000) both under governance by the state and communities and has profound effect on the sustainability of people' livelihoods. The situation has been exacerbated by overexploitation of the common pool resources to satisfy short term needs of people leading to deleterious impacts on the welfare of these economies and the future for natural resource protection.

Africa is the only continent to have grown poorer in the last 25 years (Commission for Africa, 2004) and is plagued with more conflicts than any other continent (HIIK, 2006) many of which are linked to unsustainable harvesting of natural resources and poorly regulated trade globally (Rainforest Foundation et al., 2005). Despite its endowment with natural resources, Africa is one of the world's poorest continents on planet earth (World Bank, 2008). World Resources Institute (2008) reported that three quarters of the world's poorest people live on less than $1.24 dollars a day and are highly dependent on the environment for a significant proportion of their livelihood needs. Corruption costs the continent US$ 148 billion dollars per year (African Union) and poor accountability for natural resources is a symptom of poor governance (Commission for Africa, 2004). Every year, US$ 10-15 billion are lost globally through illegal logging (World Bank, 2006). Weak governance and conflicts render sustainable management of natural resources a distant goal. Although African governments recognize the critical role common-pool resources play in economic development of the continent, the collapsing institutional frameworks coupled with poor management techniques in the governance of these resources by the state still remain a challenge. A study conducted by Forests Monitor (2007) in Democratic Republic of Congo (DRC) indicated that approximately 50,000 m^3 of timber from state governed forests are exported each year from eastern DRC, and to generate this volume of timber, 250,000 m3 of standing tree volume is cut.

6.1.1 Importance of the commons

The sustainability of common-pool resources becomes important as we consider economic and ecological services that are critical to human welfare, cultural identity, self governance, rights fulfillment, and wildlife needs. For example, FAO estimates that the world catch of fisheries is worth $80 billion annually (Carr, 1998), and Garcia and Newton (1997) estimated that 200 million people worldwide receive their income from fishing. Common pool resources provide direct benefits such as food, fuel, fodder, herbs, construction materials and income to millions of rural and urban dwellers. Common-pools resources provide a wide range of ecological services and functions enjoyed locally and globally. The ecosystem functions include habitat for wildlife, biological or system properties or process of ecosystems while the ecosystems services consist of flows of materials, energy, and information from natural capital stocks, which when combined with manufactured and human capital services support human lives. Contanza et al., (1997) estimated the average economic value of ecosystem services and natural capital in the world to be US$33 million and indicated that the real value could even double this amount. In Africa where annual rainfall is low and its distribution is erratic, the products obtained from CPR have been critical elements in the livelihood and survival of many rural communities, particularly in times of drought (Bernus, 1988). At the household level, reliance on common-pool resources have long been a method of assuring household subsistence during hunger periods and in resolving imbalances in the diets of rural households (Williams, 1998). In addition, the sale of products collected from CPR provides an important contribution to household incomes (Bush et al., 2004). Also, the spatial dispersion of common pool grazing resources and the temporal fluctuations in their availability make them an important resource for livestock production in the region. The seasonal use of CPR creates opportunities for mutually beneficial exchange relationships between various user groups, including opportunities to access financial services. The exchanges of grain, crop residue and water owned by farmers for the manure produced by pastoralists' livestock have linked crop and livestock production for many years in the Sahel and served to increase land productivity (Williams et al, 1995).

6.1.2 Nature of international commons

Common-pool resource systems under state control are characterized by two most important attributes, that is, they generate relatively substractable benefit flows and also very costly to exclude individuals from using the flow of benefits either through physical barriers or legal instruments (Ostrom, 1990; Dietz et al., 2002). It is difficult to develop physical or institutional means of excluding beneficiaries. The consumption of products or resource units from common-pool resources (CPR) by an individual, subtracts from the quantity available to others and the issue of scarcity makes common-pool resource problems more prone to conflict spirals. Consequently, common-pool resources are subject to problems of congestion, over use, pollution, and potential destruction unless harvesting or use limits are devised and enforced. It is worth noting that there are community owned common pool resources that have been successfully managed by the communities based on indigenous technical knowledge and values (Grafton et al. 2000; Agrawal 2001) as well as state managed common pool resources (Janssen et al. 2006).

Some of common-pool resources such as fisheries and wildlife are mobile and often cannot be confined to one place. The wide dispersion of these resources within the landscape and the variety of resource units they produce make them particularly useful to a diverse set of users. They are used at different geographic scales, offer multiple use values and benefits and often present management challenges, and weakens collective action among direct and indirect users. For example, local forest users derive benefits when the forest is used for timber production or harvest of non timber forest products where as global users of forests benefit from standing trees as carbon sequesters, a major global pollutant (Young 1999). Further, the use of common-pool resources often presents externalities to those who do not necessarily benefit from the use described above. According to Bruce (1999), timber harvesting may lead to deterioration of water quality downstream from the location where timber is harvested and yet those upstream continue to enjoy good quality water without compensating those polluted by their actions. Equally, water resources exhibits externality characteristics which defy full assignment of property rights (e.g. water seeps, flows, evaporates, and precipitates without regard to private, state, or national boundaries). Streams and groundwater aquifers simultaneously provide multiple benefits with varying degrees of tradeoffs between in-situ uses such as wetland preservation, transportation, and out-of-stream or off-site uses such as farm irrigation, power plant cooling, residential water services, which are further complicated by the public good nature of such benefits (Colby, 1995: p477).

At the landscape level, interstate or transitional commons become even more complex in defining their boundaries. Most local, state and regional boundaries are administrative in nature and purpose, conflicting with the geographical spread of the ecosystems making it difficult to allocate benefits and costs of resource protection. Further, managing environmental impacts associated with complex large-scale developments such as damming of rivers or conversion of forested watersheds to agricultural lands requires sound knowledge of the values and costs, and cooperation among stakeholders. Given the lack of commercial markets for most ecological services derived by the public both at the local and international level, and the valuation challenges they present (e.g. unknown future values, poor methodologies, and multiple values and downstream benefits a recipe for double counting), CPR are difficult to manage using market-based approaches (Caulkins et al., 1985; Bockstael et al., 1990; Ready, 1995; Graham-Tomasi, 1995; Bishop et al., 1995) and have not been managed through market-based systems

Common-pool resources and their users do not live in isolated communities lacking connections to the external world. As such, most users of CPR interact with other people in an institutional environment that is external to the one regulating the CPR and imposes constraints on the regime governing it. Consequently, the users are not only forced to seek external legal authorities to protect the institution governing the resource but also must accommodate to some degree the interests of the outsiders (as in geographic location) who have something to benefit from the CPR. Further, CPR differ from each other in regard to a wide variety of attributes such as size, ecological resilience and some exhibit high uncertainty in the flow of their stocks and future values. Common-pool resources, in many instances, were governed by national, regional, or local governments, by community groups, private individuals or corporations; or used as open-access resources by whoever can gain access. Each of the broad types of property regimes has different sets of advantages and disadvantages but at times may rely upon similar bundles of operational rules (Feeny et al., Supra note 20). This section was devoted to discussing the success and failures in the management of commons in Africa with a specific emphasis on watershed resources.

Global economic and regional population and development pressures have contributed immensely to high deforestation rates in the humid tropical forests (Achard et al., 2002; Coe et al., 2009). The annual deforested (10) area for the humid tropics is estimated at 5.8 ± 1.4 x106 ha, plus a further 2.3 ± 0.7 x 106 ha of forest where degradation was visually inferred from satellite imagery (Achard et al., 2002). Recent deforestation rates in the Murchison-Semliki landscape reflect the increasing regional demand for timber, and agricultural land to feed the growing human population. In Uganda, the recent report on the country's economic conditions and trends in population growth (UBOS, 2009) suggest that high rates of deforestation will continue well into the future unless measures are taken to slow down deforestation. The immediate solution, at least from the scientific community, is to generate knowledge about the importance of forests in terms of support functions and services, and make it available to management authorities and development partners, including local communities to influence political decisions.

The biota and physical structures of ecosystems (e.g. forests/savanna woodlands, and aquatic) provide a wide variety of ecosystem services such as carbon sequestration, maintenance of biodiversity, surface water and groundwater purification, nutrient recycling, and climate regulation of economic importance to human development. As such, there is an increasing attempt by environmental and ecological economists to include such services in the national environmental accounting systems (Costanza et al., 1997; Wackernagel et al., 2004; Monfreda et al., 2004). Economic valuation requires that ecosystems be described in terms of the goods and services they provide to humans, including existence values. Goods and services, in turn must be quantified and measured on a common (though not necessarily monetary) scale if comparisons between and among ecosystems, and improvements to one ecosystem are to be made.

6.1.3 Problem

Future management of common pool water resources is slowing shifting from wildernss fortress conservation o people-centered management options. Due to the high dependence of common pool resources, many rural poor households are economically and socially trapped by relying on natural resources for their welfare needs. The socio-economic factors and relations driving the depletion and degradation of common pool resources cannot be ignored. Part of the debate is how those who heavily depend on natural resources are likely to cope with the environmental challenges, particularly climate change. Water which is a very important resource in low income households is also getting scarce and sometimes one needs access rights in order to collect in it mainly from protected areas. In addition, there are signs of a transition from building new water supply systems to better operating existing ones in order to meet the growing demand for water.

Under these new models of conservation, the behaviors of communities that heavily depend on natural resources for their livelihood needs to be well understood. It is plausible to believe that many natural resource management issues still have the characteristics of a Prisoner's Dilemma game (Hardin, 1982; Fleishman 1988; Gardner et al. 1990) where those with authority and are well connected politically dominate the platform hence failing the strategy of building cooperation among the most affected. The prisoner's dilemma arises out of the circumstance that in the absence of communication between two prisoners held for the same crime and unable to make binding agreements, there is no way of rationalizing the choice of action, which if taken by both players benefits both (Hardin 1982). A prisoner's dilemma poses a conflict between the pursuit of one's individual gain and the pursuit of the welfare of a group to which one belongs. Given that the goal is a public good, someone who does not contribute to its provision can still enjoy its benefits, in effect getting a "free ride" on the contributions of others. This creates a situation in which, it is always more profitable for a person, regardless of what others do, not to cooperate (i.e., free-riding is the dominant strategy)

Research has shown that rural poverty is strongly associated with lack of land and livestock, as well as the inability to find alternative income sources (Ellis and Bahigwa 2003). In this study, it was argued that recognition of CPR user's entitlements and their dependence on CPRs provide a logic for cooperation and co-investment between stakeholders who are in a position to affect, or are affected by changes in natural resources quality and quantity. Understanding the livelihood strategies and the anticipated copying mechanisms in the face of dwindling natural resources is the fundamental underpinning of this research. The willingness to protect and contribute to the management of CPR entirely depends on how people value and benefit from them, and whether their self-governance and tenure rights are recognized. A number of studies have studied the dependence side, but none has attempted to understand how resource extraction is likely to change under changing climatic conditions.

6.1.4 Current approaches

6.1.4.1 Bureaucracy-based approach

Common-pool resources are managed under different approaches varying from state-governed, public resources, private such as logging, and fishing concessions that bans customary uses by indigenous peoples, community to no management at all, which Imperial and Yandle (1998) broadly classified as bureaucracy-based approach, market-based, and community-based approach. In the past decades,, the most economically and ecologically valued common-pool resources were typically managed by state agencies and to a limited extent under private property, however, community managed areas have demonstrated even greater economic and ecological viability mainly in the global south (Berkes 1999; Ford & Martinez 2000). The increasing area under community-oriented management is an implicit admission by central governments or provincial level administration in the case of decentralized governments that local community actors can govern their resources quite effectively when they have the opportunity to do so (Agrawal, 2007). Fisheries, wildlife and forests are some of the state governed commons under the protected area system, either as parks, wildlife reserves/sanctuaries, national wildlife refuge, marine fisheries protected areas and national forest reserves. In the Albertine rift's member states, wildlife is currently managed by state-controlled institutions: the National Institute for Nature Conservation in Burundi; the Congolese National Institute for Nature Conservation (ICCN) in DRC; Uganda Wildlife Authority (UWA) in Uganda, and the Office of Rwandan Tourism and National Parks (ORTPN). With the exception of nationally and internationally valued commercial fisheries and wetlands, commons in this category have been subject to all forms of management systems ranging from state controlled, co-managed, community conservation, private property to open access.

6.1.4.2 Community-based natural resource management (CBNRM)

Although the term CBNRM was coined in the 1980s, the notion that communities should, and could, satisfactorily manage their own resources according to their local custom, knowledge and technologies has a long history (Mamdani, 1996). In Africa, CPR typically include forests, open woodlands or grasslands for communal livestock grazing and wildlife management, wood supply, medicines, wild foods, wildlife for game meat and safari incomes, fish in freshwater lakes; surface and ground water aquifers, and irrigation systems. During the colonial period, the practice of indirect rule developed for which native institutions were reshaped for the purposes of meeting the rule of the colonial rulers, dividing the rural from the urban and one ethnicity from another, and forming an institutional segregation (Ribot, 1999, 2001). In the same way, the colonial governments went ahead to decentralize the management of natural resources by creating bureaucratic administrative units who derive the authority and power from the political constituencies resulting in the withdrawal of property rights, and subsequent collapse of voluntary service by the communities.

CBNRM was one of the new approaches for rural development paraded by international development institutions to recognize continued community governance of local resource users to participate in the management and conservation of natural environments. This is being pursued further to include the recognition of indigenous peoples’ rights to natural resources. The advocates of this management approach assumed that communities, defined by their tight spatial boundaries of jurisdiction and responsibilities, and by their distinct and integrated social structure and common interests, can manage their natural resources in an efficient, equitable, and sustainable way (Blaike, 2006). Unfortunately, the new institutional arrangements, in many ways, were and continue to be neglected by their initiators and promoters, except for purposes of political and strategic control, labor mobilization and latterly for soil and water conservation. This broad concept of community participation or involvement in natural resource management has given birth to a number of techniques being used in various African countries such South Africa, Namibia, Tanzania where community conservation (Hackel, 2001; Berkes, 2004), for parks and wildlife reserves, collaborative forest management, social forestry, integrated conservation and development programs for both wild and agricultural lands, and Beach management Units for commercial fisheries are being implemented.

Building on experiences from India (Poffenberger and McGean 1996; Kothari et al. 1996), collaborative forest management (CFM) was adopted in Uganda in 1993 around Bwindi Impenetrable National Park, BINP (Wild and Mutebi 1996), and by 1996, collaborative initiatives had spread to other protected areas (national parks) such as Mt Elgon, Kibale, Mgahinga, and Murchsion Falls (UWA 2001). In the forest sector, research on CFM began in 1996 with pilot activities in some selected Ugandan forests, for example Butto-Buvuma (Gombya-Ssembajjwe and Banana, 2000). The Forest Department, however, held consultations from 1996 to 1997, and on July 1998, the CFM programme was officially launched (Scott 2000). Since then, pilot activities were initiated by the Forest Department, emphasizing equitable distribution of benefits, participation of local people at all stages, gaining consensus on the terms of management and representation; instilling the sense of ownership and authority over the resource in local management partners, ensuring flexibility on the part of the Forest Department towards the potential compromise and building mutual trust and respect as a strong foundation for future partnership.

CBNRM has had its failures, some of which stem from assumptions, misunderstandings, and expectations associated with certain interpretations of the word “community” where “community’’ is difficult to define, and understood in many different ways (Agrawal and Gibson, 2001). A more powerful notion in the label of CBNRM is the elision of the notion of sustainable natural resource management (defined by rational and scientific criteria) with ‘‘community,’’ implying the management vehicle, is well suited for the task with its local ownership and indigenous expertise. Therefore, the communities are supposed or expected to be able to deliver on scientifically specified natural resource management principles (as defined by the elitist outsiders). Herein lay the first confrontation between formal science with its foundations in logical positivism and the independence between the observer and the observed on one hand, and local knowledge, which is embedded in particular environmental and social histories and continually negotiated on-site and face-to-face on the other hand (Kates, 2000; Usher 2000; Lynam et al.2007). It was envisioned that both decentralization and participation will help bridge this gap. Unfortunately, what has emerged from these contradictions is the divisive practice of CBNRM (its production, representation in policy documents and implementation, power structures, accountability, confidence and political sophistication by local institutions), which is situated at the interface between community, government, private business and other outside institutions (non-governmental organizations) (Brosius et al. 1998; Blaikie 2006).

Like any other crisis management approach, CBNRM approach has failed to deliver in terms of its stated aims (Campbell et al., 2001; Shackleton et al., 2002) concluded that “most devolved natural resources reflects rhetoric than substance” and that “the ways in which local people realize the benefits of devolution differ widely, and negative trade-offs, mostly felt by the poor are common” There are success stories too (e.g. CAMPFIRE project in Zimbabwe), although these stories have been told by initiating agencies themselves, since then, they have been both widely criticized (Sullivan, 2001) and broadly vindicated (Leach et al. 1999; Arntzen et al., 2003; Cox et al. 2010).

Community-based conservation was based on the idea that if conservation and development could be simultaneously achieved, then both objectives could be achieved. Conservationists and wildlife managers now strongly recognize the fact that ecosystems are complex (Levin 1999) and cross-scale approach to conservation is inevitable. Thus, matching the scale of management to the scale of the system to be managed and envisioned and implemented at the local level is very important. The core fundamentals to community conservation have been clearly articulated as the recognition and enforcement of property rights, particularly of the local communities with *de facto* rights and the sharing of management power and responsibility (Berkes, 1999). In addition, equity and empowerment are more effective than monetary incentives in community-based conservation because they guarantee legitimacy and accountability (Agrawal and Gibson, 1999), and the use of traditional ecological knowledge is fundamental step toward co-management and empowerment.

6.1.4.3 Privatization or market-based approach

Privatization is one of the new approaches for coping with the problem of rationing access to the commons, which involves the use of tradable permits. A market-based approach, privatization emerged as an important policy tool especially during the early 1980s and 90s (Yandle and Dewees, 2003). Tradable permits were perceived as a remedy for the problems of commons in rationing access (allocation of allowable annual rights or quotas) and privatizing the resulting access to the resource. It has been applied in air and water pollution control, land use control and in fisheries management. Privatization is applied in the context of optimal allocation of a resource and by extracting the principles that can be used to design economic incentive policies that fulfill the optimality conditions (Tietenberg, 2002). It relied heavily on the development of markets for user permits (e.g. total allowable catch quota, appropriation) and environmental taxes. In addition, it was assumed that the private property owners will have an incentive and the information necessary to devise optimal rules and enforce them (Ostrom et al., 1994:p18). Experiences in the United States of America concerning the market-based approach to allocation of property rights over water continue to generate conflicts (Colby, 1995) and in places where some success has been made (e.g. western states), water resources are highly regulated resulting in high transaction costs (Anderson and Johnson, 1986; Colby, 1995). While market transfers have become an increasingly common means to reallocate water, such transactions have been characterized as no free markets. As a result, policymakers cautiously explore markets only after the inefficiencies arising from their absence become unacceptable.

Tietenberg (2002) argued that it is difficult to privatize common-pool resources in the real and messy world when property rights are not easily defined and enforced, a prerequisite for efficient market functioning. The arrival of markets and new technologies, and the changes they might prompt in the existing natural resource management regimes, is not a blameless process (Oates, 1999). The key issues questioned are its ability to deliver result efficiency in terms of environmental and economic effectiveness. As local economies become better connected to larger markets and common property systems confront cash exchanges, subsistence users are likely to increase harvesting levels because of the shift to commercial resource off-takes. Thus, sophistication in business networks, technologies and the related increase in transaction costs make this approach even more uncertain in delivering sustainable management of CPR.

6.1.5 Justification of the study

Full accounting of the consequences of actions on the value of ecosystem goods and services requires concrete knowledge of the spatial links, and integrated approach at suitably large spatial scales to explicitly cover all important effects. Analysis of the role forest income and subsistence use play in rural people's livelihoods demand that present assumptions about the underlying resource user behaviors, ecosystem, and market conditions, and how estimates of benefits could change with changes in these underlying conditions is well understood.

6.1.6 Uniqueness of study

6.1.7 Objectives

The aim of the study was to improve our understanding of the level of dependence by households on common pool resources and its contribution rural livelihoods and how these benefits might be affected by climate change. The specific objectives are:

1. Assess the extent to which rural households in Greater Virunga Landscape depend on income and subsistence use from the extraction of common pool resources;
2. Identify the key income streams and their contribution to the livelihoods of rural households in Greater Virunga Landscape;

6.1.8 Hypothesis

The Null hypotheses **we**re:

1. Communities living adjacent to common pool resources do not heavily depend on common pool resources for their livelihoods;

2. Protected area frontline communities do not depend on a single source of income for their livelihoods

6.2 Literature Review

The most important challenge in governing common-pool resources lies in the fact that stocks and flows of these resources are quite often difficult to define accurately (Dolsak and Ostrom, 2003). The other challenges include the deliberate attempt by the national governments to centralize the governance and management of CPR, only worked to undermine and disintegrate the traditional institutions and authority. The increased mobility of members and centralization of local CPR resulted in the breakdown of these traditional systems, leaving inefficient institutions. Property regimes had long-run objectives in resource management such as common decision making by community trusts and user groups, regulation mechanisms based on cultural and religious norms, and traditional values (e.g. taboos, totem, sacred), voluntary supervision, and collective decisions regarding agricultural practices such as soil and water conservation, soil fertility management, pest and disease management which had impacts on ecosystems. For example, changes in management from collective action to centralised/state controlled led to the break down in the institutional arrangements originally used by the communities (e.g. norms, values, beliefs) to regulate entry and the use (rights of access, ownership, voluntary management and alienation) of forest products.

In some instances, forest management became centralized or privatized affecting the property rights, benefits, roles, and responsibilities. Changes in property regimes also meant changing the management systems, objectives and user groups and values. CPR users that did not heavily depend on forests shifted to on-farm and off-farm related activities leaving mainly the poor to negotiate the user rights and access. CPR controlled and managed by government agencies, exhibit some open-access regimes, hence lack effective rules defining property rights. Such CPR managed in public trust got affected by these open-access regimes contained within unclear state or co-managed arrangement because no entity has successfully laid claim of legitimate ownership. Other open-access regimes were created as a consequence of conscious public policies (e.g. decentralization of forestry sector leaving management to poorly financed, inadequately staffed and incapacitated departments or community groups; equity but not efficiency, increased agricultural production with emphasis on area rather than yield per acre; inconsistent energy policies that encouraged charcoal production) to guarantee the access of all citizens to the use of a resource within a political jurisdiction (e.g. and elective politics, poor policies). On the other hand, Dasgupta (1996) noted that the majority of common-pool resources have a defined set of users and a management system in place. In most cases, such resources are only open to those having historical rights through kinship or community membership and they are generally protective of these resources. Therefore, over-exploitation occurs when the management system breaks down allowing free riders to ignore the rights of other individuals and open-access regimes result from the ineffective exclusion of non owners by the entity assigned formal rights of ownership (Dasgupta, 1996).

In Uganda, community wildlife areas were recognized by the state and these were defined in the Statute as areas 'in which individuals who have property rights in land may carry out activities for the sustainable management and utilization of wildlife if the activities do not adversely affect wildlife and in which the State may prescribe land use measures'. The statute also acknowledged the rights of the communities to utilize community hunting areas. Community Hunting Areas (CHAs) were declared in areas where there were important wildlife populations, regardless of whether there were people living in these areas or not. Other than the hunting of scheduled species, the game department had no control whatsoever over any form of landuse in CHAs, such as settlement and cultivation. The concept of Controlled Hunting Area is now considered defunct in conservation terms because in many CHAs wildlife habitats have been converted to agriculture and biodiversity has been reduced. Furthermore, hunting in Uganda was banned in 1979, and therefore CHAs have no regulatory function. Nevertheless, the CHAs continue to be perceived as 'protected areas', and the Statute provides for a 24 month period in which the Minister should decide "which controlled hunting areas shall be declared as national parks, wildlife reserves, wildlife sanctuaries, community wildlife areas or any other area.... [and which CHAs] shall cease to exist."

Prior to the colonization of Uganda by the British at the turn of the 19th century, some parts of the country had well developed monarchy system, notably the kingdoms of BUganda, Bunyoro, Ankole and Toro (Were and Wilson 1970). Forests formed part of the land that was owned by the different kingdoms at the time. Within kingdoms, forests were either communally owned or used as an open access resource (Gombya-Ssembajjwe 1995). Communally owned forests were those adjacent to communities. People utilized them for wood and non-wood forest products. Communities had informal or traditional ways of managing forest resources. Controls on some forests considered sacred were presided over by a person whom the community accepted as a caretaker and believed to be possessed by the spirit after which the forest was named. The members of the clan were free to collect forest produce for domestic use such as firewood, grass for thatching and clay for making pots. Sacred controls were used to regulate the use the forests by traditional beliefs and penalties for misuse were sanctioned onto the offenders. There were no written rules to describe forest management; instead the community grew up knowing the 'dos' and 'don'ts' in relation to forest use which were passed on to future generations through oral instructions and cultural traditions (Banana et al. 2008). For example, it was believed that if a person went to the forest without reporting the purpose of the visit to the clan head or the elders or changed the purpose in the forest, he/she would get lost in the forest (Gombya-Ssembajjwe 1995).

Elders continually reminded the community about the dangers that awaited those who violated the rules. In some cases, threats of spiritual punishment were very scary and an effective deterrent to forest offenders. They included the offender's home being afflicted by insect invasions, crop failure and infertility and relatives of the offender had to cleanse the wrath of the spirits by making religious offerings (Turyahabwe and Banana 2008). Penalties often included returning to the forest whatever product had been removed and fines in form of animals or food. On the extreme, violators were chastacised or made to wear clothing with distinctive color (e.g. pink). Offenders who failed to comply with penalties were cursed by the elders and considered out-casts and despised members of their communities (Gombya-Ssembajjwe 1995). Those obeyed the rules of conduct, received gifts or elevated to positions of social status. These rules of control achieved considerable success in preventing overexploitation of the forest resources and ensured conservation of forests at that time. The continuation of such a management regime could have probably helped to guarantee the conservation of forest resources had it not been the introduction of scientific forest management by the colonialists in 1898 (Ndemere 1997).

Failure by the governments to formalize the legal status of the common-property regimes that had evolved over time, the concern for the protection of natural resources mounted during the second half of the last century, forcing many developing countries to nationalize all land and water resources that had not yet been recorded as private property. As a result, institutional arrangements that local users had devised to limit entry and use lost their total legal standing. The national governments that declared ownership of these natural resources, however, lacked monetary resources and personnel to exclude users or to monitor the harvesting activities of users. Thus, the resources that had been under a de facto common-property regime enforced by local users were converted to a de jure government-property regime, but reverted to a de facto open-access regime (Bruce et al., 1993; Gershon and Feeny, 1993). Implicitly, theorists assumed that regulators will act in the public interest and understand the ecological systems work and how to change institutions so as to induce socially optimal behavior. Regardless of the low capacity of the appropriators to communicate, coordinate their activities, and to create institutions to allocate property rights and make policies that conform to jointly owned/managed forest resources, states assumed that the resource user groups still had the incentives to manage the forests. There is growing concern, however, from many common-pool resources studies by anthropologists (Netting and McC, 1982) and historians (Glick, 1986) calling for a serious re-thinking of the theoretical foundations of such analysis of CPR.

The lack of a clear regulation of the bundles of rights (e.g. access, extraction/use, exclusion and alienation) for authorized users, (in particular for users who enjoyed both entry and extraction/withdrawal use rights) varied substantially in regard to the quantity, timing, location and use of the resource units appropriated from a common. The presence or absence of constraints upon the timing, technology used, purpose of use, and quantity of forest products harvested, greatly contributed to the loss or degradation of commons. Bruce (1995) accurately described that the constraints were usually determined by operational rules devised by those holding the collective choice rights (authority) of management and exclusion over the resource system. The deliberate attempt by the states to centralize government authority over commons destroyed long-standing traditional systems of managing them, which offered voluntary restraint among the resource users in small organized communities without providing alternative viable arrangements. These modified administrative systems have evolved undesirable traits like heavy bureaucracy and corruption, lack of reverence for people's participation and callousness to conservation needs. Ownership of resources was shifted from communities, individuals to the state and where co-management has been fronted; the roles, responsibilities, revenues and rights still remain ineffective or ambiguous (Berkes 2004; Cox et al. 2014).

One major complication to achieving effective natural resource management is human population increase coupled with changing life styles and cultural integration. The world population growth rate has been reported to be 1.3 percent per annum (United Nations, 2000) and Palmer et al. (2004) estimated that the population will reach 11 billion in the next 100 years. As the human population increases, so is the increase in demand for natural resources to provide goods and services. Studies by Ehrlich and Holdren (1971), Cleaver and Schreiber (1994), and Dasgupta (1992, 2003) have shown that population size contributes heavily to the overall demand for goods and services consumed from the environment, which in turn depends on existing institutions and the knowledge base, and the technologies that are thereby in use to exploit those resources. A combination of poverty and population growth, however, is the key driver of environmental degradation and natural resource loss or decline. For example, during the period 1980-1998, Sub-Saharan Africa experienced an annual population growth of 2.8 per cent contributing to annual forest loss of 0.8 per cent by area to deforestation from 1990 to 2000 (Dasgupta, 2003). World Bank (2001) reported that 5-12 million hectares of land are lost annually to severe degradation, and that soil degradation affects 65 percent of African croplands and 40 percent of croplands in Asia. Additionally, over 70 percent of freshwater sources are contaminated or degraded and yet groundwater withdrawal in poor countries exceeds natural recharge rates by a phenomenon 160 billion cubic metres per year (Dasgupta, 1998, 2003).

Changes in the market economy, involving policy shifts such as liberalization, capitalism, and the establishment of formal financial institutions in rural areas, to a large extent has impacted the CPR management substantially. Changes in the physical, human, and social assets that arise from micro-credit activities affect a community's production, consumption, and management opportunities and decisions around CPR (Anderson et al., 2003). In particular, the effects of changes in consumption activities afforded by increased income on the environment have been discussed (Dasgupta and Máler, 1994; Sebstad and Chen, 1996) and all seem to agree that the relationship is complex. For example, linking microcredit access to environmental goals in the form of promoting green products or ecofriendly economic activities among its loan members may trigger off both positive and negative behavioral responses, and social-capital flows. Resource user groups may become interested in owning forested lands so they can access financial loans, profit from business companies interested in large quantities of natural resource products such as timber and fish. In addition, financial credit allows micro-entrepreneurs to diversify their income base by investing in small-scale capital enterprises such as livestock, tools, fishing boats, which may free people from the CPR or increase their consumption by creating new networks, build social capital, and craft new rules of engagement. These new property regimes, however, become a driving force in influencing non-market mechanisms for transaction of businesses and exchange of goods derived from the CPR. In most cases, the harvesters shift to more sophisticated harvesting technologies, in particular mechanized systems leading to excessive waste and ecosystem degradation. The situation becomes more accentuated by failures in markets, in which case, more resource units have to be harvested in order to gain the same amount of money earned before the decline in world prices for valuable products. Little known or ignored, all these changes create rapid declines in the resource system and outpace the capacity of the community to manage, create resource user conflicts demanding multidisciplinary management strategies.

The gradual erosion of traditional systems due to external influences and modernization resulted in the breakdown of property rights and created the mismanaged commons. Modernization has resulted in the breakdown of individual trust and development of exploitative members among the common property users. Indigenous technical knowledge was deemed primitive and archaic, cultural values became shaped by those who held the authority over CPR. In the long-run, the collapse of interest in communal forests affected voluntary participation influencing the resource user's attitudes and practices. In much of Africa, public policies and government actions do not always, and in some cases rarely reflect the high priority concerns of rural people who comprise about 65 percent of the continent's population (Viet et al., 2008). As such, government policies and practices often favor the interests of small, powerful groups of political and economic elite. Socio-political crises and wars since 1993 have led to enforced migration, a refugee crisis, insecurity and deforestation, resulting in irreversible negative impacts on biodiversity. Over three million refugees lived temporarily in or around Virunga National Park between 1994 and 2002, leading to high biodiversity loss, and these problems remain today. More than 4.4 per cent of the protected area has been transformed into farmland (FAO, 1996; Laporte et al., 1998; Mayaux et al. 1998). There has been little staff training, no replacement (despite high losses of personnel) and there is still limited donor aid for biodiversity conservation.

Deforestation through timber logging and poor agricultural practices (e.g. slash and burn) has contributed immensely to watershed degradation resulting in low quality water. Further, the natural resources in the African Great Lakes region have long been interlinked with conflicts, instability and structural violence. This relationship is set against a landscape beset by corruption, porous borders and weak government, which serve to undercut legitimate institutional arrangements and undermine conservation of wildlife and sustainable economic development. Considering the value and nature of the resource, as well as perceived rights of ownership, the continued trade in natural resources under current practices is eliminating alternative future development options. It remains essential that environmental sector and trade reforms take place urgently given the currently unsustainable rates and inequitable nature of exploitation of the forested watershed resource base and distribution of the ensuing benefits.

In general, examples exist of both successful and unsuccessful efforts to govern and manage common-pool resources by governments, communal groups, corporative, voluntary associations, and private individuals or firms (Bromley and Cernea, 1989; Eggertsson Thrainn, 1992; Katar, 1994; Katar and Ballabh, 1996; Randhir and Lee, 1996). Under successfully managed commons, devising property regimes that effectively allow sustainable use of common-pool resources require one set of rules that limit access to the resource system, and other rules that limit the quantity, timing, and technology used to withdraw diverse resource units from the resource system (Baland and Platteau, 1996). Related to the resource itself, the following characteristics have been identified as conducive to successful governance of such resources; small size, stable and well-delineated resource boundaries relatively small externalities resulting from use, ability of resource users to monitor resource stocks and flows, moderate levels of resource use, and well-understood dynamics of the resource by the users (Wade, 1996; Baland and Platteau, 1996; 2003; Ostrom, 1990; Dolsak and Ostrom, 2003).

6.3 Methodology

6.3.1 Study area

Greater Virunga Landscape (GVL) is a very important landscape for conservation of wildlife and wild habitats (Plumptre et al. 2007; Plumptre et al. 2014) (Figure 6.1). Greater Virunga Landscape lies within the Albertine rift, a region characterized by steep mountain ranges, including the volcanoes, which stretches from the northern end of Lake Albert to the southern end of Lake Tanganyika in Zambia, extending through the countries of Uganda, Rwanda, Democratic Republic of Congo, Burundi and Tanzania. The Albertine Rift is one of Africa's most important sites for the conservation of biodiversity (Plumptre et al. 2002; Plumptre et al. 2003; Plumptre et al. 2007). This region is the most species rich region in Africa for vertebrates and contains many endemic and threatened species. The Albertine Rift was identified as an Endemic Bird Area by Birdlife International, an ecoregion by the World Wide Fund for Nature (Olson et al., 2001), and was named a biodiversity hotspot by Conservation International (Myers et al. 2000; Brooks et al., 2004).

The study region is also ethnically and culturally diverse (Jackon, 2006). More importantly, ethnic citizenship determines entitlement to customary power, which in turn governs the allocation of land (Jackson, 2006). There are over 200 ethnic groups in Democratic Republic of Congo that belong to four major groups namely the Bantu peoples in Democratic Republic of Congo (Mongo, Luba, Kongo, and Mangbetu-Azande peoples) who constitute around 45% of the population. Other Bantu ethnic groups documented include Hunde, Nyanga, and Nande in North Kivu, Hutu predominantly from Rwanda and Burundi). The second major groups area the Nilotes (e.g Tusi and Hima), Hamites, and the Luo. Nearly 600,000 Pygmies are the aboriginal people of the DR Congo. Although 242 local languages and dialects are spoken, the linguistic variety is bridged both by widespread use of French and intermediary languages such as Kongo, Tshiluba, Swahili, and Lingala. The population of Democratic Republic of Congo is nearly 75 million people (World Bank, 2013). In Rwanda, the largest ethnic groups are the Hutus (about 75% of the population), the Tutsis (24%), and the Twa (1%) while in Uganda, the four major ethnic groups are found in Uganda. Bantu peoples are the most numerous and include the Baganda in the central area (17%), Banyankole in the south-western area (8%), Bakiga in the most south-western area (8%), Banyoro in the mid-western area (3%), Batooro in the mid-western area (3%), Bahima in the south-western area (2%), Bafumbira in the south-western area (6%), and other much smaller ethnic groups such as Bakonzo, Basongola, Bamba, and Banyabindi. The other tribes outside the study area include the Basoga in the south-eastern area (10%), Bagwere in the eastern area (4%), Bagisu in the eastern area. The Nilotic peoples, mainly in the north, are the next largest, including the Langi, 6%, and the Acholi, 4%. In the northwest are the Lugbara, 4%. The Karamojong, 2%, occupy the considerably drier, largely pastoral territory in the northeast. Europeans, Asians, and Arabs make up about 1% of the population with other groups accounting for the remainder. The population of Ugandans in the regional is about three million people out of the total national population of 36.2 million people.

This study, however, focused on the GVL that encompasses a number eight protected areas ranging from forest to savanna woodland and grassland national parks, and is known for habiting the only remaining 800 mountain gorillas (Gorilla beringei beringei) in the world. A detailed this description of this area is provided in chapter four of this thesis, however, a brief description is provided here. Greater Virunga Landscape (GVL) overlaps international boundaries of Democratic Republic of Congo, Rwanda, and Uganda (Figure 6.1). The GVL is one of the most biodiverse landscapes in Africa (Plumptre et al., 2007;) that comprises many habitat types ranging from high altitude montane forests and alpine at 5100 m on top of mountain

Rwenzori to lowland forests at 600 m above sea level (Plumptre et al. 2007; Plumptre et al. 2014). GVL covers an area of 15,700 km^2, including three world heritage sites, one UNESCO biosphere reserve and one Ramsar site; and twelve protected areas accounting for 88% protection (i.e. 13,800 km^2) (Plumptre et al. 2007).

In this study, the area was expanded to 20,000 km^2 to include areas of importance to securing the integrity of protected areas and incorporate the influence of human activities conducted outside the protected areas. It is home to 1409 terrestrial vertebrates (mammals, birds, amphibians and reptiles, of which 100 species are endemic to the Albertine Rift and 56 species are globally threatened) and 3755 plant species and at least 100 fish species (Plumptre et al. 2007; Plumptre et al. 2014). It is worth noting that this landscape is rich in primates, specifically chimpanzees (Pan troglodytes), le'hoest, and white and black colobus monkeys, red tail monkey, and blue monkey, It is also know for one of the large populations of African elephants (*Loxodanta Africana Africana*) in the region and the tree climbing lions (*Panthera leo*) that occur under low densities but require vast amounts of area for survival. The region is characterized by very low to high (80-600 persons per km^2) population densities (Uganda Bureau of Statistics, 2002; World Bnak, 2012).

Initially, all conservation efforts were geared toward protecting the Mountain Gorillas and an International Gorilla Conservation Program (IGCP) was initiated. However, the conservation challenges increased and demanded a concerted effort from member countries that shared the transfrontier resources. As such, a number of initiatives were established to strengthen the management and conservation of GVL. Greater Virunga Transboundary Collaboration (GTVC) is one such mechanism for strategic, transboundary, collaborative management of the Greater Virunga Landscape. GTVC was established by government conservation agencies, that is, ICCN (Democratic Republic of Congo), RDB (Rwanda) and Uganda Wildlife Authority (Uganda) with their partners in the region. The GVTC is guided by a ten-year strategic plan, which was developed in 2006 and revised in 2011 (Transboundary Core Secretariat 2006). It started with ranger collaboration to protect mountain gorillas in Mgahinga, Bwindi, Virunga and Volcanoes national parks in 1991. Later, it expanded its scope to include tourism, community conservation, and research and monitoring. The area also now extends to central and north Virunga (DRC), Queen Elizabeth, Rwenzori Mountains and Semuliki NPs (Uganda). Implementation of the strategic plan is guided by a five-year implementation plan, of which the main goal is to improve conservation of species, habitats, and ecological services contributing to increased socio-economic benefits, through effective transboundary collaboration.

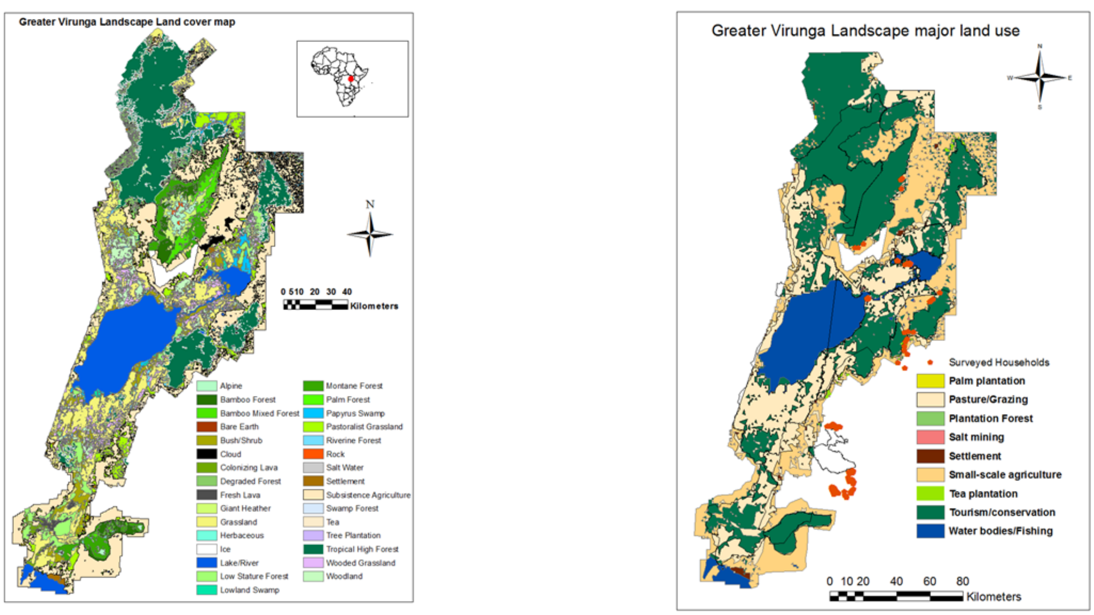


Figure 6.1 Map showing the location of GVL in the Albertine Rift in Eastern Africa

6.3.2 Theoretical framework

Many policy innovations of the 1960s and 1970s were based on the early work of resource economists and consistent with Hardin's thesis that "freedom in a commons brings ruin to all" (Hardin, 1968:1244), which assumed that all commons are open acess pool resources and has been a subject of considerable debate overlooking the collective governance of commons. The literature stressed the importance of unitary ownership including privatization as well as government ownership. However, the major policy innovation of this era was legislation in many countries, particularly developing countries that transferred forests, pasture land, in-shore fisheries, and other natural resources from their previous property rights regimes to government ownership (Arnold and Campbell, 1986). Extensive research and experience since 1968 showed that these transfers of property rights were sometimes disastrous for the resources they were intended to protect and disastrous for the people and the cause of massive human rights violations. Instead of creating a single owner with a long-term interest in the resource, nationalizing common-pool resources typically led to 1) a rejection of any existing indigenous institutions making the actions of local stewards to sustain a resource illegal; 2) poor monitoring of resource boundaries and harvesting practices because many governments did not have the resources to monitor the resources to which they asserted ownership; and 3) *de facto* open access conditions and a race to use of the resources. Thus, the presumption that government ownership was one of two universally applicable solutions to the tragedy was seriously challenged by these historical experiences (Agrawal 2007; Chapin et al.2009).

Ciriacy-Wantrup and Bishop (1975:715) and Thompson (1975) stressed that where common property existed, users had developed rich webs of user rights and responsibilities that identified who had a long-term interest in the resource, and in many cases, who had the responsibility to manage it, and thus an incentive to try to avoid overuse. Therefore, before intervening, the specifics of a particular common property regime had to be examined to avoid violating local customs, and impose a new set of rules that were unlikely to be viewed locally as legitimate. There is emerging evidence from field studies in Africa, Latin America, Asia and the United States that policy reforms that transformed resources from governance as common property by local communities into state governance were actually making things worse for the resource as well as the users (Dietz et al. 2002: p13). The governments that took these actions frequently did not have enough trained personnel on the ground to monitor resources and in many cases had little or no commitment to conservation at the same time as they had strong interest in extractive development. Thus, what had happened to de facto common property with some limitations on access and use patterns became de jure government property but due to the lack of enforcement, it frequently became de facto open access or not open access -- as indigenous peoples, local communities, and other customary users might find that their continuing use (including subsistence use) was criminalized, while access was made available by the state to certain individuals and corporations only (e.g. through concessions).

Corrupt public officials also faced opportunities to collect side payments from local resource users, corporate national and international extractive resource developers wishing to exploit resources that were officially government owned, making it to appear to be legitimate. Dietz and Ostrom (2002) noted that the best tool for sustainable management of a common-pool resource depends on the characteristics of the resource and of the users. Multiple institutional strategies are needed given the wide diversity of threatened physical and biological resources. It requires substantial ingenuity to design institutions that cope effectively with the attributes of a particular resource given the larger macro-political institutions, culture and economic environment in which that resource is embedded, including working with indigenous peoples and local communities to support their adaptation of customary ones.

An institutional arrangement is considered economically efficient if no reallocation of resources will improve the welfare of some individuals affected by the resource without making someone else worse off. Resource sustainability is attained when there is continuance (or even improvement) of the resource system, facility, or stock that generates the flow of resources units. On the other hand, an institution is regarded sustainable if there is continued use of the institution over time with adaptation occurring in the day-to-day rules within the context of a stable constitution. National governments of nearly all developing countries have turned to local-level common property institutions in the past decade as a new policy thrust to decentralize the governance of the environment (Agrawal, 2002. p41). This shift in policy is no more than a belated recognition that sustainable natural resource management can never be independent of sustainability of collective human institutions that frame resource governance, and that local users are often the ones with the greatest stakes in sustaining the resources and institutions.

Studies by Wade (1994), Ostrom (1990) and Baland and Platteau (1996) have provided empirical insights into what makes commons successful. Wade's (1994) important work on commonly managed irrigation systems in South India, noted that effective rules of restraint on access and use are unlikely to last when there are many resource users, when the boundaries of the common-pool resources are unclear, users live in groups scattered over a large area, detection of rule breakers is difficult, and when external government (high level) interference exists. According to Ostrom (1990), the design principle (refers to clearly defined boundaries and group membership) is an essential element or condition that helps account for the success of these institutions in sustaining the common-pool resources and gaining the compliance of generation after generation of appropriators to the rules in use. At a wide-ranging review of empirical studies of common-pool resource management, and focusing on several variables that existing research has suggested as crucial to community-level institutions, Baland and Platteau (1996) arrive at conclusions that significantly overlap with those of Wade and Ostrom. Small size of a user group, a location close to the resource, homogeneity among group membership, effective enforcement mechanisms, and past experiences of cooperation are some of the themes they emphasize as significant to achieve cooperation (Baland and Plattaeu 1996; Agrawal, 2007; Cox et al. 2014). These studies, however, lack careful analysis of the contextual factors that frame all institutions and that affect the extent to which some institutions are more likely to be effective than others (Agrawal 2002). For example, the same institutional rules can have different effects on resource governance depending on variations in the biophysical, social, economic and cultural contexts. It is therefore important that current studies examine how aspects of the resource system, some aspects of resource user group membership, and the external socioeconomic, political, physical and institutional environments affect institutional durability and long-term management at the local and international level.

A number of factors that were considered very instrumental in accounting for the success in resolving institutional dilemmas in the sustainable management of common include, rules are simple and easy to understand (Baland and Platteau 1996), locally devised access and management rules (Wade 1988; Ostrom 1990; Baland and Platteau 1996), ease in enforcement of rules (Wade 1988; Ostrom 1990; Baland and Platteau 1996) , graduated sanctions (Wade 1988; Ostrom 1990), availability of low-cost adjudication (Ostrom 1990), and accountability of monitors and other officials to users (Ostrom 1990; Baland and Platteau 1996). Among other conditions, the external environment in terms of technology such as low-cost exclusion technology (Wade 1988) and the state involvement, central governments should not undermine local authority (Wade 1988; Ostrom 1990), but offer support to decentralized institutions (Baland and Platteau 1996). In addition, central governments are supposed to put in place appropriate levels of external aid to compensate local users for conservation activities and nested levels of appropriation, provision, enforcement, governance (Ostrom 1990).

The set of factors identified by Wade, Ostrom, and Baland and Platteau are relatively deficient in considering the resource characteristics. Even if the climatic and edaphic variables that may have an impact on levels of regeneration and possibility of use are not considered, there are grounds to believe that other aspects of a resource may be relevant to how and whether users are able to sustain effective institutions (Netting 1981). After examining the impact of these two physical characteristics of resources on externalities, Blomquist et al (1985) concluded that mobility and storageability have an impact on management because of their relationship to information. Greater mobility of resources and difficulties of storage make it more difficult for users to adhere to institutional solutions to common-pool resource dilemmas because of their impact on the reliability and costs of information needed for such solutions (Ostrom 1990; Naughton-Treves and Sanderson 1995). There is some degree of understanding among scholars that external social, institutional, and physical environment impact strongly on the management of commons. Demographic pressures resulting from changes in local population and migrations contribute heavily to resource degradation (Fischer 1993; Hardin 1998). There is also wider agreement that increased integration in markets has had an adverse impact on the management of common-pool resources, especially when roads begin to integrate distant resource systems and their users with other users and markets (Chomitz 1995; Young 1994). Carrier (1987), and Colchester (1994:86-87) noted that as local economies become better connected to larger markets and common property systems confront cash exchanges, subsistence users are likely to increase harvesting levels because they can now exploit resources for cash income as well.

Market integration has been documented (Carrier, 1987; Colchester, 1994) to introduce new ways of resolving the risks that common property institutions are often designed to address. Mobility over space and through time is a compelling factor to address production fluctuations, but markets and exchange compete with them by encouraging individuals to specialize in different kinds of economic activities. By specializing in different occupations and exchanging their surplus output, individual producers can alleviate the need for migration (with or without means of production) and storage. Markets also contribute to alternative arenas for the provision of credit (Anderson et al., 2003) and generation of prestige in ways that can undermine the importance of other local institutions (Tietenberg, 2002; Yandle and Dewees, 2003). Furthermore, upland economic activity, such as intensification and conversion of forested lands for agriculture may impact the ecology of the watershed adversely reducing water flows into it or polluting the water entering it. In turn, this impacts the amenity value of the downstream wetlands, thereby reducing the direct and indirect economic and ecological benefits generated. Significant trade-offs between different kinds and levels of economic activity may arise, a number of which work through the impact they have on ecological structure.

Research on local CPR problems has demonstrated that under these circumstances, solutions worked out by those individuals directly affected prove to be more successful and enduring than resource regimes imposed by the central political authorities (Ostrom, 1990; Bromley et al., 1992; Keohane and Ostrom, 1995). National governments make environment and natural resource policy choices based on the notion of intrinsic and moral value, and rights to society. The rights and values, however, have to be identified, quantified through a systematic and fair manner, and expressed in a single unit of measure that is legally recognized. Under this decision criterion, policy decision makers place less importance on ecosystem services (Goulder and Kennedy, 1997). On the other hand, strict utilitarianism approach demands that a decision has to be based solely on economic efficiency, that is, maximization of the net benefits to society (Goulder and Kennedy, 1997). This decision rule is implemented through the use of benefit-cost analysis (BCA). As such, economic valuation plays a central role in the application of BCA, since BCA requires an estimate of the benefits and costs of each alternative using a common method (i.e. economic valuation) and metric (dollars) so that comparisons among policy choices can be made. Decision makers, and conservationists, however, need to recognize that achieving a particular conservation objective or goal such as conservation of watersheds for water provision comes at a cost, since the resources that must be devoted to this preservation are not currently available for use in providing other goods and services. Therefore, a typical economic valuation process asks whether the benefits of that decision are worth the cost borne by individuals or society that has something to lose if the policy decision is passed. The challenge with this econometric tool is that all costs and benefits must be captured or at least considered. In the interest of environmental justice and equity, those who stand to benefit should be willing and able to compensate the losers. The challenge then is how to determine a fair and reasonable compensation value, and whether society is willing to pay to redistribute the benefits.

The other approaches such as household production function (HFP) involve modeling consumer behavior based on the assumption of a substitutional or complementary relationship between an ecosystem service and one or more marketed commodities. These include time allocation models for collecting water or firewood, averting behavior models, hedonic property value approach, and travel cost method. The main advantage of these methods is that they help to determine household resource allocation and consumption decisions by examining the interactions between purchases of marketed goods and the availability of nonmarket environmental services, which are combined through a set of technical relationships to produce a utility-yielding final good or service. On the other hand, BCA assumes that all environmental goods and services are marketable and can be determined using individual indifference curves, which is never the case because some of the services have no market mechanism in place. Therefore, the use of personal interviews to elicit the benefits and costs associated with each good or service is the only logical way to estimate the monetary value of those environmental values.

In this study, detailed economic analysis of the socioeconomic data collected in 2006, and descriptive statistics for the socioeconomic data collected in 2008, 2009, and 2012 are presented here. These socioeconomic data combined with the environmental variables specifically slope, elevation, precipitation and temperature were used to evaluate the contribution of common pool resources to rural livelihoods, and assess the relationship between the extraction patterns under different climatic conditions and recommend action-oriented policy interventions.

6.2.4 Empirical model

This paper sought to examine the linkages in the context of the Albertine Rift watershed (drainage basins) in Africa by setting up an integrated dynamic ecological-economic model. It derives sensitivity of water storage to the ecological health indices in a series of simulated scenarios with respect to the current and future pressures on the watersheds. In the process, a more complete characterization of factors impacting on the sustainable watershed management is arrived at and its linkages with both the ecological and economic driver explored. The forces impacting on the watersheds can be viewed in the internal commons framework, where continuous feedback between socioeconomic drivers, environmental pressures created, changes in the state of the ecosystem and consequential impact on economic activities (land use options) envisioned to describe the economy-ecology interactions (Wade 1988; Ostrom 1990; Baland and Platteau 1996). At times, the impacts may induce responses which feedback into the socioeconomic drivers and result in the initiation of a new set of interactions.

The application of Total economic value (TEV) framework (Bishop et al. 1987) would have been very useful in revealing the holistic behavior of resource users, however, the complications associated with establishing the non-market values in developing countries, no effort was made to use contingent valuation methods to elicit these values. TEV is based on the As such, mainly direct use values such as raw materials and physical products which are used are consumed or sold were recorded and their economic value computed based on local market prices.

6.3.5 Data

There is considerable evidence to suggest that socioeconomic heterogeneity has significant implications for resource use as well as management of CPRs (Adhikari et al., 2004; Agrawal and Gupta, 2005; Mamo et al., 2007; Mishra, 2007). For example, studies have shown that CPR extraction is influenced by household socioeconomic attributes such as education, gender, household size, income, ethnicity, distance to the resource, productive assets (e.g. size of landholdings, livestock, harvesting tools), and markets (prices, distance to markets) (Adhikari et al., 2004; Mitra and Mishra, 2010). Other factors that influence CPR extraction include physical conditions (e.g. temperature, precipitation), and resource governance type (i.e. government, communal).

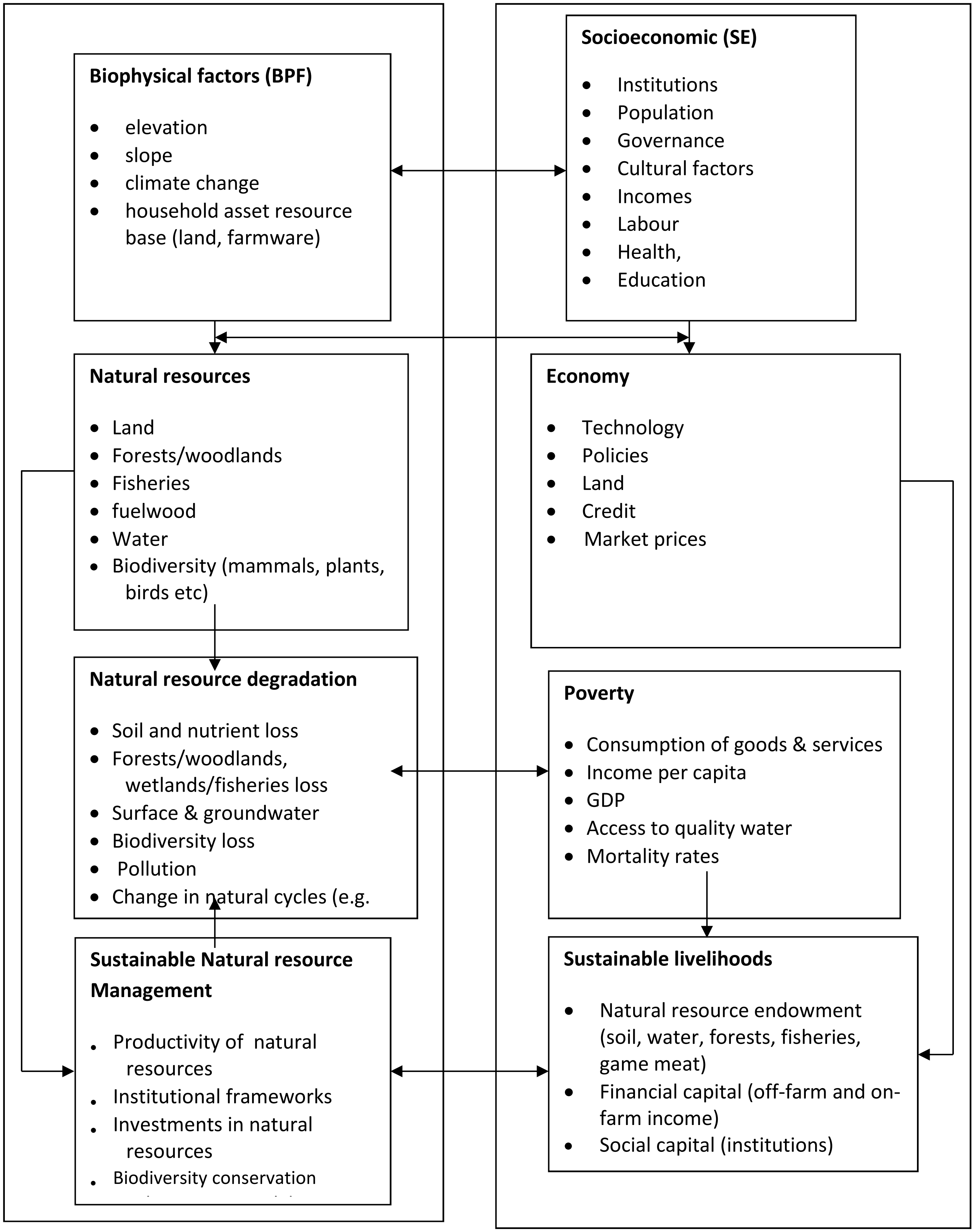


Figure 6.2 Conceptual framework of a bio-economic model for the Albertine Rift region

6.3.5.1 Socioeconomic data

The principal tools that were used to collect the socioeconomic data include 1) Well being or wealth ranking, where wealth was defined by the local community members and common attributes (e.g. size of land owned, number of livestock, type and number of housing structures, household assets) were identified and used in the household ranking exercise; 2) Household questionnaire interviews; 3) Rapid Social Impact Assessment (RSIA); 4) Focus group discussions; 5) Secondary data and key informant interviews were conducted during data collection. Purposive sampling was used from identified strata covering a range of different socio-agroecological contexts and proximity to the protected area representing varied ecosystems, that is, a radius of 0-3 km was considered near, medium (3.1-5 km) and far (>5 km) (Figure 6.3), income and well-being factors, agro ecological conditions, and other livelihood support systems such as livestock production, monoculture agriculture, agroforestry, fishing and mining.

Household considered for interviews were randomly selected in a hierarchical manner based on the political administrative (Bush, Nampindo, and Aguti 2004; Bush and Mwesigwa, 2008). In Uganda, a district in which a protected area or ecosystem is located formed the first selection level, followed by the subcounty, a parish and lastly the local council (LC), which is the smallest political boundary. In Democratic Republic of Congo, determination of where to select the households for interview commenced with a province, followed by a territory and lastly a commune (DR Congo). Typically, the smallest political administrative unit is constituted by averagely 100 households and a membership of seven people per household. After selection of the village, households were then listed and categorized by key village informants and opinion leaders according to the wealth ranking (i.e. poor, middle, rich) and assigned numbers to ensure anonymity. The numbers were placed in a transparent container and reshuffled thoroughly and

30 community members (male and female) were asked to pick one piece of paper and the corresponding household was selected for interview. The sampling of households from wealth categories was done proportionately to ensure representation of the wealth categories in the community. In Uganda, a total of 690 households covering 36 villages selected from 18 parishes and six districts in Uganda were interviewed representing over 4500 households where 480 households live adjacent to forests/woodlands, wetlands (150), and fisheries (60). Another 60 households were selected and interviewed from Democratic Republic of Congo, making an overall total of 750 households.

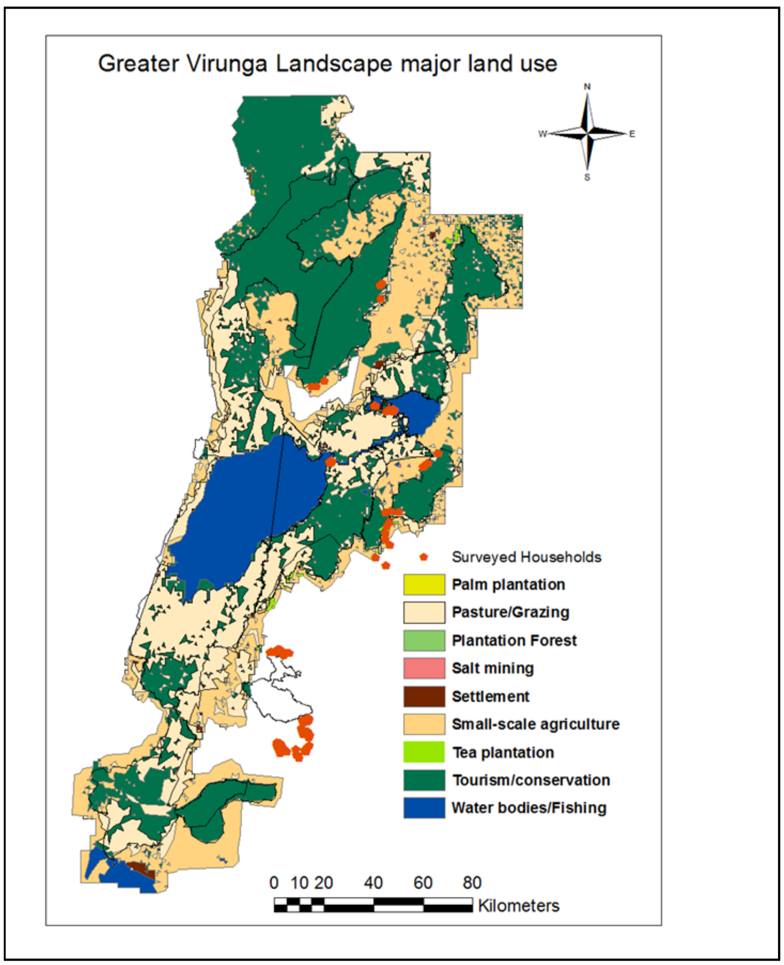


Figure 6.3 Map showing the locations of households selected for interview

Generally, the datasets include information on household characteristics (i.e. household size, age, gender, education level), household assets disaggregated by in-house material possessions, livestock holdings, land resources, including woodlots/natural forests, and agricultural crops therein. In addition, information on household use of the forest, benefits derived, and a period of intensive resource use, information on food scarcity, household expenses, and most important medicinal plants harvested to treat illnesses is documented. On top of documenting water sources and their location, trends in water quality was also recorded. All benefits derived by households from the forest and consumed at home or sold, and the product prices offered at farm gate or local market were also recorded. The incomes and costs associated with each agricultural crop and product, including livestock and forest/wetland products were recorded. Other sources of income earned by the household members were also documented. For some parishes critically affected by problem animals, problem animal damage to crops and livestock predation occurrence and frequency was recorded.

In order to answer this objective, used a combination of tools such as focus group discussion; conduct a meta-analysis of the household socioeconomic survey data which was collected in 2006, 2008, 2009 and 2012 to understand resource users' consumption behaviors. The selection of households was conducted through a multistage sampling design process, where districts in which common pool resources are located, constituted the first stage of the selection procedure, then subcounty followed by a parish. At least two villages adjacent to CPRs were selected randomly from three parishes for household survey. In the selected villages, households were randomly picked following the village household list disaggregated by wealth levels as defined at a local scale.

6.3.5.2 Data Analysis

Data analyses involved plotting of scatter plots and pearson correlation was conducted in both R software and Stata ver 11 software (Stata Corporation, College Station, Texas: http://www.stata.com/) before the critical variables were selected for input in the logistical model. Kruskal Wallis test was performed to test the statistical significance of the relationships between selected variables in the extraction model. A logistic model of the form represented by the equation below was used to assess the effect of various explanatory variables on the extraction of common pool resources. The explanatory variables considered for input in the regression model were household socioeconomic characteristics, asset base, environmental factors such as temperature, precipitation, temperature, elevation, slope, latitude and longitude, and physical attributes such as distance from the protected area, and markets

Logit model is as follows:

$$log\left[\frac{\pi(x)}{1-\pi(x)}\right] = logit(\pi) = \alpha + \beta_1 x_1 + \beta_2 x_2 + \cdots + \beta_k x_k$$

6.4 Results

This is study was not necessarily designed to undertake socioeconomic analysis, however, data mining was done to understand the local factors, particularly at the household level that influence land use change with a major emphasis on forest and wetland conversion to agriculture and tree plantations. Majority households (91% N= 690) are involved in agriculture as subsistence mixed crop-livestock farmers and on a small scale commercial farming. Of the total number of households surveyed (N= 690), 45.3% are involved in livestock keeping. Over 50 crops are grown in this region, however, results of the comparison between what is produced and domestically consumed vs. what is sold by the household is compiled for only 37 crops

(Figure 6. 3). With the exception of commercial crops (e.g. cotton, coffee, tea, tobacco) most of the crops produced are consumed at home and only the surplus is sold. Horticulture products dominate the list of crops that are 100 percent consumed at home. Similarly, most livestock products produced are consumed at home (Figure 6.4), where manure and blood are domestically consumed. It is important to note that the two major categories of African cattle reared in East Africa are the humpless (*Bos taurus*) and humped (*Bos indicus*) types (Rege 1999; Rege and Tawah 1999). The humpless category is subdivided further into longhorn (B. taurus longifrons) and shorthorn (B. taurus brachyceros), while the humped category is subdivided into zebu proper and zebu crossbred-types (Rege 1999; Rege and Tawah 1999). In this region, the humpless longhorned cattle are the most common.

The main livestock products produced include milk, meat, eggs, manure, and blood (Figure 6.5). Interestingly, live animals are predominately kept as stock and rarely sold for cash. It is also reasonable to suggest that livestock keeping is highly valued as a source of protein in form of milk and blood for the rural households because it dominates the household diet. In addition, livestock provide draft power, manure and urine used as fertiliser. In addition, owning ruminants encourages households to plant browse trees, shrubs, leguminous forages, which help to control soil erosion, promote water conservation and increase soil fertility (Rege and Gibson 2003). Livestock keeping is a form of insurance against shocks and financial stress, only sold when there is urgent need for cash. Besides, it could also be argued that livestock is kept as a cultural practice to derive prestigious values. Many of the cattle keeping families have strong cultural attachments to the livestock, including social status, use in marriage ceremonies and other cultural festivities. All these benefits aside, it must be acknowledged that where population pressure and poverty coincide, such as in pastoral areas, poor management of livestock has resulted into land degradation, which is a recipe for cutting down forests known to be fertile and conducive for farming. It is estimated that 70% of the world's rural poor depend on livestock as a component of their livelihood and it has been suggested that focusing on improving the sustainable livelihoods of these people can do more to reduce poverty than increasing productivity in intensive industrial systems (Rege and Gibson 2003).

In terms of provisioning ecosystem services, the rural poor communities depend on forest and wetland resources for their livelihood. Majority (85% N=690) of the households surveyed in GVL harvested wild foods, fibers, wood products, medicinal plants, water, bushmeat and construction materials from forests/woodlands and wetlands mainly for domestic consumption (Figure 6.3). Ecosystem products such as water, medicinal plants, game meat, and rattan (used in making craft products) are harvested and entirely consumed at home. Importantly, both rural and urban populations (98% N =690) depend on fuelwood energy (i.e. firewood and charcoal) for cooking and heating. We also noted that even the livestock producing households, majority (70% N=416) graze their livestock inside protected areas, that is, forest reserves, national parks and wildlife reserves, and wetlands. In terms of household income streams, 40.2% of the household income is derived from agriculture contributing a total of US$186,646.05 with an average household income of US$ 270.50 (Table 6.1). The second most important income stream is trade or businesses contributing nearly 25% followed by income from remittances/gifts and casual employment on-farm and off-farm (18.0%). Other sources of income include on-farm forestry contributing slightly less than one percent to the household income, fisheries (6.2%) for the fishing households, and property income from renting out land, farm equipment (e.g. tractors, draft power) and buildings. At the household level, the average income from fisheries (US$477.62) is greater than the rest of the other sources of income. This could be attributed to the fact that the costs associated with fishing are relatively low, particularly the lack of a direct economic rent paid to the government to access fishing grounds. Crop farming, livestock production, and businesses/trade involve a lot of transaction costs and a long marketing chain that reduces the profit margins for farms.

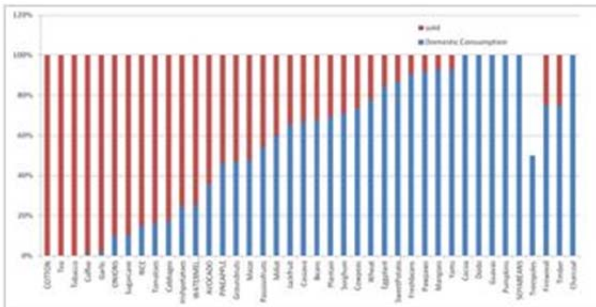
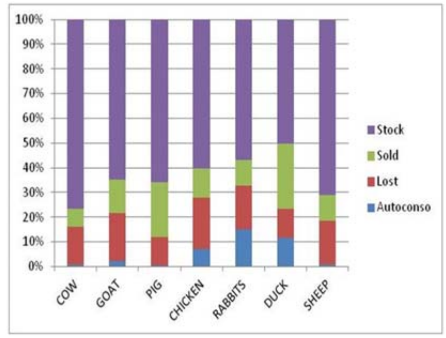
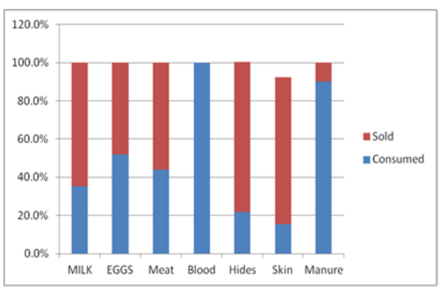



Figure 6.4 Proportion in percentage of consumption and sale of crops grown by households (left) and proportion in percentage of livestock by stock, consumption and sale (right)



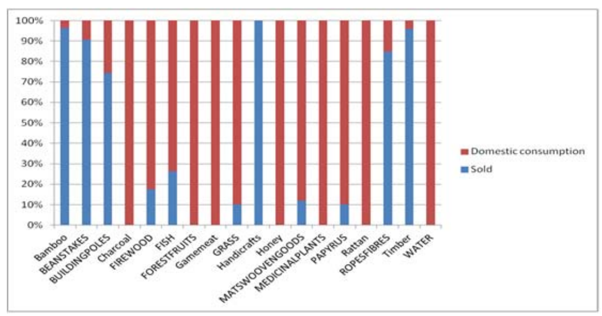


Figure 6.5 Livestock products showing the proportion of consumption and sold Ecosystem provisioning products broken down by the proportion of consumption and sold

Income from forest was expected to be equally high since no rent is paid to harvest natural products mainly from protected areas. The presence of tight enforcement of rules and regulations, types and quantities of products that one can harvest from protected areas is reduced. For example, in some National Parks (e.g. Queen Elizabeth National Park), harvesting of natural products is prohibited and most harvesting is done illegally.

Table 6.1 Income streams for rural communities

Income stream	N	Amount in US$	Average	S.E	Percent
Crop farming	629	186,646.05	270.50	45.08	40.2
Livestock	416	22,878.85	33.16	14.83	4.9
Forestry on-farm	43	3,557.42	5.16	6.04	0.8
Natural Forests/woodlands/wetlands (PA)	323	15,972.91	23.15	6.27	3.4
Business (agricultural trade + merchandize)	187	113,697.50	164.78	39.15	24.5
Other sources (remittances, gifts, casual labor)	318	83,437.80	120.92	33.42	18.0
Fisheries	60	28,657.14	477.62	59.43	6.2
Property income	52	234.62	3.45	11.35	0.1
Total Income		463,728.40	672.07		
Total HH Interviewed	690				
Total population from surveyed HH	4340				

At the ecosystem level, the net adjusted income from forests/woodlands is higher (US$80,470.8) than incomes from fisheries and wetlands (Table 6.2). This is attributed to the large number of households that live adjacent to forested parks compared to those close to wetlands and lakes. The perimeter of protected areas is bigger than for other ecosystem systems resulting in high household access to resources. It is important to note that mean annual adjusted income from fisheries is higher (US$280.26) than the income from forests/woods and wetlands. The incomes were adjusted for economies of scale using equivalent adjustment scales proposed by (Deaton and Paxson 1998) and the Organization of Economic Cooperation for Development (OECD 2013). The factors commonly taken into account to assign these values are the size of the household and the age of its members (whether they are adults or children) (Deaton and Paxson 1998).

Table 6.2 Incomes by adjacency to ecosystem type

Ecosystem	N	Net Annual Income Adjusted (US$)	Min	Max	Mean Annual Adjusted HH Income (US$)
Fisheries	60	16,815.8	60.60	1,236.6	280.26
Forests/Woodlands	480	80,470.8	62.8	1600.0	167.65
Wetlands	150	24,232.0	42.2	1672.5	161.55

6.5 Conclusions

Ecosystem-based approaches can contribute to climate change mitigation and adaptation and to sustainable development more broadly. Spatial planning for ecosystem services at international, national and local levels will be an important component of ecosystem-based approaches. However, because not all elements of biodiversity are critical for ecosystem services, it is important to also target critical areas for purposes of protecting biodiversity. The concluding message from the assessment on people's dependence on common pool resources is that total protectionism makes rural communities more vulnerable, and socially and politically weak. The idea of shifting the responsibilities to local communities and indigenous people is a good idea, but the mechanism for collaboration needs to be strengthened in terms of power showering and offer reasonable benefits to the communities.

Protected area frontline communities incur a lot of costs in foregone consumption of common pool resources in order to promote conservation. It very logical that those who stand to benefit from the protection of ecosystem services undertake to invest in social service provision to improve the lives of the vulnerable people through investment in their education, health and creation of enterprises that are not nature-based. Exciting partnerships and collaborations among different institutions need to be pursued and new networks created in order to talk the major threats to conserving the Congo basin.

Any effort geared toward improving governance and reduction of corruption among government civil servants and politicians a like, will help to save nearly five percent of the funds swindled and use it to finance conservation and support rural communities.

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CHAPTER 7

CONCLUSION AND RECOMMENDATIONS

Conclusions and recommendations of this study are presented here according to the chapters. Under chapter two the objectives were to develop species distribution models for the endangered and threatened species in the Congo basin, and identify areas with high biodiversity concentrations. The most important achievement out of this study is that endangered and threatened species in the Congo basin are not uniformly distributed and that their ecological needs needs vary from species to species interacting with environmental, biophysical and ecological factors. In representing the distribution of nearly 45 species, variables such as topographic factors land cover in particular forests and precipitation were universal to nearly 90% of the species modeled. Areas in the basin considered marginal at present, seem to be the next best available suitable habitat of quite a large number of species. Unfortunately, most of those areas are equally habited by humans and continue to be the next available land for agricultural expansion. Under chapter three, the objectives were to identify ad delineate priority areas for water resources management. The study showed that the central region of the Congo basin is experiencing more runoff and less soil erosion than other parts of the basin. This partly region is benefiting from the natural presence of the main Congo tributary which hinders people from conducting activities that would otherwise facilitate runoff and soil erosion. At the sub basin level, some of the subwatersheds that are dominated by human settlements and cultivation, runoff and soil loss are relatively high compared to other places. The main conclusions are that runoff in the Congo basin is quite varied and not uniform as it was hypothesized. At the sub basin level, subwatersheds in mid central part of the basin are experiencing a lot of runoff and this is not necessary due to human disturbance given the high forest cover, rather it due to the presence of a high water table and saturated soils implying that no more infiltration can take place. These subwatersheds also have high streamorder to the size of 7-9. It is possible that after a storm event, free flowing water is drained into the river channels. The region may become water stressed if part of the catchment area continues to undergo drastic degradation. The other part of the conclusion, which can be inferred from these results, is that soil loss is occurring at a very high level compared to other parts of Africa such as the Lake Victoria basin where Lufafa et al (2003) reported very high soil loss. Lastly, it is reasonable to say that this study has made some substantial contribution to the understanding of the hydrological process in the Congo River Basin as well as generating useful information for the management of water resources in the region. The lack of empirically collected information on the streamflow, surface runoff, and soil erosion studies still continue to be a challenge for scientist that attempt to understand the hydrology of the Congo basin (Tshimanga and Hughes 2013). To circumvent this problem, this study relied on expert opinion that provided useful comments on the runoff and soil risk map. The next step is to disserminate these findings among the relevant institutions to get them to appreciate the challenges and begin to take positive steps toward developing strategic action plans for the management of Congo basin.

Chapter four which focused exclusively on understanding population dynamics of the savanna elephant generated from interesting results, from which good conclusions can be drawn. Elephants in Africa are disappearing at an alarming rate mainly due to poaching, habitat degradation and loss, and retaliation killings due to human-wildlife conflicts. The situation is expected to become worse with the advent of climate change impacts resulting in high occurrence of prolonged droughts in both arid and semiarid regions. Understanding how animal populations will respond to such dramatic events is very crucial to the design and implementation of mitigation strategies, and development of adaptive capacity of wildlife managers to adequately respond to these challenges. An assement was done to understand how GVL age specific elephant populations are likely to respond to habitat change, war, poaching, water resources and climate change. It is reasonable to conclude that climate change, wars, poaching, and habitat degradation, if they continue unbated, GVL elephant population is likely to decline tremendously and the population structure is skewed to the left where more young and sub adult elephants dominate the population. The seniors are disappearing at a very high rate mainly due to poaching. An improvement the suitable habitat area by 50% would result in an upward surge of elephants in all age classes attaining the climax in the year 1970s and the later dropping in the mid 1990, only to start rising again steadily.

In terms of addressing the challenge of poaching, the focus should be on the strengthening law enforcement, monitoring and research, and working towards creating alternative income sources for the frontline communities. The aim of incorporating the income enhancement program is clearly for rewarding the effort of those communities that ensure compliance with the law and report those who violate the laws, including poaching. It should be designed in such away that communities make an obvious link between conservation benefits associated with the compensation scheme. The compensation scheme could made part of the revenue sharing money given to protected area adjacent communities. To achieve this, a minimum increase by five percent in funding towards those three critical interventions is need. This could also be accompanied by strengthening the protected area institutions and human-wildlife conflict management association already on the ground through training, empowerment and provision of basic tools for reporting of illegal activities in the area. Furthermore, regional governments together with conservation organizations and donor community must be willing to invest in poverty reduction programs, scale up the effort to reduce the international demand for ivory through CITES, treat ivory business as a national and global security threat, and invest in human population reduction strategies such as family planning, and education. Elephants in GVL are a transboundary resource which requires transboundary management approach, cooperation between conservation agencies, and development of effective partnerships with all relevant stakeholders such as local and regional governments, wildlife protected area authorities of DRC, Rwanda, and Uganda, law enforcement agencies such as the military, police, judiciary, customs and border control authorities is very highly recommended. Collaboration is fundamental to effective conservation (Wondolleck & Yaffee 2000; Plumptre et al. 2009) and regional governments must be willing to develop institutional frameworks to allow it to happen.

Chapter five addressed itself to mapping ecossystem services and made an attempt to develop an ecosystem services map for the region based on carbon sequestration, watershed values, and biodiversity conservation. Ecosystem services are high in the central part of the basin with relatively intact forests, including areas under protection. From this study, it is very clear that effort should be placed on areas already protected as they house high biodiversity values. Generally, water quality and quantity is compatible with carbon sequestration, but the latter reduces biodiversity values, particularly in agricultural and savanna dominated areas. Biodiversity values are relatively high inside protected areas and relatively remote and undisturbed areas. It is important to note that biodiversity values are derived from an ecosystem's suitability to support a diversity of species with respect to their ecological and environmental requirements. The entire study, particularly the chapter on elephant population dynamics attempted to account for declines in biodiversity due to legal or illegal harvesting, an aspect that is very important to consider. In the analysis, only the removal of a suitable habitat due to land conversion, fires, flooding among others stressors were considered very important that did not constitute part of the main study. As such, it is not surprising to note that protected areas, which are expected to suffer less or gradual conversion and experience relatively less wildlife harvesting, showed high biodiversity values. In particular, maximizing rates of carbon sequestration could lead to potentially perverse conservation outcomes when silvicultural interventions that are good for timber and carbon are detrimental to biodiversity.

Ecosystem-based approaches can contribute to climate change mitigation and adaptation and to sustainable development more broadly. Spatial planning for ecosystem services at international, national and local levels will be an important component of ecosystem-based approaches. However, because not all elements of biodiversity are critical for ecosystem services, it is important to also target critical areas for purposes of protecting biodiversity. The concluding message from the assessment on people's dependence on common pool resources is that total protectionism will not work and the idea of shifting the responsibilities to local communities and indigeneous peoples is a good idea, but one which needs to be backed up they action-based research to prove beyind doubt that the commubities don't become even more deprived of their resources. Devolution of power and authority should be absolute and allowed to be exercised through legal instruments such as bye-laws, memorandum of understanding or statues. Protected area frontline connmunities incur a lot of costs in foregone consumption of common pool resources in order to promote conservation. It very logical that those who stand to benefit from the protection of ecosystem services undertake to invest in social service provision to improve the lifes of the vlnerable people through investment in their education, health and creation of enterprises that are not nature-based. Exciting partnerships and collaborations among different institutions need to be pursued and new networks created in order to talk the major threats to conserving the Congo basin. Any effort geared toward improving governmance and reduction of corruption among government civil servants and politicians a like, will help to save nearly five percent of the funds swindled and use it to finance conservation and support rural communities.

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