Understanding the ecological impacts of water-abstraction and changes in surface water availability in the Kilimanjaro landscape, Tanzania.

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Contents

List Figures

List of Tables

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Abstract

For decades, surface water has been extracted for human use in the wildlife-rich Kilimanjaro landscape, northern Tanzania that includes the Kilimanjaro and Arusha National Parks. This study evaluates the natural and human-induced changes in the availability and quality of surface water, and the ecological impact on the surrounding vegetation and large herbivorous mammals. Excessive and unsustainable water abstraction takes place within and outside the parks, as between 70% and 90% of the available water is removed within 20 km of the park boundaries. Annual rainfall did not decrease in recent decades, suggesting that the water shortage is not currently exacerbated by climate change. Abstraction resulted in reduced water quality due to increased evaporation and reduced dilution, including of the organic and inorganic material emanating from crop irrigation. Salinity, fluoride and nitrate, increased downstream in the Ngarenanyuki River and water holes in the dry season to concentrations that may be harmful to wildlife. Levels of iron and aluminium were above acceptable limits for wildlife use in the downstream reaches of the Simba River and, again, in some water holes. Because of upstream water abstraction in the National Parks by the local communities, the large herbivores concentrated around the remaining surface water sources in the lowland semi-arid areas. Animal abundance increased towards the water sources, including those with high salinity and mineral content, suggesting that water availability overrides water quality during periods of water scarcity. Plains zebra and wildebeest, which are among the water-dependent species, were more associated with the available surface water sources than browsers such as giraffe and impala. An increase in riparian wetland vegetation was observed in Arusha National Park, and this was likely due to an increase in surface water following increased rainfall. In contrast, the downstream semi-arid lowland areas showed a substantial loss in riparian and adjacent floodplain vegetation due to excessive upstream water abstraction and associated increased siltation. Vegetation cover loss was caused by overgrazing and trampling by wild animals and livestock seeking drinking water from the scarce water resources. Therefore, an increase in rainfall leads to an increase in water in the upstream and hence increased riparian vegetation. However, excessive water abstraction leads to water shortage in the downstream areas, increased mineralisation, decreased riparian vegetation and increased number of mammals. This thesis has demonstrated that the existing water abstraction in

the National Parks and the upland villages around the parks is unsustainable, leading to a developing water crisis that is adversely affecting the ecology of the Kilimanjaro landscape. A number of solutions are proposed to improve water resource management and to mitigate the ecological impacts of water abstraction.

Declaration

that no portion of the work referred to in the thesis has been submitted in support of an application for another degree or qualification of this or any other university or other institute of learning.

Copyright Statement

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Chapter 1: General introduction

This chapter commences with a review of the literature relating to the study topic followed by aims and objectives, a description of the study area and the structure of the thesis.

1.1 Literature review

Surface freshwater is vital for the survival and health of biotic communities and entire ecosystems, supporting both aquatic and terrestrial ecosystems, which in turn provide a wealth of ecosystem goods and services for humans. Thus, globally surface water plays a pivotal role for human health and socio-economic development (Duda and El-Ashry, 2000; UN-WWAP, 2012, 2015). Freshwater is defined as the liquid component of the hydrosphere containing less than 1000 ppm of dissolved salts, and exists as surface water, ground water and water vapour (Groundwater Foundation, 2018; Postel, 2000; USGS, 2020). However, in another definition, freshwater has total dissolved solids (TDS) less than 3,000 mg/L (EPA-SA, 2015). Surface freshwater exists in lakes, ponds and reservoirs (standing or lentic water), or rivers and streams (lotic or running water) (Jury and Vaux, 2007). While it is such an important resource, harbouring nearly 10 % of the world's animal species and sustaining the majority of terrestrial species, available surface freshwater mainly in lakes, reservoirs, rivers and streams form only about 0.01% of the of the total water on earth as most is either saline or locked up in polar ice, vapour and glaciers. Surface freshwater covers only 0.8 % of the earth's surface, compared to the more than 70% covered by seas and oceans (Balian et al., 2008; Dudgeon et al., 2006; Jury & Vaux, 2007). Global annual surface freshwater that is readily available for human consumption is estimated at around 12,500 km^3/year (Falkenmark & Rockstrom, 2014; Postel et al., 1996), compared to a total of 1,386 million km^3 in the earth's hydrosphere (Du Plessis, 2017).

Surface freshwater availability is naturally affected by rainfall, evaporation and geology (Gaylard et al., 2003; Conway et al., 2009). However, such water availability is also subjected to human impacts particularly excessive use, pollution, and human-induced climate change.

Surface freshwater is rapidly declining due to unsustainable use as the increasing human population continues to over-abstract and pollute freshwater to meet socio-economic demands while often disregarding the environmental impact (Grafton et al., 2013; McClain, 2013). For example, more than 50% of the world large river systems have been seriously affected by dams, resulting in potentially damaging decreases in downstream discharge (Nilsson et al., 2005). Humanity is already consuming more than 50% of all spatially and temporally accessible runoff freshwater from rivers, lakes, streams, and shallow aquifers (Hinrichsen, 2003; Postel et al.,1996). Irrigation farming is the largest consumer of extracted freshwater, accounting for more than 70% of all water consumed by human activities worldwide (Hinrichsen, 2003; Jury and Vaux, 2007; UNESCO-WWAP, 2012). At least 20% of the global population live in places with water scarcity (defined by Jury and Vaux (2007) as per capita water resource availability (PWR) which is higher than 500 m^3/year but less than 1000 m^3/year), and where water extraction to meet socio-economic needs exceeds 75% of available river flow. This scarcity is expected to grow as demand approaches or exceeds the available supply (Gleick, 2014). According to a report by the UN-WWAP (2012), global food and energy demand will increase by almost 70% and 60% respectively by 2050, further increasing the demand for freshwater.

The growing human pressure on surface freshwater resources also is manifested through pollution (Dudgeon et al., 2006; Richter et al., 2015). Pollution is a threat to surface freshwater security to an extent that it is limiting food production and damaging ecosystem function and human health. Water pollution is ranked as the number one cause of deaths world-wide (Jury and Vaux, 2007).

Climate change impact on water is predicted to vary spatially and temporally. In some regions it may reduce surface water availability and predictability in the twenty-first century, with some regions being particularly vulnerable. For instance, for different periods of time in this century, it is projected that Colorado River (USA) discharge will decline by between 4 and 18%, the Yellow River (China) by between 9 and 29%, and the Murray–Darling River (Australia) by almost 70% due to climate change impacts (Grafton et al., 2013). While climate change and non-climatic human induced impacts are likely to synergistically affect surface freshwater availability, non-climatic human-induced impacts represent the greater threat in many places (Vörösmarty et al., 2007; Grafton et al., 2013).

Change in surface water often results into change in the associated biodiversity. The world has witnessed over the last century an unprecedented degradation of biodiversity and ecosystems, and the rate of damage is accelerating. For example, almost half of the world's wetlands were destroyed through human activities in the last 50 years (Hinrichsen, 2003; Jury and Vaux, 2007). Loss of freshwater biodiversity is largely due to over-abstraction of water, habitat destruction and water pollution (WWF, 2016). Since 1970s there has been high rate of species extinctions, especially freshwater species, largely attributed to human impacts (Hinrichsen, 2003; WWF, 2016). Flow reduction and regulation by water abstraction do not merely affect surface water but also associated ecological systems, especially wetlands and wildlife protected areas, which provide critical habitats to a variety of wildlife species. Impacts range from a reduction in the inundated area, blockage of wildlife migration routes, loss of wildlife habitats, reduction or change in species abundance, richness and distribution, and encroachment by invasive exotic species. Damming and weirs often cut off river flow and act as barrier to transport of particulates, nutrients, and inhibit species movement, and therefore adversely affecting hydrological and ecological processes (Kingsford, 2000).

Due to natural low and episodic high flows, plus stream/river diversions, damming, reservoirs, and land use changes, the surface freshwater resources of sub-Saharan Africa are among the most affected in the world (UNEP, 2006; Jury and Vaux, 2007). In addition, widespread poverty, high population growth, poor planning and rapid urbanisation, plus dependence on rain-fed agriculture (which accounts for 95% of farmland) place sub-Saharan Africa as the most vulnerable region in terms of surface freshwater supply (UNEP, 2006, 2010; Wani et al., 2009). Irrigation farming which in most cases is unsustainably practiced, accounts for the largest share of human water use, where in some cases of the least developed countries including those of sub-Saharan Africa, this proportion reaches almost 90% (Hinrichsen, 2003; Jury and Vaux, 2007; UNESCO-WWAP, 2012). Under these challenging conditions, and in the era of global warming which increases the likelihood of drought, sub-Saharan Africa remains highly vulnerable in terms of sustainability of surface water availability (UNEP, 2006, 2010; Wani et al., 2009).

Surface freshwater availability in sub-Saharan Africa is somewhat unpredictable and unreliable due to highly variable precipitation and other climatic conditions. Moreover, the distribution of freshwater in this part of Africa is highly uneven (UNEP, 2010). This leads to water shortages and hence water management challenges (Conway et al., 2009; Taylor et al., 2009; UNEP, 2010). Climate change is likely to disproportionally affect sub-Saharan Africa in the twenty first century as the region will experience a large rise in temperature of almost 1.5 times the global average, however projections show mixed pattern of rainfall changes (Christensen et al., 2007), with some places expected to receive more rain e.g. East Africa, while others receiving less e.g. Southern Africa, and no clear trend in annual rainfall for the Sahel and West African regions (Christensen et al., 2007; Giannini et al., 2008). However, there is a significant degree of uncertainty in the climate variable projections and associated impacts because climate models still fail to have a robust agreement with current observations and because further uncertainties arise when the climate projections at a global level are downscaled to a regional level. For instance, the annual rainfall and the inter-annual rainfall variability at Narok in southwest Kenya has not increased, contrary to climate projections (Bartzke et al., 2018).

Surface freshwater pollution, especially through direct solid waste disposal into streams, rivers and lakes, is prevalent in sub-Saharan Africa (UNEP, 2006). Sub-Saharan Africa has more than 40 % (over 300 million people) of people world-wide that do not have access to uncontaminated (especially from faecal matter) drinking water sources (UNDESA, 2014). Eutrophication and production of toxins are common in surface waters and have led to an increase in invasive exotic weeds, excessive growth of indigenous macrophytes, blooms of cyanobacteria and hence threats to human health in many surface freshwaters in the region (UNEP, 2006; van Ginkel, 2011). Damming of rivers also results in water pollution through changing the temperature and chemical composition of impounded water as well as causing less dilution of effluent in the downstream areas as is the case for the Hadejia-Nguru wetlands in Nigeria, part of which is protected as the Adiani Forest Reserve, Baturiya Game Reserve, and Chad Basin National Park (Lemly et al., 2000; Ringim et al., 2017).

Reduced flow often leads to reduction in water quality and the inundated area, in turn this leads to reduced wetland flora and fauna species diversity and abundance in several regions of sub-Saharan Africa (Lemly et al., 2000; Zwarts et al., 2005). This often leads to the degradation and/or loss of biodiversity, and the emergence of invasive exotic species both in aquatic and terrestrial ecosystems (Drijver and Marchand, 1985; Scholte, 2005). Natural river flow variations and connection between upper and lower catchments are essential in maintaining species migration, abundance, and distribution, diverse habitat, and transportation of sediments and nutrients downstream to other areas such as wetlands and delta which harbour a great number species (Duvail and Hamerlynck, 2003; Lehner et al., 2011; Stommel, 2016; WWF, 2016). For example, the establishment of more than 20 dams and associated numerous irrigation schemes upstream regions of the Hadejia River basin in Nigeria resulted into reduced wet season inundation, riparian vegetation and wildlife habitats, especially for migratory water birds (Lemly et al., 2000; Ringim et al., 2017).

There are several examples of detrimental ecological impacts from unsustainable abstraction and damming of surface freshwater throughout sub-Saharan Africa (Drijver and Marchand, 1985; Scholte, 2005). The construction of the Kariba dam on the Zambezi River resulted in an ecological disaster that caused many wild animals to drown in the reservoir, necessitating a rescue operation (Operation Noah) where more than 5000 animals including black rhinoceros, bushbuck, baboons, monkeys, genet, were saved (WCD, 2000). The impoundment resulted in an invasion of an aquatic floating weed, *Salvia auriculata* (Scudder, 2005). The extent of the impact on biodiversity is however not clearly known as there was no adequate benchmark ecological information prior to the construction of the dam. However, the lake did favour some wildlife species such as hippopotamus and resulted in the establishment of a number of wildlife protected areas and associated tourism, especially in Zimbabwe (WCD, 2000). The Diama dam in Senegal, and the Manantali dam in upper valley Mali on the Senegal River basin in the 1980s constructed for hydropower and agricultural production, caused a reduction in the Senegal River flow. While environmental impact analysis was conducted before building the two dams, performance constraints and environmental issues were not adequately addressed by the assessment (Degeorges and Reilly, 2006). Hydrological impacts from the dams included disruption of the timing of peak flow and substantial loss of the floodplain and estuarine areas on the Mauritanian bank (Degeorges and Reilly, 2006). In addition, the dams and downstream barrage and embankments in the river disrupted an alternating flow of brackish and salt water that used to extend for more than 200 km from the sea, creating an ecologically conducive environment with high biodiversity. Ecological impacts from the dam and irrigation included reduced fish populations especially due to blockage of migratory paths for marine and freshwater species and invasion by the water weed *Pistia stratiotes*, (Drijver and Marchand, 1985; N'diaye, 1997; Lemly, Kingsford and Thompson, 2000; Duvail and Hamerlynck, 2003). Another case is the Kihansi hydropower project in Tanzania where a dam diverted water that flowed into a downstream wetland associated with the Kihansi River in Udzungwa Mountains (a biodiversity hotspot), leading to an extinction in the wild of one of the most endangered amphibians, the Kihansi toad *Nectophrynoides asperginis*. Again, no comprehensive environmental impact assessment was conducted before the construction of the dam and, hence, important biodiversity impacts were not taken into account (Channing et al., 2006).

Excessive surface water abstraction and pollution from irrigation and mining activities have affected several rivers such as the Olifant River that supplies water to Kruger National Park located downstream, and have led to water scarcity and metal contamination that adversely affected aquatic ecosystems in the park (Smit et al., 2013). However, there is little evidence for the impacts of abstraction on downstream biodiversity, especially in East Africa. In Kenya, a country whose land is largely arid and semi-arid, more than 60% of the water is over-abstracted in the upper Ewaso Ng'iro River during the dry season, and of this abstracted water 40-98% is unauthorised (Gichuki, 2002). Similarly, unsustainable water abstraction in Mount Kenya is documented (but not quantified) to cause harmful impacts on biodiversity (Liniger et al., 2005). Aeschbacher et al. (2005) drew attention to a water shortage on the slopes of Mount Kenya and the adjoining lowlands due to unsustainable and illegal water extraction for supplying domestic and irrigation activities in areas with high rate of population growth. In Naro Moru River in the slopes of Mount Kenya for instance, less than 20% of the total water extraction adheres to legal requirements and thus the environment and downstream users are left without adequate water. The Ewaso Ng'iro and Naro Moru are among the catchments in the mountain ecosystem whose ecology is seriously affected by the unsustainable water abstraction, but the extent of the ecological impact is not clearly known. There is also an excessive dry season abstraction of water for irrigation in the upstream areas of the Great Ruaha and Katuma Rivers, and also for domestic use within Arusha National Park in Tanzania, resulting in downstream drying out during the dry season of these former perennial rivers (Mtahiko et al., 2006; Elisa et al., 2010, 2016; Stommel, 2016).

It is evident from these case studies that inadequate relevant scientific information has contributed to poor environmental assessments and/or the establishment of unsustainable water development projects and practices which in turn have significantly damaged natural environment in sub-Saharan Africa, including East Africa. In addition, insufficient political will and poor/lack of governance have contributed to the mismanagement of water resources as manifested in ineffective policies and implementation, lack of independent and comprehensive environmental assessments, as for instance, in the case of recently built Gibe dam III in Ethiopia (UNEP, 2010).

Excessive water abstraction often leads to water scarcity, which in turn is detrimental to the ecosystem and the wildlife habitat. Some of the adverse impacts in the wildlife rich semi-arid areas and African savannah include: escalation of human-wildlife conflicts occurring in most cases at the expense of the wild animals (Gichuki, 2002; Kikoti, 2009; Mariki et al., 2015; Elisa et al., 2010), and high risks of contracting infectious diseases (Ogutu et al., 2010). Scarcity of surface water in the semi-arid areas and African savannah often increases dry season animal congregations around the remaining water sources which enhance the risk for disease transimission and also contribute to the degradation of riparian and adjacent vegetation through overgrazing and soil trampling (Allsopp et al., 2007; Ogutu et al., 2010). For instance, in the Lake Naivasha ecosystem in Kenya, overgrazing by livestock and wildlife has caused loss of riparian vegetation, soil degradation and lake sedimentation (Otiang'a-

Owiti and Oswe, 2007). A similar case has also been reported in a semi-arid water scarce and downstream Ruaha National Park, Tanzania where water scarcity is caused by excessive abstraction of the Great Ruaha River for irrigation farming in upstream areas (Epaphras et al., 2008).

Protected Areas (PAs) form the cornerstone of biodiversity conservation strategies of many nations (WWF, 2004). However, human-induced impacts on water resources are threatening biodiversity in both upstream PAs (those located in the upper catchments where surface water originates) and downstream PAs (those located in the lower catchments). Though less documented, water abstraction may also cause negative ecological impacts in the upstream areas. Over-abstraction of water by two dams at the outlet (White Nile River) of Lake Victoria in Uganda caused a decline in the lake surface area and consequentially of the papyrus wetlands, and then of tilapia fish as the papyrus wetlands are an important habitat for fish larvae. Over-abstraction also resulted into eutrophication; together these effects reduced the recruitment of tilapia by 80% in the papyrus-fringed Mlaga Bay in Rubondo Island National Park in Lake Victoria, Tanzania (Kiwango and Wolanski, 2008). Unsustainable abstraction within the upstream PAs can also be ecologically detrimental. Elisa et al. (2016) found that almost 70% of water is completely extracted in an upstream Arusha National Park during the dry season in 2012/2013, leading to a decrease in the downstream diversity of riparian vegetation, and influencing the distribution of large herbivores within the park. However, downstream PAs are more vulnerable to the change on surface water quantity and quality emanating from mismanagement of water resources taking place in the upper areas.

Surface freshwater plays a significant role in biodiversity conservation both within and outside of the wildlife protected areas. Availability of surface water strongly influences wild mammals space use, and abundance within or outside protected areas in sub-Saharan Africa (Chamaillé-Jammes et al., 2007; Shannon et al., 2009). Under natural conditions and where there are less or no direct human interference, herbivores would usually be associated with surface water. A study by de Beer & van Aarde (2008) in the southern African savannah, demonstrated that the size of elephant home range is affected by the density of water points and landscape heterogeneity, and that, such home range is usually smaller in the heterogeneous habitat and with high density of surface water points. Further, the findings of Muruthi and Frohardt (2006) and Kikoti (2009) suggest that the movements of elephants and other ungulates are often associated with surface water availability in the Kilimanjaro-Amboseli ecosystem. However, the influence of water on species is not uniform across all species, and largely depends on various species-specific factors such as degree of dependence on water, type of herbivore (e.g. grazer, browser, mixed feeders, omnivore), size and gut morphology (Redfern et al., 2003; Stommel, 2016). Besides, other factors such as vegetation quality and quantity (though may cause overgrazing if animals are forced to aggregate at a few remaining waterbodies in the river during the dry season) are also essential to wild herbivores and may thus contribute to their space use, distribution and migration (Redfern et al., 2003; Ndaimani et al., 2017). In addition, external factors such human activities and livestock may also affect access of wild animals to water as shown in a study by De Leeuw et al. (2001) in an unprotected arid area in northern Kenya where animal distribution was not associated with surface water because the animals were displaced by livestock that concentrate close to water points. Further, in the East Africa savannah, livestock are known to partially displace some wildlife species from accessing the water points (Ogutu et al., 2014). However, in the savannah, the wildlife is known to avoid the livestock only to the extent permitted by the availability of water and food resources (Valls-fox et al., 2018).

Surface freshwater is undeniably an important resource for meeting ecological and social-economic needs. However, despite being a critical resource for biodiversity and human well-being, its sustainability is jeopardised by the lack of or inadequate environmentally sustainable management and development strategies, in part due to insufficient scientific information. In many places of sub-Saharan Africa, only little is known about the changes in surface freshwater and the associated impacts on biodiversity. Many areas of high importance in biodiversity conservation are yet to receive sufficient attention with regard to their eco-hydrology. One such area is the Kilimanjaro landscape, which is one of the wildlife-rich landscapes located in northern Tanzania. The landscape consists of wildlife-protected areas, livestock ranches, farming lands and village settlement areas. There is a fast-growing rural population, and like in the rest of the country, management of water resources falls under basin boundaries which in this case are Pangani basin and the internal drainage basin (van Koppen et al., 2016). One of the key challenges in this landscape is the rapidly growing human population and activities causing a high competition pressure for natural resources, especially water resources, between different land uses particularly in the semi-arid areas. This in turn leads to excessive utilisation of resources and conflicts between different users. For instance, there is excessive water extraction to support irrigation farming mainly for vegetable, legume and cereal production. Rapid growth in human population plus poor governance and inadequate land use practices are among the factors behind natural resources degradation and depletion in this landscape (Istituto Oikos, 2011). At present, very little is known about the changes and status of the surface water especially with respect to water extraction in the Kilimanjaro landscape (Elisa et al., 2016). While there has been rapid increase in human population and associated pressure on natural resources, the impacts of such pressure on the quality and quantity of surface water resources and associated biodiversity is not clearly understood. Impacts of unsustainable irrigation farming and livestock grazing on the riparian vegetation and wild animals' access to water resources, especially in the wildlife rich areas of the landscpae, are poorly understood. It is acknowledged that there is insufficiency in terms of quantity and quality of data to enable robust assessment of the current eco-hydrological status as well as projections of the future water availability in the landscape (Said et al., 2019). The few existing studies, e.g. Røhr (2003), Kaseva and Moirana (2010), Mckenzie et al. (2010) and Ndalilo et al. (2020), have not addressed the nature and extent of the existing water extractions and the associated eco-hydrological impacts. Moreover, these and other studies have largely been confined to the windward southern slopes of Mount Kilimanjaro and Meru with relatively high rainfall and water availability, while disregarding the leeward northern side and its plains, which is a semi-arid, water scarce area but nevertheless extremely rich in wild herbivores. Consequently, this lack of knowledge is impairing sustainable water resources management essential for human well-being and biodiversity conservation. Therefore, this study examines the ecological impacts of water abstraction and changes on surface water availability in the Kilimanjaro landscape, Tanzania, taking into account both natural and anthropogenic factors. The overall aim is to provide information on existing impacts and to provide a rigorous baseline data set to help assess the impact of any subsequent change or proposed development and to inform wider water and biodiversity management decisions, policies and practices for sustainable water and biodiversity resources conservation and human development.

1.2 Thesis objective

The overall objective of the study is to examine the ecological impacts of water abstraction and changes in surface water availability in the Kilimanjaro landscape, Tanzania (Figure 1.1).

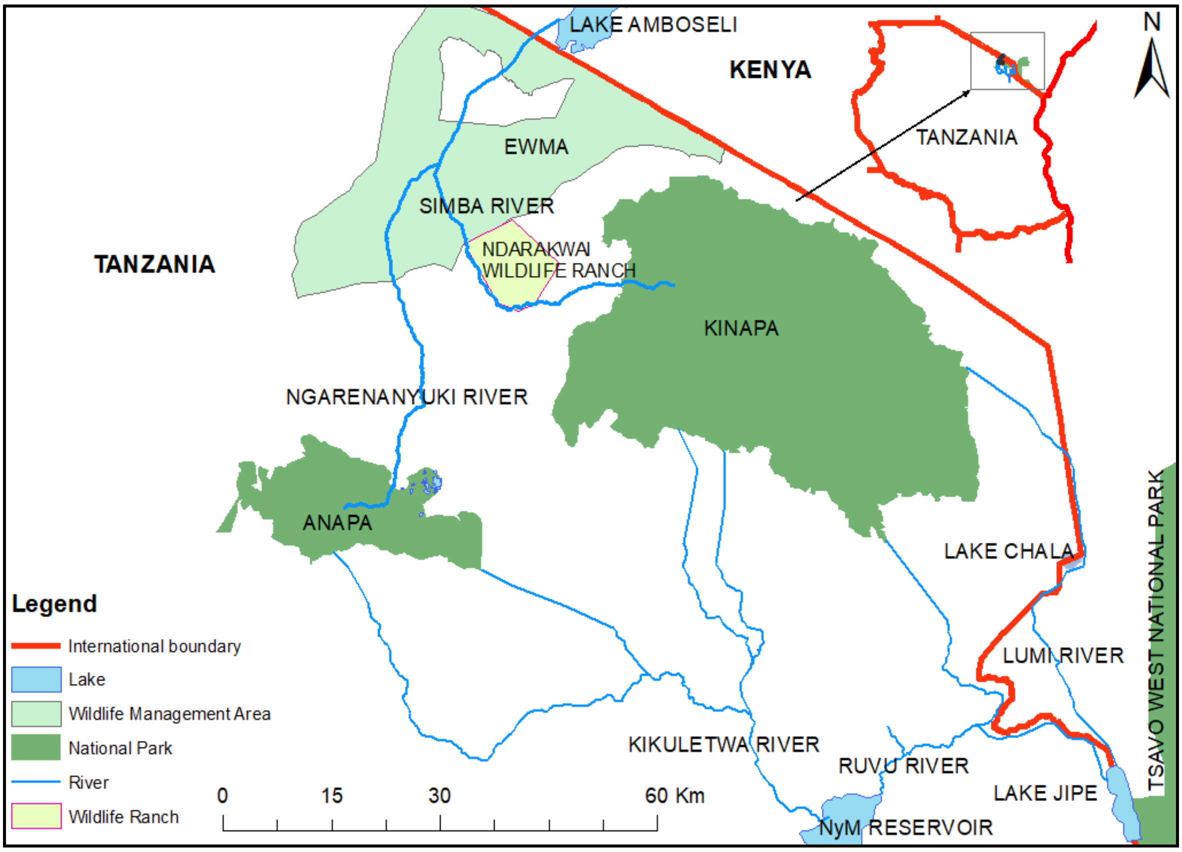

Figure 1. 1: Sketch map showing the study area, including the main surface water sources.

The study area encompasses most of Arusha National Park, and the north-western parts of Kilimanjaro National Park, both of which harbour a diversity of wildlife species and serve as key water catchments where water has been extracted for many decades to meet socio-economic needs. The study area also extends to the lowland semi-arid areas of western Kilimanjaro that are rich in wild animal species and some of surface water bodies around Mount Kilimanjaro and Meru that depend on water draining from these mountains.

The key parameters that were measured include water quality, quantity, rainfall, temperature and evaporation, yearly water budgets, riparian vegetation diversity, vegetation cover change, and herbivore density and distributions. In addition, water budgets were evaluated from the field based and satellite based data. A comparison was also made with the water budget of the Katavi-Rukwa ecosystem in western Tanzania, which suffers similar problems as in the Kilimanjaro landscape of upstream over-use of water that affects the downstream areas.

The study area is described below, followed by specific study objectives (that will be further be elaborated in respective chapters), then the structure of the thesis that describes a layout of the thesis.

1.3 Description of the study area

The Kilimanjaro landscape consists of a mosaic of environments, ranging from wet mountainous to semi-arid lowland areas. It straddles the border between Tanzania and Kenya. The landscape includes communal grazing land, small- to large-scale agricultural land, village settlement areas, and several conservation/protected areas, specifically National Parks, ranches, wildlife corridors, and a wildlife management area (WMA) (Figure 1.1; Kikoti, 2009). Human population is growing rapidly and is associated with poor governance and unsustainable land uses (Istituto Oikos, 2011). The main livelihood activities in the semi-arid areas are livestock keeping and farming for vegetable, legume and cereal (maize and wheat) production. The major uses of surface water are irrigation farming for cash crops (mainly vegetables and legumes), domestic use, and livestock watering and for sustaining biodiversity. Water management in the landscape falls under the Pangani and internal drainage basins authorities, which facilitate formation of water user associations as legal local water management bodies (van Koppen et al., 2016). However, the landscape is faced by competing and unsustainable uses of land and water resources among the different users. Available surface water largely results from run-off from Mount Meru in Arusha National Park (ANAPA) and Mount Kilimanjaro in Kilimanjaro National Park (KINAPA). ANAPA and KINAPA are mountainous parks that contain key water catchments on the slopes of Mounts Meru and Kilimanjaro respectively that supply freshwater to the entire landscape including the surrounding lowland semi-arid areas, which are rich in wildlife. Kilimanjaro is the highest mountain in Africa and the highest free standing mountain in the world, with its highest point 5895 m above sea level (Kaseva and Moirana, 2010). Both parks have high annual rainfall of up to 2200 mm in the southern slopes of Kilimanjaro (Røhr and Killingtveit, 2003). The remaining areas are semi-arid on the lowland leeward slopes of Mounts Meru and Kilimanjaro which receive annual rainfall ranging from 400 to 890 mm (Rey and Das, 1997; Kenya Wildlife Services, 2008; Kikoti, 2009). There are two ranches in the study area (West Kilimanjaro/NARCO (303 km^2) and Ndarakwai (44 km^2) and two wildlife corridors (Kisimiri that links ANAPA Park with the West Kilimanjaro- however this is severely narrowed by expanding human settlements and farming; and Kitendeni that links KINAPA in Tanzania with Amboseli National Park in Kenya). The community lands, wildlife management area, ranches and wildlife corridors are all characterised by thickets, woodland, and scrubland. They are located in low altitude semi-arid areas, and mainly depend on water draining from the mountainous parks (Kikoti, 2009; Elisa et al., 2016).

There are also several small freshwater lakes in the study area, including Lake Amboseli located in Amboseli basin in Kenya, and Lakes Chala and Jipe, which are both trans-boundary lakes between Kenya and Tanzania and located at the base of Mount Kilimanjaro. Lakes Jipe, Chala and Amboseli are in the wildlife transboundary wildlife ecosystems between Tanzania and Kenya, and therefore they are important source of water for a number of wildlife species (Ruwa et al., 2004; Njiriri, 2016). Lake Chala straddles the Tanzania and Kenya border at the foot (840 m a.s.l) of Mount Kilimanjaro covering a catchment of about 16.23 km^2 between longitude 037^0 29' E and 037^0 45' E and latitudes 030 6' S and 03^0 29' S (Mwega et al., 2013). This is a volcanic-crater freshwater lake with an area of about 4 km^2 and a maximum depth of 100 m. It has no surface inflow and outflow, and is thus fed by precipitation, small local surface run-off and underground water from Mount Kilimanjaro. Lake Chala loses some water through evaporation and a few springs (Payne, 1970). The lake is relatively clear and unpolluted as it is not exposed to significant human activities (Ruwa et al., 2004). Lake Jipe, which is bounded to the southeast by Tsavo West National Park (TSWENAPA) in Kenya (see Figure 1.1), is at an altitude of 700 m a.s.l, and with an area of about 30 km^2. Both groundwater and the Lumi River, which largely drains water from Kilimanjaro Mountain, supply the lake. However, the Lumi River has been excessively abstracted for irrigation farming, leading to reduced flow. Further, poor land-use has resulted in increased siltation of Lake Jipe (Ruwa et al., 2004; Ngugi et al., 2015). Some of the wildlife species that depend on Lake Jipe include crocodiles and hippos, with other species being various water-birds, elephants, zebras, impalas and gazelles (Ndetei, 2006. There is also one hydro-electric reservoir known as Nyumba ya Mungu (NyM) to the southeast of Mount Kilimanjaro in Tanzania (Payne, 1970; Ruwa et al., 2004; Ndetei, 2006; Njiriri, 2016). The NyM reservoir which is fed by Kikuletwa and Ruvu Rivers, is a source of water for hydro-electricity, and forms a lake. Its outlet is the Pangani River, which is of high ecological and socio-economic importance, supporting irrigation, fishing and livestock keeping activities, and is also a source of water for hydro-electricity further downstream (Murashani,2012; Lalika et al., 2015).

The Kilimanjaro landscape is a wildlife-rich area in northern Tanzania that harbours a number of wildlife species including elephant (*Loxodonta africana)*, eland (*Taurotragus oryx*), cape buffalo *(Syncerus caffer*), wildebeest (*Connochaetes taurinus*), zebra (*Equus quagga quagga*), Thompson's gazelle (*Eudorcas thomsonii*), grant gazelle (*Nanger granti*), giraffe (*Giraffa camelopardalis ssp. Tippelskirchi*),water buck (*Kobus e. defassa),* warthog (*Phacochoerus africanus)*, lesser kudu (*Tragelaphus imberbis*), impala (*Aepyceros melampus*), and striped hyena (*Hyaena hyaena*) (Kikoti, 2009). This ecosystem is also part of the larger West Kilimanjaro-Amboseli ecosystem and harbours wild carnivores such as wild dogs, cheetah and leopards; however, they are in decline and thus are rarely sighted (Kissui et al., 2012).

The study mainly examines the surface water and associated ecological issues for the following protected areas: Arusha National Park (552 km^2), Kilimanjaro National Park (1,665 km^2) and the Enduimet Wildlife Management Area (1100 km^2) in Tanzania, and Ndarakwai wildlife ranch (44.5 km^2) (Kikoti, 2009). It also examines water availability in the lakes Amboseli (seasonal), Jipe (30 km^2), Chala (4 km^2) and Nyumba ya Mungu reservoir (100 km^2) (Payne, 1970; Ndetei, 2006; MEMR, 2012; Nyingi et al.,2013).

1.4 Specific objectives

1. Assessment of the water availability in the Kilimanjaro landscape.

- Examine the impacts of natural factors and current water abstraction on changes in surface water availability with respect to discharge and volume.
- Examine the historical surface water quantity changes based on the data from manual river gauging, data loggers, satellite altimetry data and hydrological models.
- Compare water budgets for the key rivers and lakes between Kilimanjaro landscape in northern Tanzania, and Katavi-Rukwa ecosystem in the western Tanzania, focusing on the amount of water available and natural (rainfall, evaporation) and anthropogenic (water abstraction).

2. Assessment of the current water quality status in the Kilimanjaro landscape.

- Examine the water quality in the landscape with reference to standards/guidelines, in particular to what is preferred/tolerated by wildlife and vegetation.

3. Assessment of the impacts of water abstraction and changes in surface water availability on wild herbivores in the Kilimanjaro landscape.

- Evaluating the impacts of change in surface water on the distribution and abundance of wild herbivores in the Kilimanjaro landscape.

4. Assessment of the impacts of change in surface water on the riparian and floodplain vegetation in the Kilimanjaro landscape.

- Using field observations, determine the changes in vegetation cover in the lowland semi-arid areas, and the diversity of riparian vegetation in ANAPA, resulting from the water abstraction in the upstream areas.
- Using Landsat satellite imagery, examine the change in riparian and floodplain vegetation communities in the dry areas of the Kilimanjaro landscape.

1.5 Structure of the thesis

The thesis is organised into five additional chapters. Chapter 2 examines the surface water availability, extractions, and budgets in rivers and freshwater lakes in the Kilimanjaro landscape based on the current and past information and mainly with respect to wildlife needs. It also gives a brief comparative view of the surface water budget between Kilimanjaro landscape in northern Tanzania and Katavi-Rukwa ecosystem in western Tanzania, as the two contrasting wildlife ecosystems that are challenged by a growing water crisis from excessive river water abstraction. Chapter 3 provides an assessment of the water quality status in the Kilimanjaro landscape with an emphasis on the wildlife needs. Chapter 4 examines the impacts of surface water change on the herbivores' distribution and abundance with a main emphasis on the dry, wildlife-rich areas north of ANAPA and KINAPA. Chapter 5 presents the impacts of surface water changes on the riparian and floodplain vegetation based on both historical and current status, and focusing on Arusha National Park, and the low-lying semi-arid areas. Finally, Chapter 6 provides the thesis synthesis.

1.6 References

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Chapter 2: Assessment of water availability in the Kilimanjaro landscape

Abstract

As surface water quantity is changing due to both natural and anthropogenic factors, an understanding of the nature and contribution of each factor, the extent and impact of the resulting water change is crucial for guiding the sustainable management of the water and biodiversity resources. However, the impact of the change in the quantity of surface water, has not received sufficient attention, particularly in sub-Saharan Africa (Duda and El-Ashry, 2000; Stommel et al., 2016). In particular, the evidence available for informing sustainable water use policy in and around National Parks in Tanzania is extremely limited. Water from several rivers within and outside of protected areas is abstracted for domestic use, and irrigation farming as the rivers traverse community lands in the low-altitude areas (Mnaya et al., 2021), and streamflow and water use data are often lacking. The resulting reduced flows have negative impacts on the human community and the environment particularly in the downstream areas. Thus, even with the best intentions, many recommendations for sustainable water use are often based on guesswork and intuition rather than evidence. In particular, at present little is known about the changes and status of the current surface water regime in the Kilimanjaro landscape (Elisa et al., 2016). This chapter quantifies natural impacts, and anthropogenic water abstraction and the resulting change in surface water availability to contribute to the improvement of water and biodiversity management in the Kilimanjaro landscape. The chapter also presents a comparative view of the water budget evaluation for the key rivers and lakes between Kilimanjaro landscape in northern Tanzania, and Katavi-Rukwa ecosystem in the western Tanzania, both ecosystems suffering similar anthropogenic impacts on water. Surface water availability was assessed by measuring the amount of water available, extracted and that left for the environment in rivers and streams in Arusha National Park (ANAPA), Kilimanjaro National Park (KINAPA) and in the surrounding wildlife rich dry areas. In addition, water budget for freshwater lakes and waterholes in the lowland semi-arid wildlife areas was evaluated. Finally, long-term rainfall data and the Southern Oscillation Index (SOI) data were examined to establish the influence of climate change on surface water in the landscape.

This study demonstrated that the spatial and temporal changes in the availability of surface water in Kilimanjaro landscape are dependent on both natural and anthropogenic factors, but their relative importance varied. There is no evidence that the mean annual rainfall changed significantly in recent since the 1970s when data became available in the study areas. Anthropogenic impacts did however result in a significant reduction in the river discharge with distance downstream, as water is excessively abstracted mainly for irrigation farming. The existing water abstraction is excessive and unsustainable as it causes increasing and serious deprivation of water to the downstream areas populated by people, livestock and wildlife. Such unsustainable water abstraction is likely to have far-reaching social-ecological and economic impacts. There is an urgent need for establishing ecologically sustainable water resources management plan and practices at the watershed scale in the entire Kilimanjaro landscape.

2.1 Introduction

While the impact of climate change is an important factor affecting water availability, studies have shown that freshwater availability is, and in the future will still be more affected by over-abstraction rather than climate change (Vörösmarty et al., 2000; Grafton et al.,2013). An increasing number of unsustainable water development projects that significantly regulate flow regime and over-abstract water to meet the growing human demand has substantially affected most of the surface freshwater sources in the sub-Saharan Africa (Drijver and Marchand, 1985; Vörösmarty et al., 2000; Ndetei, 2006; Grafton et al., 2013; Mnaya et al., 2021; Elisa et al., 2021). While the surface water in sub-Saharan Africa is usually subject to seasonal fluctuations, existing water abstractions often take practically high and constant amount of surface water without consideration to seasonal fluctuations in flows, leading to deprivation or shortage of water to the downstream areas during the dry seasons (Gichuki, 2002; Zwarts et al., 2005). In Kenya, a country with over 80% of its land classified as arid and semi-arid but whose economy is largely dependent on agriculture, the Upper EwasoNg'iro River basin experiences water abstraction of 60 to 80 % of the available water in the upstream areas during the dry season, and this results in serious water shortage in the downstream areas (Gichuki, 2002). In the semi-arid and arid areas of Kenya and Tanzania, over-abstraction is known to increase frequency of zero flows downstream of rivers, especially during the dry seasons (SMUWC, 2001; Elisa et al., 2010; Grafton et al., 2013; Mnaya et al., 2021). One such case is the Great Ruaha River that supplies water for Ruaha National Park in Tanzania; the river was originally perennial but, as a result of rice irrigation in the upstream areas, it has dried out in the downstream areas in the dry season each year for up to 111 days between 1990s and 2000s (SMUWC, 2001). In addition, excessive water abstraction often leads to a decline in lake levels and areas, in many parts of sub-Saharan Africa (Lemly et al., 2000; Zwarts et al., 2005). Unsustainable water abstraction from rivers that drain to Lake Chad accounted for the 50% decrease in the lake area since 1960s and 1970s (Coe and Foley, 2001). Likewise over-abstraction of water for irrigation in the upstream areas of the Katuma River in Tanzania is linked to a decline of water level in Lake Rukwa by about 4 m since 1992 (Elisa et al., 2021).

Such shortage or deprivation of water in the downstream areas in turn leads to a number of adverse ecological impacts on both aquatic and terrestrial ecosystems, such as disruption of wildlife movements, space use, change in behaviour, and degradation of wetlands and riparian vegetation (Richardson et al., 2007; Stommel, 2016).

Insufficient data plus a lack of benchmark information on eco-hydrology is indicated by several studies as one of the main factors contributing to unsustainable water development projects that do not comprehensively take into account ecological issues in sub-Saharan Africa (Drijver and Marchand, 1985; WCD, 2000; Conway et al., 2009; Viviroli et al., 2011). The available hydrological data are insufficient to enable a robust assessment of the current hydrological status as well as projections of the future water availability in the Kilimanjaro landscape (Said et al., 2019). While the landscape is faced with a rapidly growing human and livestock population, and associated pressure on the freshwater resources (Mbonile, 2005; Munishi et., 2009), very little is known as to the degree of surface water extraction, and its impact on the water quality, quantity and the ecology of the landscape. Specifically, it is not clearly known how much water is available, extracted, and left for the downstream ecosystems in the Kilimanjaro landscape. There were a few studies that focused mainly on changes in hydrology (quantity and quality) as a result of land use changes and land degradation, pollution, geological interactions and climate change (Røhr, 2003; Kaseva and

Moirana, 2010; Mckenzie et al., 2010; Ndalilo et al., 2020). However, none of these studies quantified water extractions by humans, nor did they assess the resulting impacts on the wildlife and livestock in the downstream areas during the dry season when water is a limiting factor. Moreover, in the Kilimanjaro landscape, the few available studies have been confined to the windward southern slopes of Mount Kilimanjaro and Meru where rainfall and surface water availability are relatively high, and have neglected the leeward northern side, which is a semi-arid, water-scarce area that is, however, extremely rich in wild herbivores. The windward side is relatively accessible, has high socio-economic importance, and hence attracts more attention, but higher rainfall and associated larger amounts of surface water might possibly contribute to less attention from studies focusing on water budget evaluation. Insufficient information and knowledge on surface water extraction and change, and related ecological impacts, has a great potential to impair effective policy and management of water resources for the benefit of both people and wildlife in the Kilimanjaro landscape. Thus, before this thesis, there were no reliable data on how much water is available, is extracted, and is left for the downstream ecosystem, and how the existing water extraction is impacting wetlands, plant species, and distribution and abundance of the terrestrial mammals in the Kilimanjaro landscape. My study aimed at quantifying the impacts natural factors and anthropogenic water abstraction on the surface water availability to the ecosystems in Arusha National Park (ANAPA), Kilimanjaro National Park (KINAPA), and the dry wildlife areas around Mounts Meru and Kilimanjaro (collectively called the Kilimanjaro landscape). It also examines the temporal changes in the water budget in the fresh-water lakes Jipe, Chala, Amboseli and Nyumba ya Mungu dam which receive water that largely drains from the National Parks. In addition, it also presents a brief comparative view of the surface water (key rivers and freshwater lakes) budget between the Kilimanjaro landscape and the Katavi-Rukwa ecosystem in western Tanzania; these are two contrasting wildlife ecosystems, which are challenged by a growing water crisis from excessive river water abstraction. In all cases, the emphasis in terms of sampling was given to the dry season when water is scarce and thus a limiting ecological factor. Therefore, the question addressed by this chapter is: What is the impact of natural factors and anthropogenic water abstraction on the dry season water availability in the downstream areas of the Kilimanjaro landscape? It is hypothesized that the dry season availability of surface water in the downstream areas of the Kilimanjaro landscape is affected much more by the anthropogenic water abstractions in the upstream areas than by natural factors.

2.2 Methodology

2.2.1 Description of the study Area

The Kilimanjaro landscape is a wildlife-rich area in the northern Tanzania. The landscape includes four fresh-water lakes; Jipe, Chala, Amboseli and the man-made Nyumba ya Mungu formed by damming downstream of the confluence of the Kikuletwa and Ruvu Rivers (Figure 2.1). All these water bodies receive water that drains Mt. Kilimanjaro and/or Meru. The landscape is also home to several protected and non-protected areas including Arusha National Park (ANAPA, 552km^2), and Kilimanjaro National Park (KINAPA, 1,665km^2), NARCO livestock ranch (303 km^2), the Ndarakwai (44.5 km^2) wildlife ranch and the Enduimet wildlife management area (1100 km^2), plus two potential wildlife corridors; Kisimiri (that links ANAPA to West Kilimanjaro area, although it is now largely blocked by expanding human settlements and farming) and Kitendeni (that links KINAPA to the Amboseli basin in Kenya; Kikoti, 2009). This landscape which is a trans-boundary ecosystem straddling the Kenyan and Tanzanian border and it harbours a number of charismatic wildlife species including elephant (*Loxodonta africana)*, buffalo*(Syncerus caffer*), wildebeest (*Connochaetes taurinus*), zebra (*Equus quagga*), Thompson's gazelle(*Eudorcas thomsonii*),Grant's gazelle (*Nanger granti*), giraffe (*Giraffa camelopardalis ssp. Tippelskirchi*), lesser Kudu(*Tragelaphus imberbis*), striped hyena(*Hyaena hyaena*), and leopard (*Panthera pardus*) (Kikoti, 2009). It also contains communal grazing land, small- to large-scale farms, and human settlements (Kikoti, 2009). ANAPA and KINAPA are mountainous parks and encompass important water catchments on the slopes of Mount Meru and Mount Kilimanjaro respectively. Kilimanjaro is the highest mountain in Africa and the tallest free standing mountain in the world, with its highest point 5895m above sea level (Kaseva and Moirana, 2010). Both parks have high annual rainfall of up to 2200 mm on the southern windward slopes of Kilimanjaro (Røhr and Killingtveit, 2003), while annual rainfall is much lower at about 1480 mm in ANAPA (ANAPA, 2020). The leeward, northern slopes of Mount Meru and Kilimanjaro are semi-arid with annual rainfall ranging from 400 to 890 mm (Rey and Das, 1997; Kenya Wildlife Services, 2008; Kikoti, 2009).

The community lands, wildlife management area, ranches, and wildlife corridors are all characterised by thickets, woodland and scrubland. They are located in the leeward, lowland semi-arid areas, and all depend on water draining from the mountainous parks (Kikoti, 2009; Elisa et al., 2016). The main perennial sources of water in these downstream dry areas are the Ngarenanyuki and the Simba Rivers that drain water from Mount Meru (ANAPA), and Mount Kilimanjaro (KINAPA) respectively. Water from these rivers, which also supply water for domestic, livestock and wild animal use, is excessively abstracted for vegetable irrigation farming (which started in earnest in the 1990s) in the upstream villages particularly during the dry season. As a result the Ngarenanyuki River that historically used to flow all the way to the Amboseli basin in Kenya during the dry season, does not now do so now (Istituto Oikos, 2011). According to 2018/2019 high resolution Google Earth images, the irrigated areas along the Ngarenanyuki and Simba Rivers were about 90 km^2 and 25 km^2 respectively. According to Istituto Oikos (2011), the irrigated area for tomato production mainly covered about 90% and 80% of all cultivated land in the Uwiro and the Ngabobo villages which are located upstream along the Ngarenanyuki River. Further, the area has a very high human population growth rate. For instance in 2002, the Arumeru district where ANAPA and large part of the Ngarenanyuki River are located, recorded a population growth rate of 3.1%, which was higher than the national 2.9% (NBS, 2002). In 2002, the Meru district council recorded a total population of 225,000, and in 2012, the population has grown to 268,144, implying an increase of almost 20% in 10 years. Similarly, in the upstream region of the Ngarenanyuki River, the Ngarenanyuki ward recorded a population of 16,939 in 2002, and in 2012 this population had grown to 20,379, which again represents a 20% increase in population in 10 years (NBS, 2002, 2012).

There was a number of water extraction sites in the National Parks and about 10 water extractions canals were located on the Ngarenanyuki ad Simba Rivers (Figures 2.1 and 2.2)

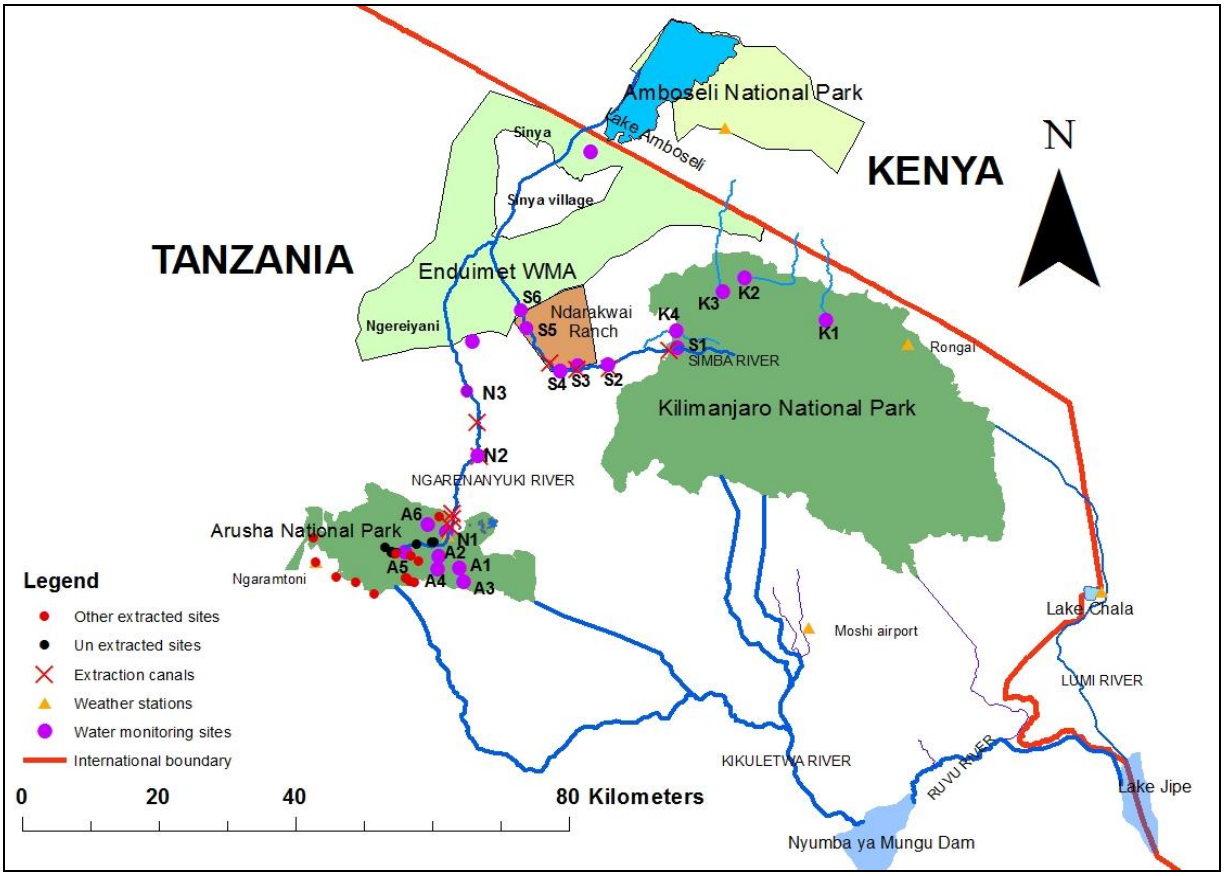

Figure 2. 1: Sketch map showing the study area and location of water extraction and monitoring sites.

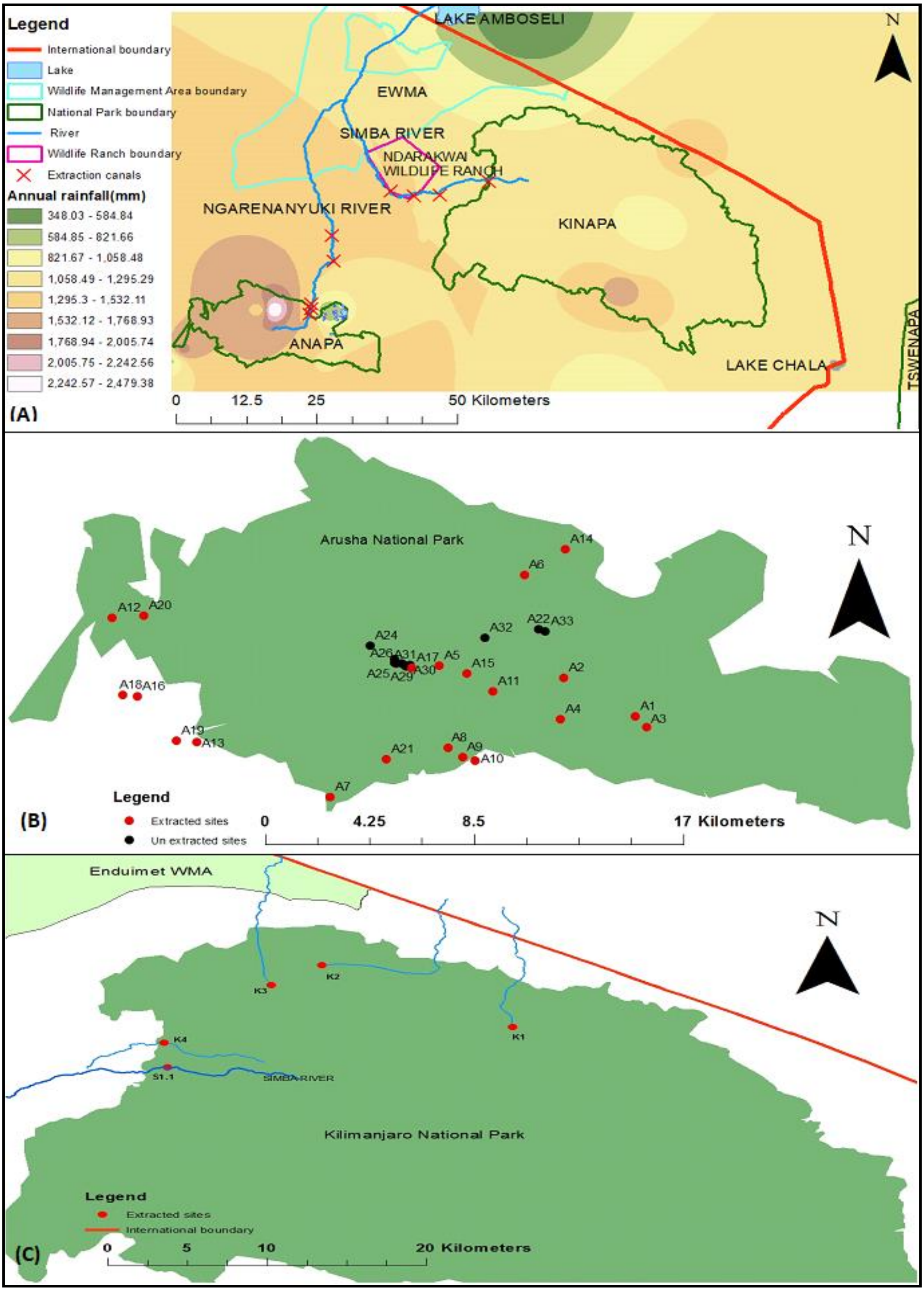

Figure 2. 2: (A) The rainfall distribution in the Kilimanjaro landscape. A zoom in location map of the extracted and un-extracted sites that were examined in this study in (B) ANAPA, and (C) KINAPA

2.2.2 Identification of study sites

The selection of sampling sites was informed by discussions with the park management and rangers, and took into consideration the degree of water extraction for human use. Therefore, the selected water sources were sites either of water abstraction, or likely to be in the future abstracted for human use, and/or consumed by wild animals and/or sustaining wetlands, within and outside the National Parks (Table 2.1, Figures 2.1 and 2.2). The selection of the monitoring sites also ensured a representative sample from the leeward and windward catchments due to different levels of precipitation and hence hydrology. Sites where water changes deemed to affect biodiversity were included. Site selection was informed by experience from the previous study by Elisa et al. (2016) which encompassed more than 50% of the areas where water over-abstraction is impacting biodiversity in ANAPA. In ANAPA, the current study hence included most of these sites, in addition to a few others, to ensure adequate representation of the current water extraction in the park.

The study focused on the wildlife-rich semi-arid areas of West Kilimanjaro, ANAPA and the adjacent forest, plus the north-western side of KINAPA where there is a high diversity and density of herbivore species and where water extraction is taking place but is poorly monitored. Because of the low abundance and diversity of large herbivores in the rest of KINAPA and of the limited budget, this study did not cover the entire park. A total of 21 sites subject to extraction and 12 sites without water extraction in ANAPA, and 5 sites with water extraction in the north-western parts of KINAPA were identified (see Table 2.1 and Figure 2.2). The north-western area of KINAPA, which was the focal area for this study, had fewer surface water sources. Other sites were located in the semi-arid but wildlife-rich areas of West Kilimanjaro (Figure 2.2A), which consist of several protected and non-protected wildlife areas, ranches, and wildlife corridors, where the main sources of water are the perennial Ngarenanyuki and Simba Rivers that respectively drain Mount Meru in ANAPA, and Mount Kilimanjaro in KINAPA. Along these two rivers, a total of 9 sites located in the upstream and downstream were identified, and of these 6 sites, i.e. 3 sites in each river, were systematically monitored; two sites were located upstream of the water extraction points, and the other sites were located downstream of the water extraction points.

Being on the leeward side, the focal area of West Kilimanjaro is often water-scarce. In order to cope with water scarcity, local communities have dug several water holes and built troughs for livestock watering. These either collect rainfall water and/or receive water from the Ngarenanyuki and Simba Rivers. Water volume in the waterholes varies markedly ranging from no water in the peak of the dry season to almost 20,000 m^3 during the wet season. Wild animals also drink in these man-made water sources, usually during night-time. Therefore, 12 water holes were identified and the amount of water available quantified over time. In addition, 4 fresh water lakes were also identified for water level monitoring; Lakes Chala, and Amboseli on the northern side of Mt. Kilimanjaro, and Jipe, and Nyumba ya Mungu reservoir on the north-eastern and south-eastern side of Mt. Kilimanjaro. As these lakes receive water from Mt. Kilimanjaro and Meru, they are good reference sites with regard to the changes of surface water availability in the ecosystem. Thus, the main inflow and outflow rivers associated with these lakes were identified and their discharge data were obtained.

Water quantity in terms of volume, or discharge was measured either directly or using stream discharge data published by the Government of Tanzania (Pangani Basin Water Office (PBWO), 2020) and/or remote sensing data (satellite images and altimetry-derived satellite data) for all of the identified sites, with an emphasis on the dry season when water is scarce. All identified water sampling sites were recorded using a hand held GPS unit (64sx) and mapped using ArcGIS software 10.4.1.

Table 2. 1: River and stream sampling sites showing the location, the distance from the first sampling point and whether there was water extraction upstream.

Area	River	Site no.	Site name	Coordinates		Distance (km) from first sampling site	Remarks
				Lat	Long		
ANAPA	Ngarenanyuki	N1	Upstream	-3.23577	36.847344	0	In the park before abstraction
		N2	Ngabobo	-3.129475	36.891015	15	Main river downstream
		N3	Madebe	-3.044362	36.864067	26	Main river downstream
		A14	Nasula	-3.212339	36.840517	Tributary	Extracted, downstream of N1
		A22	Arc tree	-3.245922	36.830798	Tributary	Un-extracted
		A23	Crater I	-3.258442	36.778079	Tributary	Un-extracted
		A24	Crater II	-3.252662	36.769174	Tributary	Un-extracted
		A25	Crater III	-3.259856	36.778056	Tributary	Un-extracted
		A27	Crater IV	-3.260185	36.778653	Tributary	Un-extracted
		A28	Crater V	-3.26069	36.781616	Tributary	Un-extracted
		A29	Crater VI	-3.261297	36.782241	Tributary	Un-extracted
		A30	Crater VII	-3.261109	36.783081	Tributary	Un-extracted
		A31	Crater VIII	-3.26086	36.783611	Tributary	Un-extracted
		A32	Maiyo	-3.251928	36.807165	Tributary	Un-extracted
		A33	Malama	-3.2487	36.829507	Tributary	Un-extracted
ANAPA	Kikuletwa	A1	Ngongong are1	-3.282681	36.866132	Tributary	Extracted
		A2	Mweka	-3.266374	36.839918	Tributary	Extracted
		A3	Ngongong are3	-3.286907	36.870107	Tributary	Extracted
		A4	Kilinga	-3.283571	36.838666	Tributary	Extracted
		A5	Malemeo	-3.261086	36.794541	Tributary	Extracted
		A6	Mwakileng a	-3.223177	36.825629	Tributary	Extracted
		A7	Bangata	-3.31607	36.754476	Tributary	Extracted
		A8	Kikololomu	-3.295568	36.797456	Tributary	Extracted
		A9	Kira hill	-3.299325	36.802794	Tributary	Extracted
		A10	Lewate	-3.300897	36.807531	Tributary	Extracted
		A11	Maambure ni	-3.271821	36.81403	Tributary	Extracted
		A12	Nading'oro	-3.240897	36.674902	Tributary	Extracted
		A13	Narok	-3.292865	36.705664	Tributary	Extracted
		A15	Nshupu	-3.26458	36.804375	Tributary	Extracted
		A16	Olmotoni	-3.273835	36.68398	Tributary	Extracted
		A17	Sajona	-3.262055	36.78432	Tributary	Extracted
		A18	Sambasha	-3.273131	36.678527	Tributary	Extracted
		A19	Masaga	-3.292435	36.698169	Tributary	Extracted
		A20	Nading'oro	-3.239962	36.686314	Tributary	Extracted
		A21	Seela	-3.300321	36.774981	Tributary	Extracted

Table 2.1: Continued

Area	River	Site no.	Site name	Coordinates		Distance (km) from first sampling site	Remarks
KINAPA	Simba	S1	'Upstream'	-2.983864	37.117039	0	In the park prior to abstraction
		S1.1	Longido project	-2.983772	37.116799	1	Extraction point in park
		S3	Mitimirefu	-3.007739	37.023086	16	Main river downstream
		S4	Ndarakwai	-3.015586	37.000166	20	Wildlife ranch
		S5	Tingatinga	-2.956694	36.955889	29	Main river downstream
		S6	Enduimet WMA	-2.932757	36.948795	32	River within the WMA
KINAPA	Kamwanga	K1	Kamwanga	-2.94703	37.349895	-	Extracted
	Kitendeni	K2	Kitendeni	-2.889967	37.242284	-	Extracted
	Lerangwa	K3	Lerengwa	-2.908208	37.213565	-	Extracted
	Simba trib.	K4	Londorosi	-2.961003	37.152938	-	Extracted

2.2.3 Methods for surface water quantity assessment

Assessment of water quantity was conducted in ANAPA, KINAPA and in the surrounding wildlife rich lowland semi-arid areas to determine the amount of water prior to extraction, the extracted water, and the water remaining for the environment. In addition, a water budget evaluation was conducted for the four freshwater lakes in the landscape.

Rivers, water holes and lakes gauging

Spot gauging using a stream flow meter was used to establish a flow rating curve for each river that was monitored. From regularly obtained water level data, the rating curve was used to calculate the discharge in the identified rivers and streams to estimate the amount of water available, removed, and that left to flow downstream. The choice of the technique varied depending on the geomorphological nature and size of the extracted stream/river. To measure water discharge in a river or stream, a stream flow meter (Geopacks) was used to measure velocity, water depth using a meter ruler and wetted widths using a tape measure. Very small streams that could not be measured in this way were subjected to volumetric method, which involved tapping of all flowing/falling water in container of specific volume, and recorded using stop-watch, the time it takes to fill the container to know volume per time. At the extraction sites, discharge for each water source was measured before extraction to find total available water and then immediately downstream of the extraction point to estimate the amount of water remaining downstream. The difference in the amount of water before and after extraction is the amount of water extracted. In addition to spot sampling, one year-long time-series water level data were collected using water level loggers (HOBO) installed in two upstream points (sites N1 and S1 in Figure 2.1) within the parks, one in the Ngarenanyuki River and the other in the Simba River. To estimate the Lumi River mean inflow (discharge) to the Lake Jipe, the river discharge was calculated by using the Kikuletwa River discharge data and adjusting it for the respective forest drainage areas and rainfall over their forest areas for a period of one year starting from October 2018. Drainage areas were estimated from high-resolution Google Earth images. For those lakes in the Kilimanjaro landscape (i.e. Lake Jipe and Chala) with no satellite altimetry data, water levels data were obtained from water level loggers (HOBO) installed in each lake for one year starting from October 2018. Water loggers were programmed to record data at hourly intervals, and then placed in a strong metal frame that allows free water movements but protect it against any physical damage. The loggers, which were set to start recording after deployment, were firmly placed in the bottom of the water bodies and at a location with minimal human disturbances, so as to reduce the risk from theft and vandalism. The depth at the points of logger installation were measured during logger deployment and at removal, and these data were used in the calculation of the datum (calibration value/point) from the logger records. Long-term water levels data from manual gauging were obtained for the Simba River downstream at Ndarakwai and the Ngarenanyuki River upstream within the park for one year during the study period. Also, at daily intervals, manual gauging data were obtained for Lake Chala from 2011 from the Pangani Basin Water Board-Moshi (2020). These manual gauging data supplemented and further facilitated the calibration and processing of the water logger data. Satellite altimetry (lake level) data for Lake Amboseli (from 2008) and Nyumba ya Mungu dam (from 2008), were obtained from USDA (2020). In addition, the wetted lake surface area was obtained from occasional, cloud-free satellite Landsat and Sentinel images. All these data were merged to quantify the water level variations and the water budget over a period of 10 years.

The water volume in the water holes located in the dry areas was estimated by measuring the mean depth using a graduated metal rod and the surface area area using a hand held GPS unit to record coordinates of the water perimeter that were used to calculate area by the function area of the Arcmap 10.4.1. Water volume was calculated as the area times the mean depth. In case of no water, the water holes were classified as 'dry'.

Water budget

The water budget of the freshwater lakes was based on the mass balance formula in Elisa et al. (2021):

$$dV/dt = \text{Inflow} - \text{Outflow} \quad (1)$$

where V is the lake water volume, t is the time, d is the differentiation,

Inflow= Rainfall over the lake + Groundwater inflow + River inflow (2)

Outflow= Evaporation from the lake + Groundwater outflow + River outflow (3)

Data for all the parameters, except one that was then calculated from Equation (1), were collected from several sources including satellite altimetry data (i.e. Lake Amboseli and NyM reservoir), water level loggers (i.e. Lakes Chala and Jipe), river and lake gauging, and meteorological stations. Where manual readings on lake/river water levels were available, they were combined with logger readings, and calibrated to obtain an accurate long-term time-series data (i.e. Lake Chala). High resolution Google Earth images were analysed to estimate the area under irrigation farming for the Ngarenanyuki and Simba Rivers. Using the irrigated area, the amount of water extracted from each river for irrigation farming in the dry season, average annual rainfall, and the run-off coefficient, it was possible to estimate the amount of rain water retained in the irrigation farming, and also the amount of river water consumed per km^2 of irrigation farming (Elisa et al., 2010; Elisa et al., 2021).

The amount of rainwater retained in the irrigated farms was calculated as Rainfall x Irrigated area x Run-off coefficient.

The amount of river water consumed per km^2 of irrigated farms during the dry season was calculated as the amount of water extracted from the river in the dry season/total dry season irrigated area

Climate change

To examine the influence of climate change on surface water availability, long-term rainfall data-sets, dating back to 1970 and 1988 were collected from three nearby meteorological stations in the Amboseli National Park, and stations in the Arusha and Kilimanjaro regions, and they were then analysed to identify trends. In addition, the long-term (from 1988) maximum and minimum air temperature was collected from Moshi airport station, and also from Amboseli National Park (since 1997) and then analysed for trends. Data on the Southern Oscillation Index (SOI) since 1876 were collected online from Australia Bureau of Meteorology (ftp://ftp.bom.gov.au/anon/home/ncc/www/sco/soi/soiplaintext.html), and they were correlated with rainfall and lake levels to further examine if climate change might be impacting on the natural changes in surface water availability in the ecosystem.

Data analysis

Data on water quantity were presented as mean values and subjected to descriptive and regression analysis, performed on MS Excel. Data were also subjected to a generalised linear model on a statistical package-R software (version 3.6.1) to elucidate trends and key relationships. The parameters that are likely to influence a change in water quantity were included when carrying out the modelling, e.g. for lakes; the change in lake water level was taken as a function of inflow (groundwater inflow, surface water inflow and rainfall) and outflow (surface water outflow, ground water outflow and evaporation). Regression analysis was mainly used to explore the nature and strength of the relationships among various parameters, but it was also preceded with correlation coefficient to determine if there was any significant relationship between two variables (Bhat et al., 2014). A t-test, mainly two-sample assuming unequal variances, and ANOVA two factor with replication, were used to explore spatial and temporal variations in surface water availability. To address the issue of data auto-correlation, e.g. in the case where data had less spatial or temporal independency, the data were averaged and the analysis carried out on the means (Crawley, 2005). A combination of ArcGIS (version 10.4.1) and cloud free Google Earth, and Landsat images were employed in mapping and quantifying of the extent of surface water bodies and irrigated areas along the Simba and the Ngarenanyuki Rivers downstream catchments, all of which resulted into important parameters for estimating the change in the availability of surface water in the ecosystem (Ndaimani et al., 2017).

2. 3 Results

2.3.1 Climate change and variability

Climate variables were examined to see whether there are any significant impacts on the availability of surface water in the study ecosystem. Figure 2.3 shows the monthly rainfall over the study period as recorded in ANAPA and KINAPA at the sites where rain contributes to the flow of rivers and streams within the parks. In turn these rivers and streams contribute surface or ground water to the downstream water bodies such as rivers and lakes located outside the parks. The rainfall also directly contributes to the flow of key rivers outside the parks such as the Ngarenanyuki and Simba Rivers. Only rain from KINAPA contributes to the Simba River, and only indirectly to the Ngarenanyuki River below its confluence with the Simba River.

Figure 2.3 shows a marked difference in rainfall between 2018 and 2019. Seasonal and inter-annual rainfall variability is evident in both parks with May and October recording relatively high amount of rainfall.

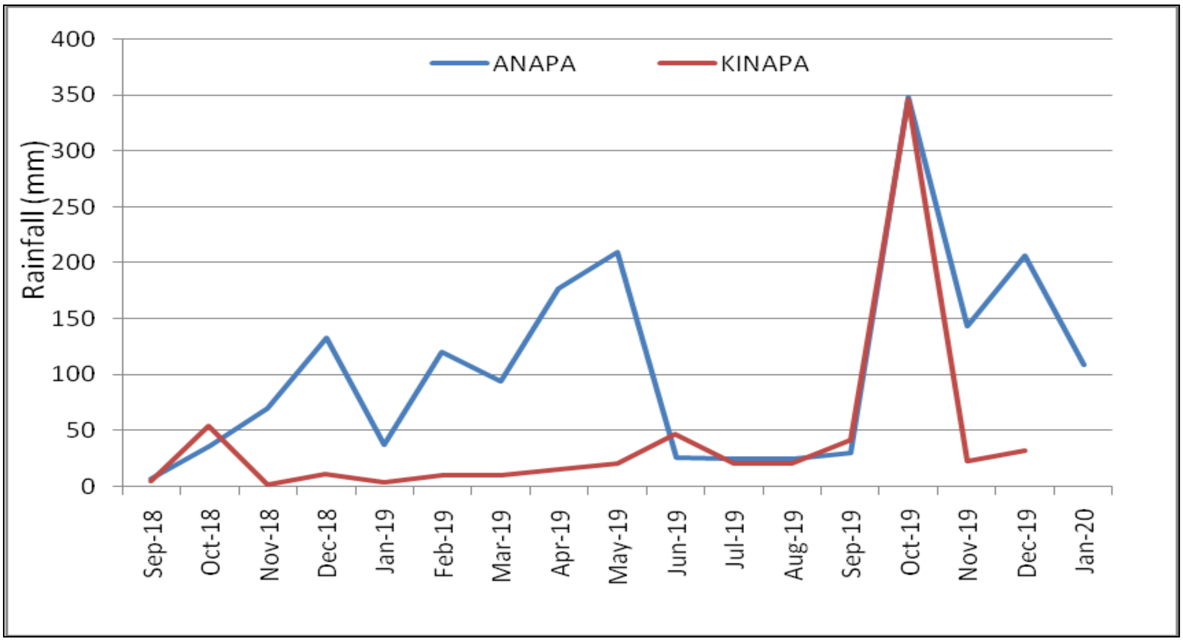

Figure 2. 3: Time series plot of the monthly rainfall in ANAPA (across 7 stations) and KINAPA (at Rongai- weather station), Sept 2018 – Jan 2020.

Source: (Arusha National Park (ANAPA), 2020; Kilimanjaro National Park (KINAPA), 2020).

Figure 2.4 shows the long (March-May) and the short (October-December) rains recorded over an extended time period from 1970s for the longest data set, in Arusha (Ngaramtoni), Moshi (Moshi airport) and Amboseli National Park, all these sites are in the Kilimanjaro landscape. The weather stations in Arusha and Moshi are located in the southern windward slopes, whereas the station in Amboseli is in the northern leeward plain (Figure 2.1). The long rains showed a relatively high inter-annual variability (SD=±146.51mm) compared to the short rains (SD=±109.34mm). Both the long and short rains show somewhat different long-term trends, and these trends are small and likely not significant compared to the high interannual variability. The long rains in the southern slopes at Arusha and Moshi stations showed a weak declining long-term trend (R^2= 0.02, β =-1.21 mm/year, p>0.05, t=-0.88, n=48, and R^2=0.05, β=-5.07, p>0.05, t=-1.19, n=32 respectively), while at these sites the short rains showed no clear long-term trend (R^2=0.0002, β=-0.14mm/year, p>0.05, t=-0.09, n=47, and R^2=0.04, β=2.09 mm/year, p>0.05, t=1.09, n=32 respectively). The long and short rains in Amboseli (northern slopes) also showed a weak long-term trend (R^2=0.01, β= 0.71 mm/year, p=0.48, t=0.71, n=43, and R^2=0.0006, β=-0.12 mm/year, p>0.05, t=-0.15, n=43 respectively). The annual average rainfall at Moshi was about 840 mm, 810 mm at Arusha and 350 mm in Amboseli, and the interannual variability was very large (SD= ±284 mm at Moshi, ±245 mm at Arusha, and ±130 mm at Amboseli).

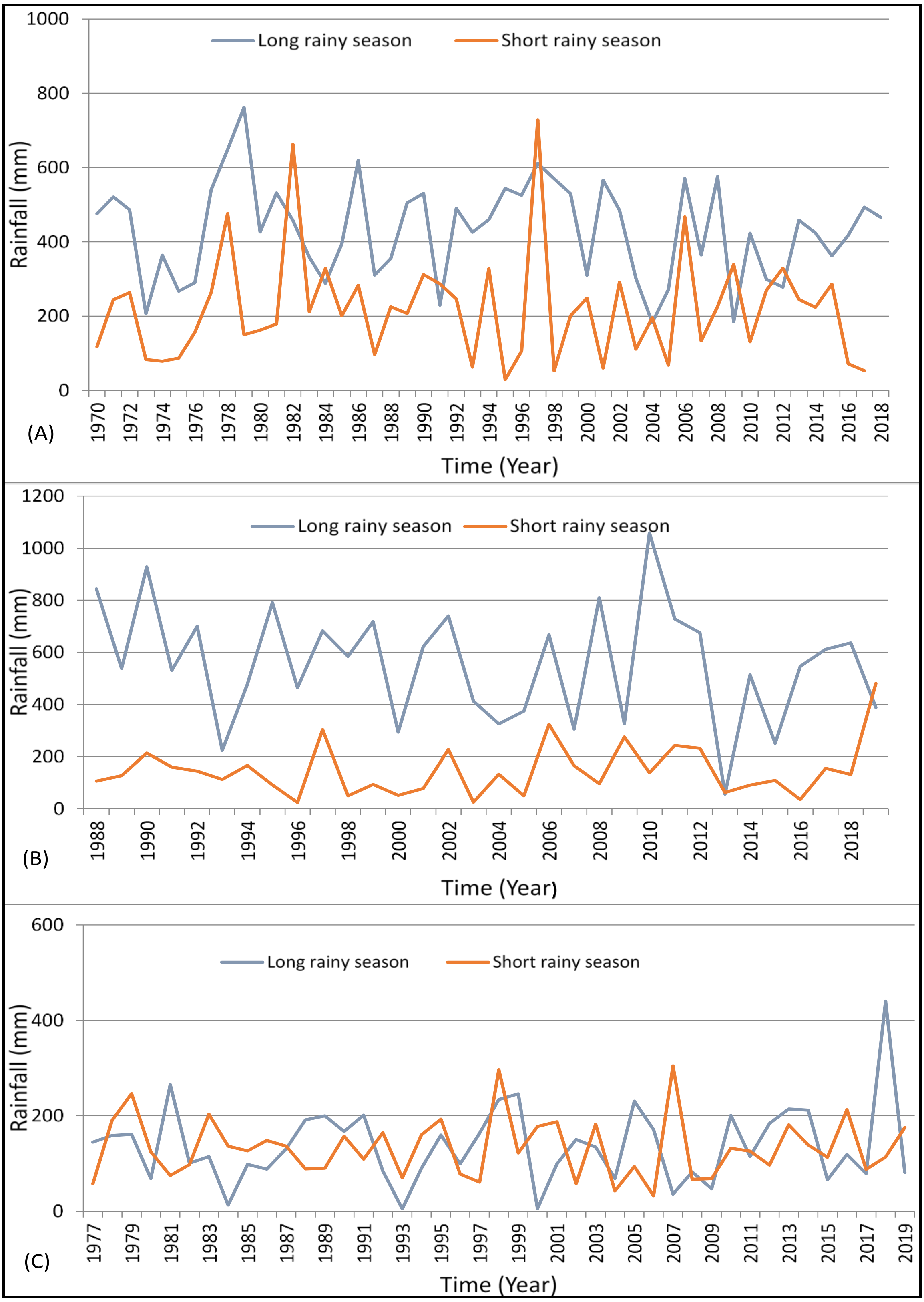

Figure 2. 4: Time series plot of the rainfall during the long rainy season (March-May) and the short rainy season (October-December) at (A) Arusha (B) Moshi (C) Amboseli National Park. Source: (Agricultural Seed Agency (ASA), 2018; Altmann and Alberts, 2020).

Figure 2.5 shows from 2001, the annual rainfall and the long rain during the wet season (March-May) at Rongai station (Figure 2.1) in the northern leeward-forested slopes of KINAPA at an altitude of almost 2550 m a.s.l. Both the annual and the wet season rainfall from 2001 respectively showed apparently increasing long-term trend that was however not statistically significant (R^2=0.02, β=9.76 mm/year, p>0.05, t=0.53, n=18, and R^2=0.08, β=14.21 mm/year, p>0.05, t=1.14, n=18). The rainfall at Rongai represents catchment rainfall for the rivers that drain the northern (leeward) slopes of Mt. Kilimanjaro and flow downstream into the semi-arid areas to the north of Mount Kilimanjaro. The mean annual rainfall at Rongai was 1370 mm (SD=±386)

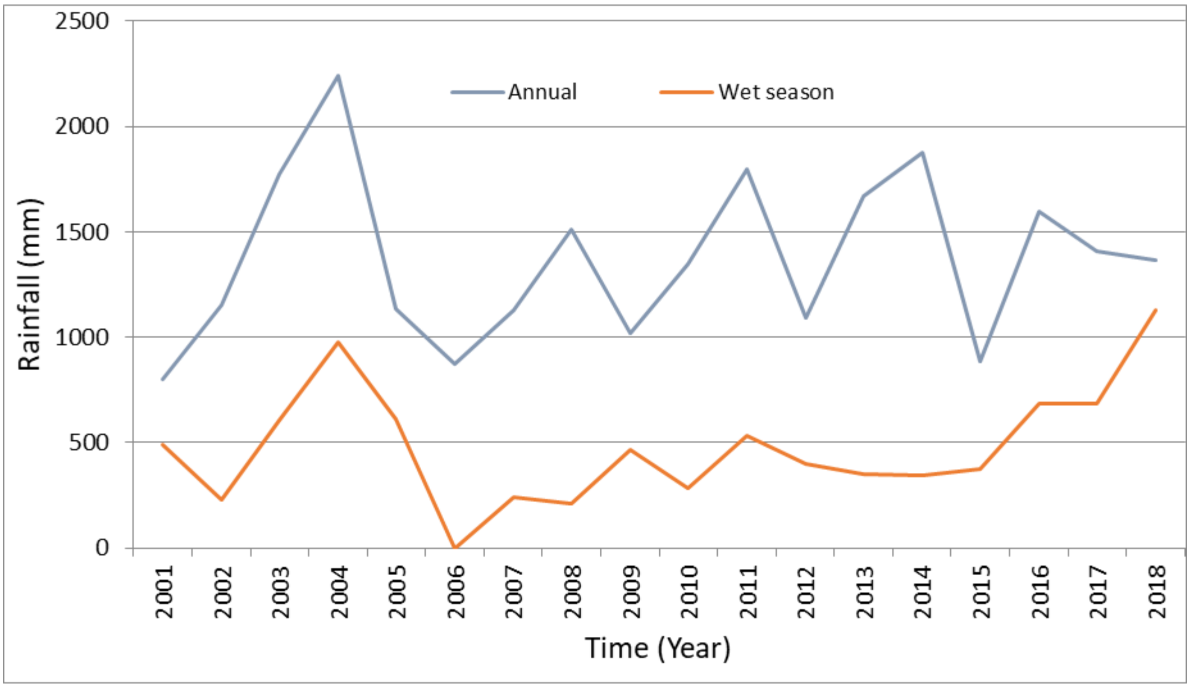

Figure 2. 5: Time series plots of the annual rainfall and the wet season rainfall at Rongai station on the northern slopes of Mt. Kilimanjaro in KINAPA. Source: Kilimanjaro National Park (KINAPA,2020).

Figure 2.6A shows the mean maximum and minimum temperature at Moshi airport (on the windward side of Mt. Kilimanjaro). Both maximum and minimum temperature indicate a rising trend that was significant (R^2=0.5, β=0.08 °C/year, p<0.001, t= 5.17, n=31) only for the maximum temperature. The maximum and minimum temperature in Amboseli National Park (on the leeward side) respectively showed an apparent decreasing long-term trends (R^2=0.19, β=-0.04°C/year, p<0.05, t=-2.21, n=23 and R^2=0.09, β=-0.02°C/year, p>0.05, t=-1.48, n=23), which however were not significant (Figure 2.6B).

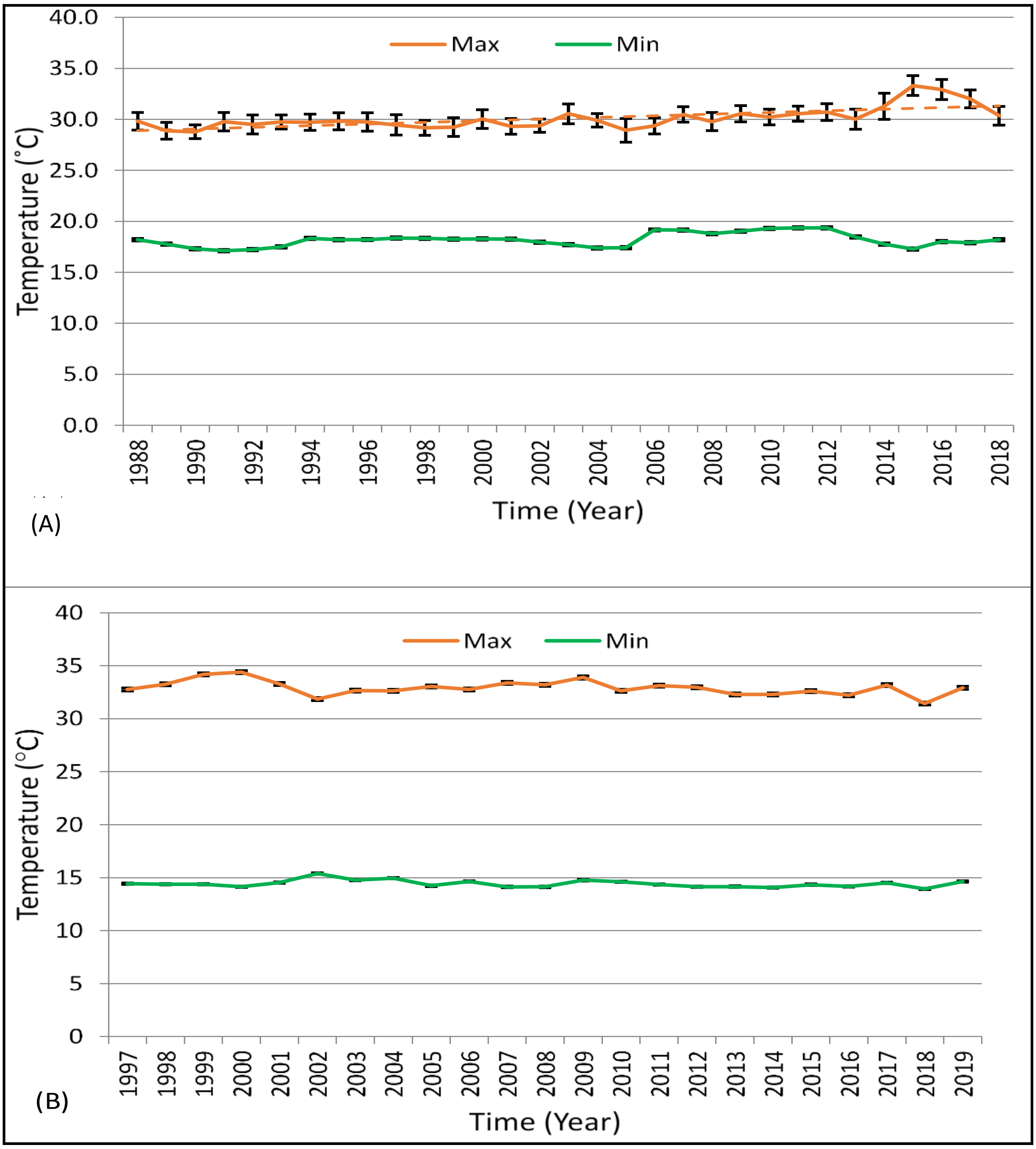

Figure 2. 6: Time series plot of the maximum and minimum air temperature at (A) Moshi and (B) Amboseli National Park. Source: (Tanzania Meteorological Agency (TMA)-Moshi, 2018; Altmann and Alberts, 2020).

The rainfall in the landscape (Figure 2.4) as well as the Southern Oscillation Index (SOI; Figure 2.7) manifest a large inter-annual variability. However, the rainfall variability was not correlated with the Southern Oscillation Index (SOI) at Amboseli station (R^2=0.0004, p>0.05,t=0.13, n=43). There was also no significant relationship between the SOI and the annual rainfall in Moshi (R^2=0.05, p>0.05, t=1.26, n=31), and Arusha (R^2=0.03, p>0.05, t=-

1.05, n=42). There was no significant relationship between the SOI and the water levels of Lake Chala (R^2=0.002, p>0.05, t=-0.39, n=94), Lake Amboseli (R^2=0.012, p>0.05, t=-1.27, n=136), Lake Jipe (R^2=0.004, p>0.05, t=0.19, n=13) or the NyM reservoir (R^2=0.01, P>0.05, t=1.27, n=135).

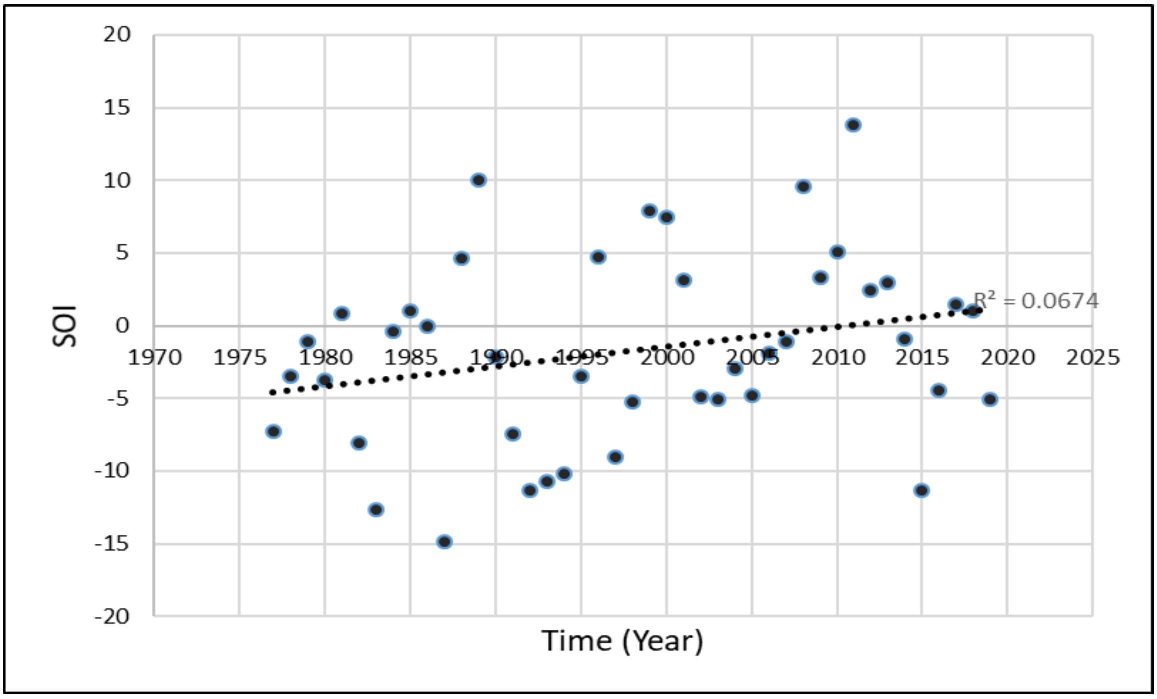

Figure 2. 7: Time series plot of the yearly-averaged Southern Oscillation Index. Source: (Australia Bureau of Meteorology, 2020).

2.3.2 Rivers and streams

Table 2.2 below indicates the mean amount of water available, extracted and that which remained for the environment at various sites in ANAPA and KINAPA. Generally, most of the water extraction sites for domestic use were small in size and with relatively low amount of water but often of good quality (see Chapter 3). Most of the extraction water sources in ANAPA had total available water of less than 2 m^3/min. A few of them had higher values reaching up to 14 m^3/min for instance at the Bangata intake (see Figure 2.2A, site A7). On the other hand, in KINAPA the river discharge ranged between 1 and 1.6 m^3/min upstream of extraction sites and, overall, the amount of water left in the parks after extractions was 15% and 29% in ANAPA and KINAPA respectively (Table 2.2).

Table 2. 2: The mean amount of water available, extracted and that which remained for the downstream environment at the monitoring sites in ANAPA and KINAPA (n=6 to 13) in both the dry and wet seasons.

	ANAPA				
Code	Name	Mean available(m^3/min)	Mean extracted water (m^3/min)	Mean remaining water(m^3/min)	Mean % remaining water
A1	Ngongongare 1	0.394	0.101	0.293	74.37
A2	Mweka	0.554	0.371	0.183	33.08
A3	Ngongongare3	1.136	0.563	0.573	50.42
A4	Kilinga	0.158	0.133	0.025	15.86
A5	Malemeo	0.378	0.362	0.016	4.19
A6	Mwakilenga	0.845	0.431	0.379	48.97
A7	Bangata	14.828	11.834	2.994	20.19
A8	Kikololomu	1.993	1.993	0	0
A9	Kira hill	0.427	0.413	0.016	3.125
A10	Lewate	0.354	0.354	0	0
A11	Maambureni river	0.264	0.261	0.007	1.26
A12	Nading'oro 2	0.749	0.750	0	0
A13	Narok B	1.141	0.555	0.586	51.37
A14	Nasula	1.222	1.222	0	0
A15	Nshupu	0.012	0.012	0	0
A16	Olmotony intake	2.770	2.235	0.535	19.32
A17	Sajona	0.407	0.407	0	0
A18	Sambasha	0.096	0.096	0	0
A19	Masaga	4.898	4.898	0	0
A20	Nading'oro1	0.148	0.148	0	0
A21	Seela	0.120	0.120	0	0
	KINAPA				
K1	Kamwanga	1.513	1.234	0.279	18.5
K2	Kitendeni	0.817	0.513	0.304	37.19
K3	Lerangwa	0.789	0.614	0.175	22.13
K4	Londorosi	1.708	1.039	0.669	39.19

The current level of water extraction in ANAPA takes between 50% and 100% of the available water at the extraction sites in the dry season (Figure 2.8). Most of these extraction sites were at streams and rivers ultimately supplying the Kikuletwa River. Likewise, the assessed intakes in KINAPA extract between 60% and 85% of all water available in the dry season. For instance, the Kamwanga and Lerangwa intakes respectively took 85% and 81% of the available water. Averaged across the sites, the existing intakes in

ANAPA and KINAPA respectively abstracted almost 90% and 70% of the available water in the dry season. The majority (~70%) of the existing water intakes in ANAPA abstracted nearly all (>90%) of the available water in the dry season.

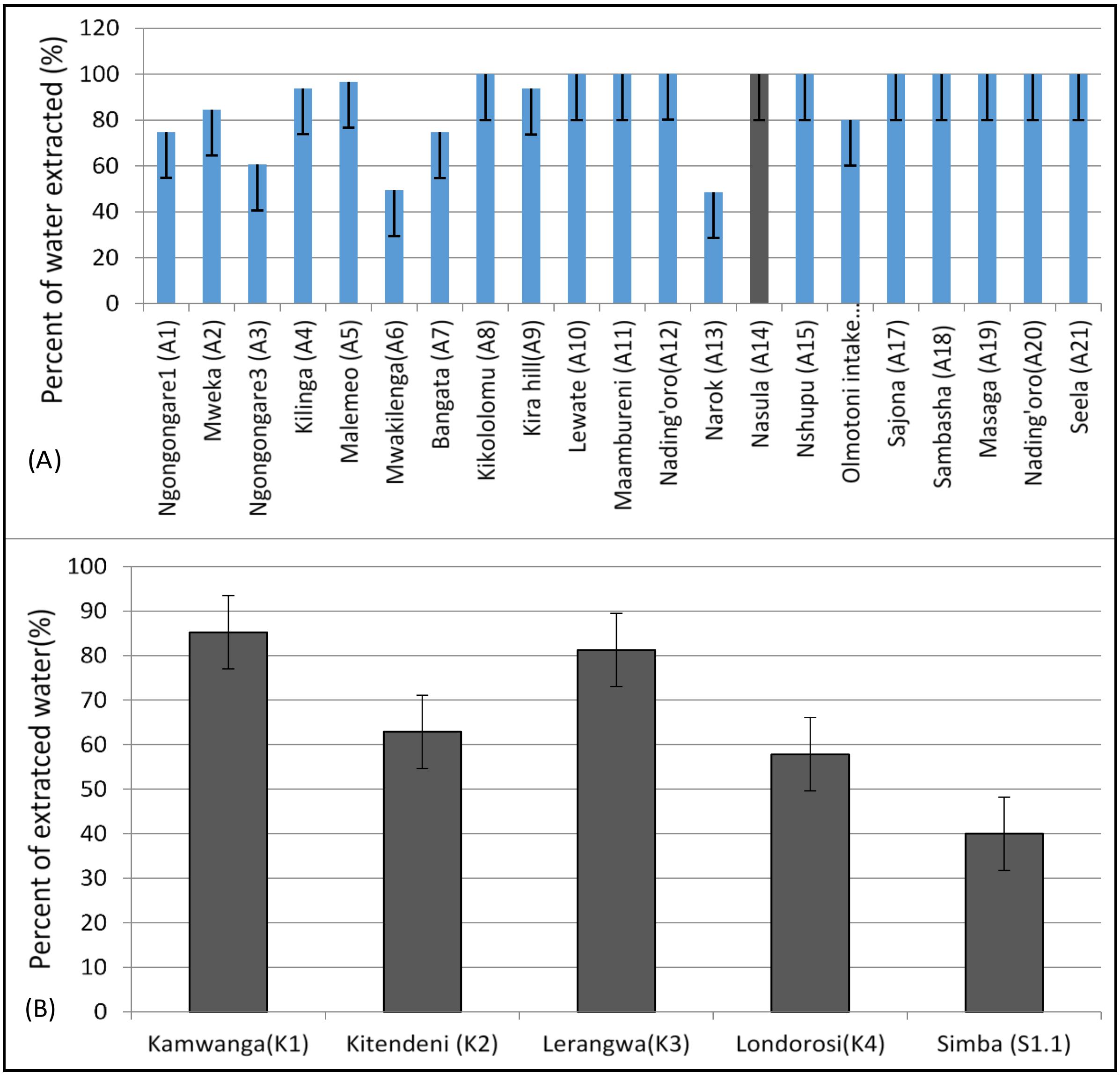

Figure 2. 8: (A) Mean percentage water extracted during the dry season within (A) ANAPA (±SE, n=4), and (B) KINAPA (±SE, n=10). Blue columns represent tributaries on the windward side (i.e. the Kikuletwa River) and the black columns represent tributaries on the leeward side (including the Ngarenanyuki and Simba Rivers).

The inter-annual variability of river flow and water extraction is evident. Indeed, the proportion of available water and that which remained in the streams downstream in the wildlife-rich areas during the dry season (Sept- March) was significantly higher ($p<0.01$, $F=5$, $n=6$) during this study (2018/2019) than in 2012/2013 as revealed by ANOVA-two factor with replication (Elisa et al., 2016; Figure 2.9). This was clearly evident as even some of the streams that recorded no water in 2012/2013 such as the Ngongongare 1 had a significant amount of water downstream of extraction site with an average dry season flow of about 0.12 m^3/min during this study.

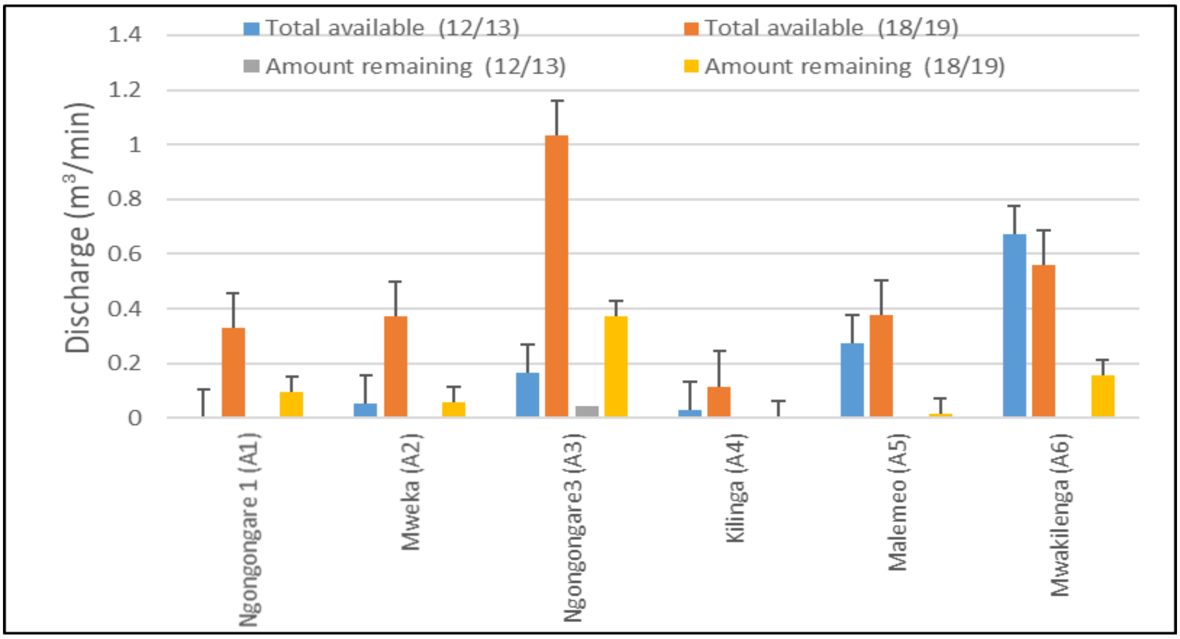

Figure 2. 9: Comparison of the mean total amount of water available upstream of extraction sites and amount of water left (amount remaining) for the downstream environment after extraction, between 2012/2013 (±SE, n=5) and 2018/2019 (±SE, n=6) for the wildlife-rich sites in ANAPA.

Figure 2.10 shows the extent of water extraction in the wildlife-rich areas (areas with high wild animal density and where the animals are frequently sighted according to park rangers) within ANAPA and KINAPA. While considerably greater amount of water was abstracted from these sites, on average none of them completely extracted all of the available water. Comparatively, extraction in wildlife rich areas in ANAPA took proportionally a larger amount of the available water (77%) than in KINAPA (72%). However, in terms of the actual average amount of water extracted in the wildlife-rich areas, ANAPA was subject to less extraction (0.32 m^3/min) than in KINAPA (0.83 m^3/min). At Mweka (A2), Kilinga (A4) and Malemeo (A5) in ANAPA, the water extraction was the largest (above 80% of the available water) and consequently they had the lowest (less than 0.1 m^3/min) amount of water remaining in the downstream environment. Similarly, at Lerangwa(K3) and Kamwanga(K1) in KINAPA the lowest amount of water (about 0.2 m^3/min) was left for the environment.

Overall, there was a significant difference (p<0.01, t=3.14, n=10) between discharge in up- and- downstream in both dry and wet season in 10 wildlife rich sites of ANAPA and KINAPA. However, such difference was not observed for Ngongongare 1 (A1) in ANAPA and Kitendeni (K2) in KINAPA, which had a relatively large downstream flow.

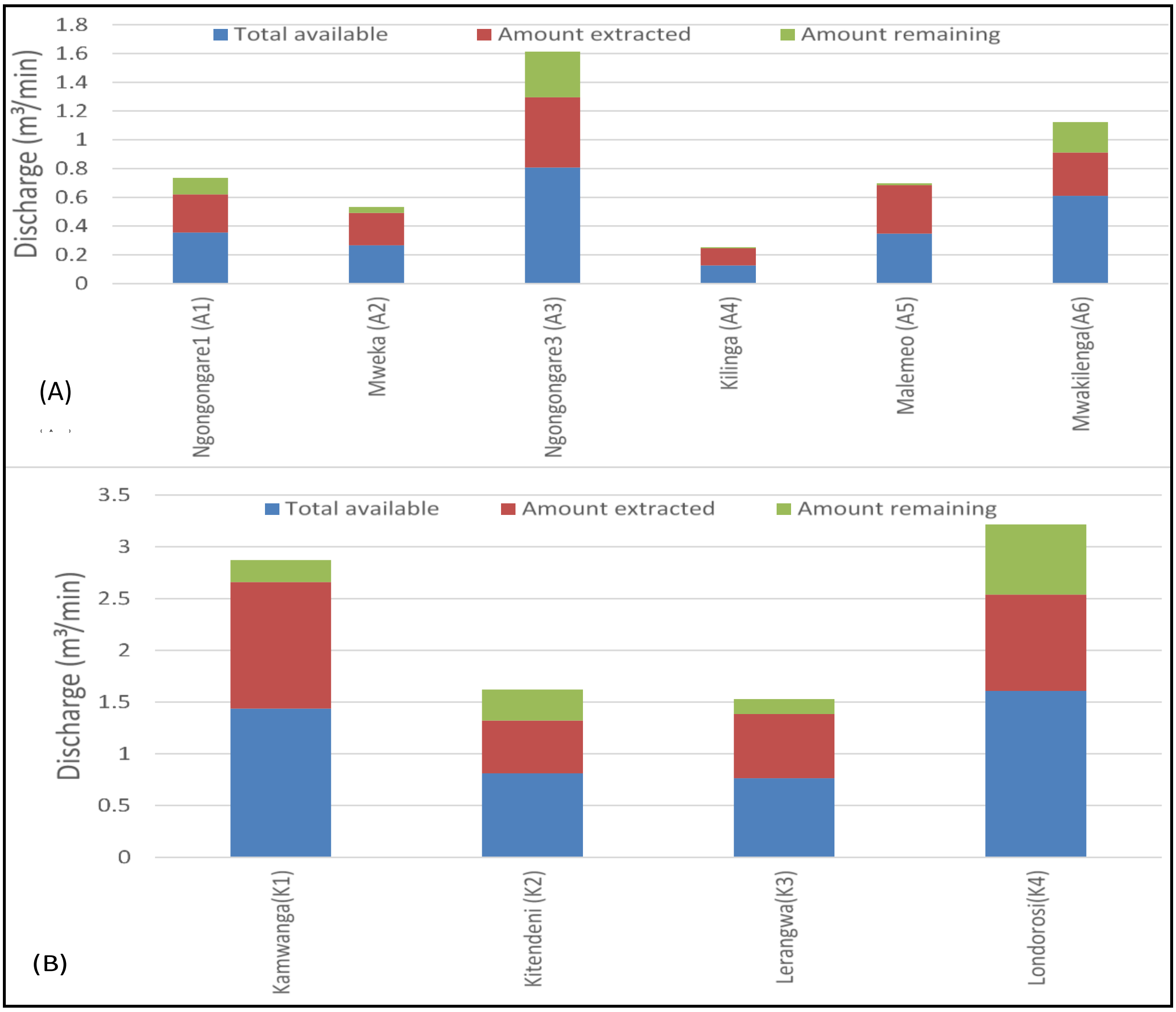

Figure 2. 10: The mean discharge (A) ANAPA (±SE, n=4), and (B) KINAPA (±SE, n=10) of total amount of water available upstream of the extraction sites, the amount of water extracted, and the amount of water left for the downstream environment during the dry season in the wildlife rich areas in ANAPA and KINAPA.

The assessment of the available water at the non-extraction streams (Figure 2.11), that may likely be extracted in the future, confirmed that most of the streams had a water discharge less than 2m^3/min per stream in the dry season. While the number of unextracted sources is unknown, the discharges shown in Figure 2.11 are likely to be typical. However, most of these un-extracted sites were either of poor quality (fluorine and saline rich) or located in the remote areas of the parks.

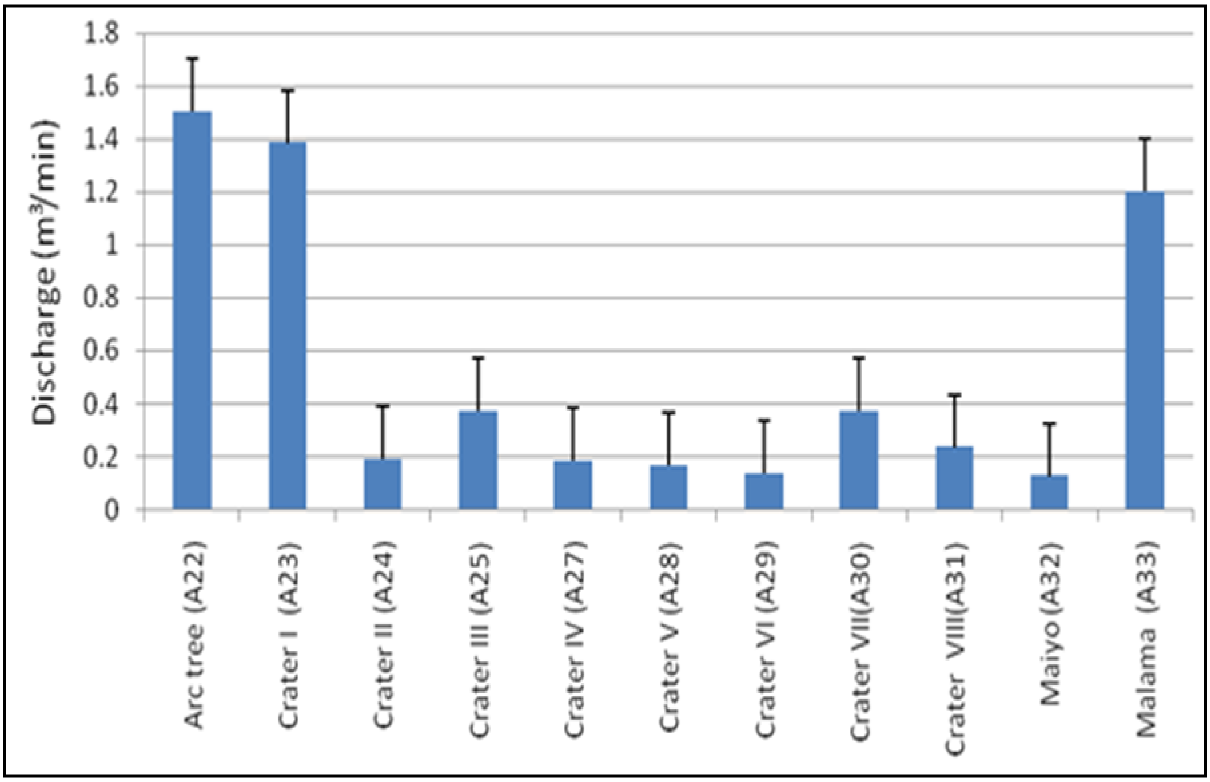

Figure 2. 11: The mean discharge (±SE, n=3) at some of un-extracted water sources, which drain to the Ngarenanyuki River in ANAPA during the dry season.

The discharge in the upstream sites without extraction on both the Ngarenanyuki and Simba Rivers varied over time by up to 55% (Figure 2.12). As expected, the discharge somewhat followed the rainfall patterns in the parks (see Figure 2.3), as it peaked in the rainy periods in May-2019 and from October-2019 through to February 2020. The discharge declined in the period of low rainfall in September-2018, Jan-March-2019 and July-September-2019. In all the periods the Ngarenanyuki River recorded on average 39% higher discharge than that of the Simba River and this difference was significant($p<0.01$, $t=9.6$, $n=17$). The highest recorded discharges were around 100 m^3/min and 60 m^3/min in February 2020 in the Ngarenanyuki and Simba Rivers respectively, while the lowest were around 60 m^3/min in September-2018, and 35 m^3/min in September-2019 respectively in the Ngarenanyuki and Simba Rivers.

As both rivers flow downstream of extraction points, the discharge varied markedly in space and time. An analysis of variance test shows a significant difference ($p<0.01$, $F = 3.2$, $n=16$) between the upstream and downstream discharges. For instance, there was less than 1 m^3/min discharge at Ngabobo and Ndarakwai (see Figure 2.1, sites N2 & S4; both located about 20 km downstream of the extraction sites) in April and August-2019, as compared to above 70 m^3/min and 35 m^3/min respectively in the upstream of the extraction sites in the Ngarenanyuki and Simba Rivers.

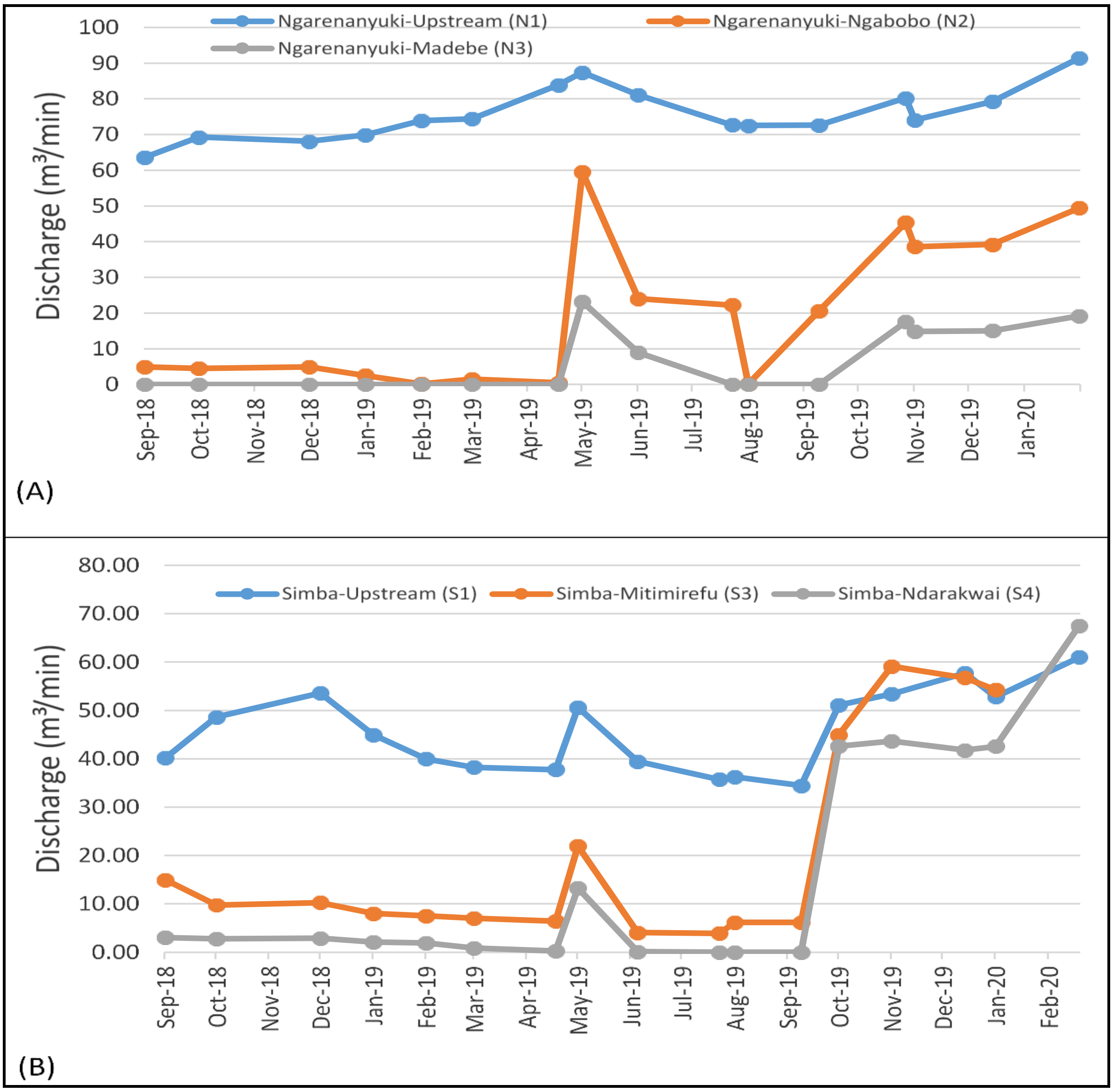

Figure 2. 12: Time series plot of discharge in the river upstream within the parks (N1, S1), mid (N2, S2) and low sections located in the village lands(N3, S3) in (A) Ngarenanyuki River and (B) Simba rivers. Water abstraction took place between N1 and N3 on the Ngarenanyuki River, and S1 and S3 on the Simba River.

Figure 2.13 below shows the percentage of the upstream discharge measured immediately downstream of the Ngabobo extraction canal, and at Ndarakwai wildlife ranch located about 1 km downstream of the Mitimirefu extraction canal (see Figure 2.1, sites N2 & S4). The discharge at these downstream locations showed a marked temporal variation and on average was less than 20% of the upstream water during the dry season. In contrast, during the wet season, at the same locations the discharge was about 60% and 90% that at the upstream sites.

Generally, there were low discharges from September 2018 to September 2019 except in May-2019, which recorded high discharge values similarly to those values during the period from October 2019 to February 2020. On average, less than 30 % of all water flowing at the upstream sites (before extraction) in both dry and wet seasons, remained following removal by the extraction canals between sites N2 and N3, and S4 and S6 (see Figure 2.1) in the Ngarenanyuki and Simba Rivers respectively. Further, the dry season discharges for both the Simba and the Ngarenanyuki Rivers were significantly different ($p<0.001$, $t=17.8$, $n=9$) between the discharges upstream of the abstraction points and at the sites located 20 km during the dry season. However, there was no such difference over the wet season (Figure 2.13).

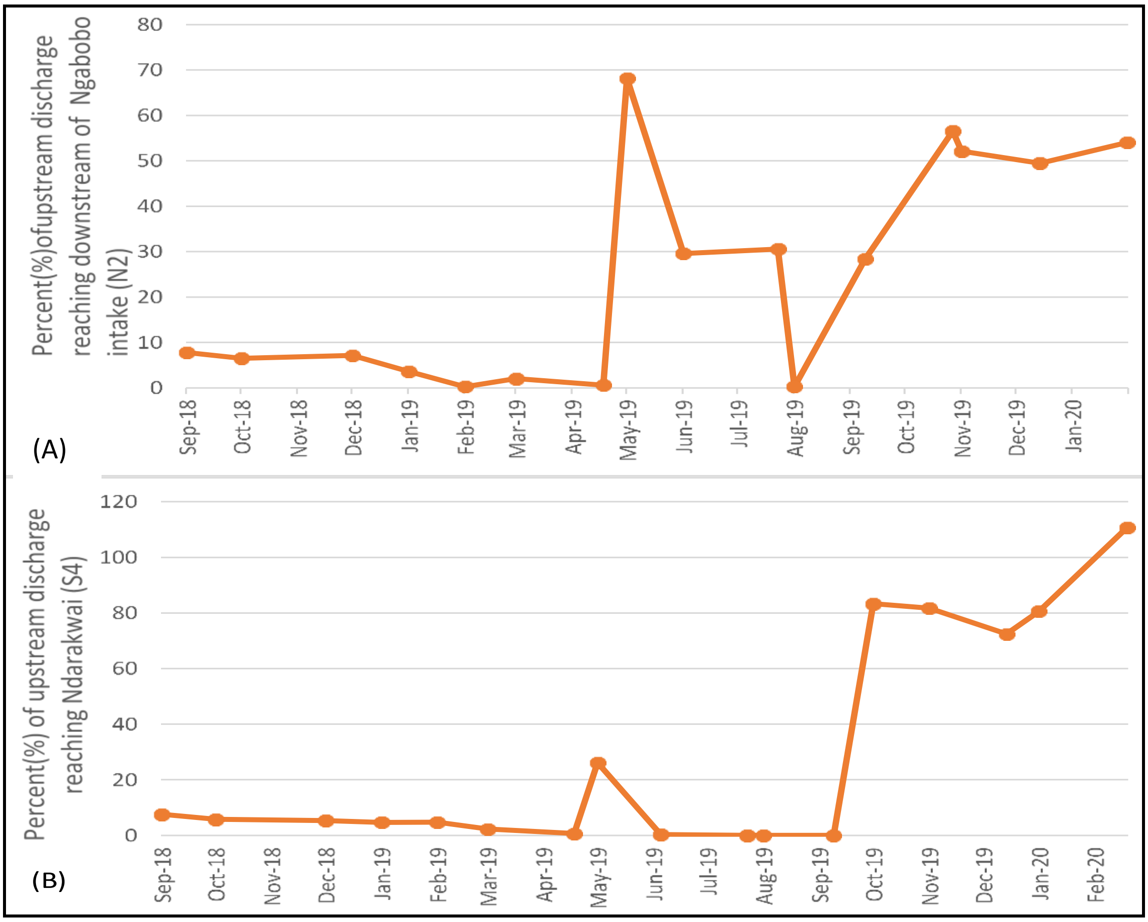

Figure 2. 13: Time series plot of the percentage of the monthly upstream river discharge that reaches immediately downstream of (A) Ngabobo (Site N2) on Ngarenanyuki River, and (B) Ndarakwai wildlife ranch (Site S4) on Simba River, September 2018 to February 2020.

Over 50% of the total upstream water was extracted from the rivers above the Ngabobo and Mitimirefu extraction canals in, respectively, the Ngarenanyuki and Simba Rivers (see Figure 2.1, sites N2 and S3). The canals then extract about 70% to 80% of this remaining water in the dry season, further reducing the amount of water available to the downstream areas, particularly during the daytime, when most irrigation takes place. Following a similar calculation as that of Elisa et al.(2010,2021), it was estimated that, irrigation farming consumed about 0.75m^3/min per 1km^2 and 1.23m^3/min per km^2 in the Ngarenanyuki and Simba Rivers respectively during the dry season. The total irrigated area for the Ngarenanyuki River was larger (90km^2) than in the Simba River (25km^2). The amount of rainwater retained by these farms and not returned to the river during the wet season was estimated at 0.67 m^3/s and 0.19 m^3/s, respectively, for the Ngarenanyuki and Simba Rivers.

The river discharges for both the Ngarenanyuki and Simba Rivers, thus declined substantially downstream, mainly due to excessive extraction of water for irrigation farming. This was evidenced by the significantly small amount of water (less than 1 m^3/min) that was allowed to flow downstream (about 20 km from the ANAPA and KINAPA after several extractions that occurred within the first 20 km from the Park's boundary. In addition, this small remaining amount of water did not reach to the community wildlife areas situated about 30 km further downstream due to other extractions, evaporation loss, and infiltration.

2.3.3 Lakes and water holes

Several other surface water bodies that include man-made water sources and fresh-water lakes were monitored (Figure 2.14). They are not only important sources of water for humans, livestock and wildlife, but also serve as an important indicator of changes in surface water availability.

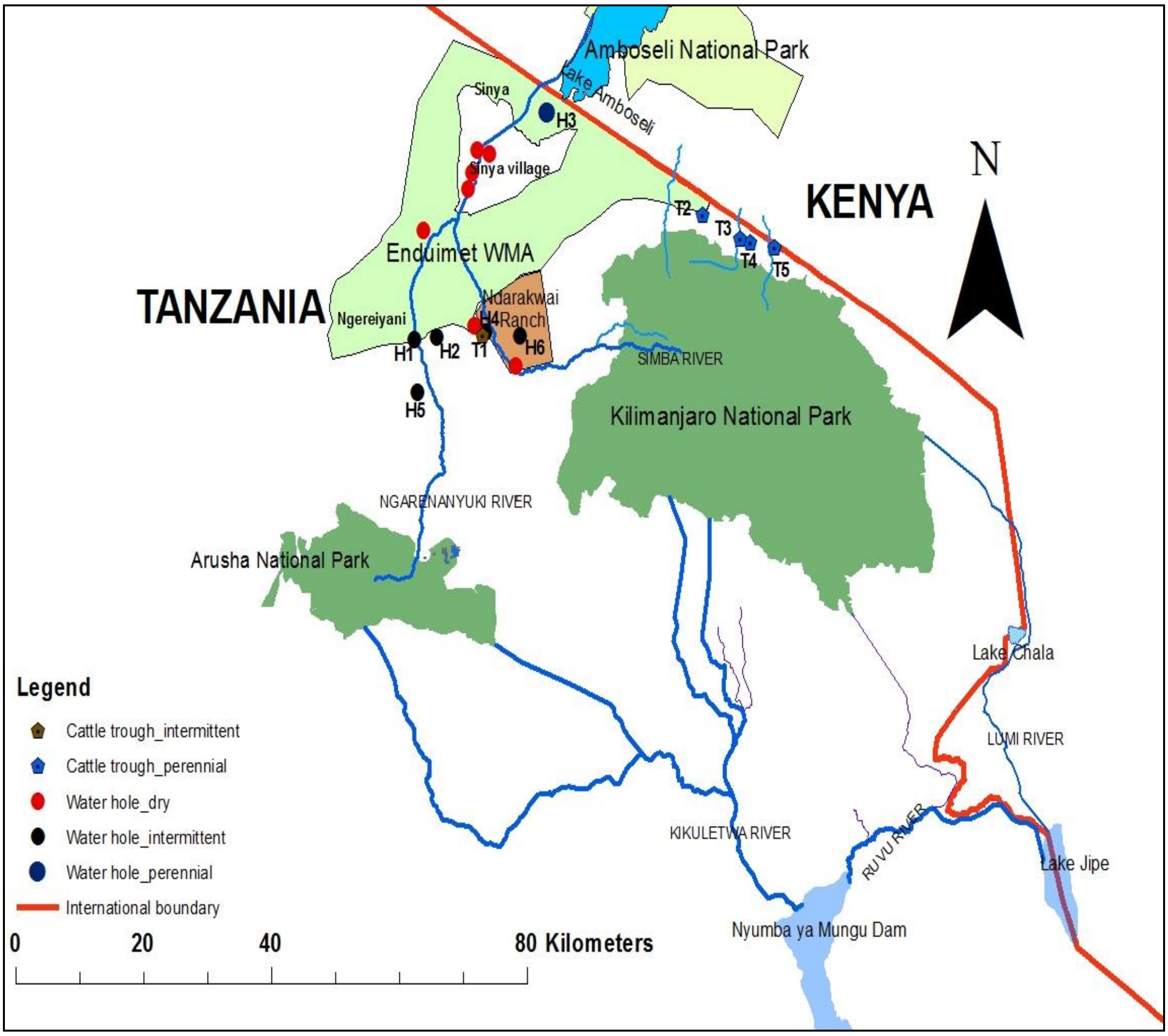

Figure 2. 14: Sketch map showing the natural and the man-made surface water sources in the Kilimanjaro landscape.

Lakes

Figure 2.15 shows a time series plot of the water levels of Lakes Amboseli obtained from satellite altimery, and that of Lake Chala obtained by combining the manual readings data and water level logger data. The lake levels for the two lakes experienced seasonal and inter-annual variability similar to that of rainfall. The time-series data do not show a significant long-term trend in either rainfall or water level for the lakes, compared to the variability ($R^2 < 0.13$). Both Lakes Chala and Amboseli showed slightly positive trends (R^2= 0.036, β= 0.00004m/month, p>0.05, t=1.84, n=94; and R^2=0.12, β=0.0008 m/month, p<0.001, t=4.37, n=141 respectively).

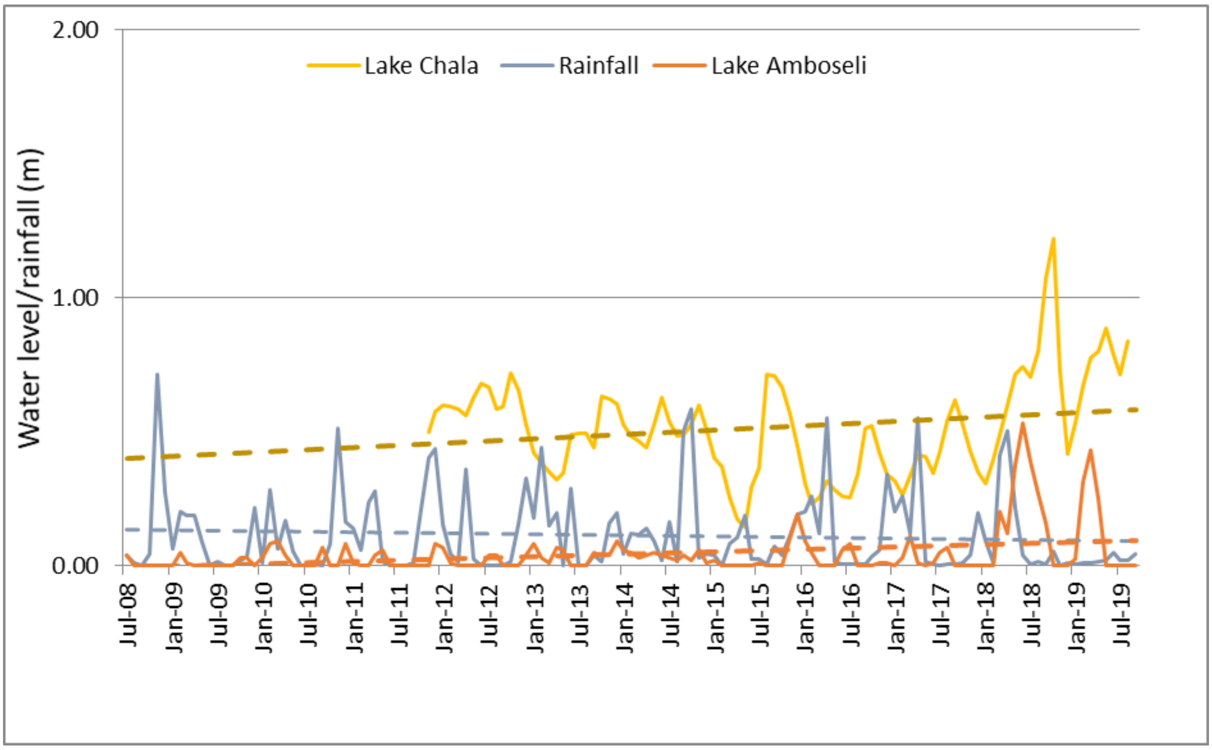

Figure 2. 15: Time series plots of the monthly rainfall at Moshi station and the water level of Lake Amboseli, July 2008 to July 2019 and Lake Chala, July 2011 to July 2019. Source: (Pangani Basin Water Office (PBWO), 2020;Tanzania Meteorological Agency (TMA)-Moshi, 2020; USDA, 2020).

Figure 2.16 shows how the groundwater inflow of Lake Amboseli water level varied with time. The ground water inflow was derived from the mass balance equation (Eq. (1) for the lake. The water budget of Lake Amboseli depends on groundwater seepage from Mt. Kilimanjaro, local rainfall and evaporation. The Ngarenanyuki and Simba Rivers once flowed into the Amboseli basin, but no longer do so (Istituto Oikos, 2011) due to over-extraction of water upstream plus the siltation of the river channel effectively blocking the canal. In turn, this leads to local flooding in those areas during the wet season. Thus, the river water inflow was not considered in computing the Lake Amboseli water budget. In addition, it was assumed that there is no groundwater outflow from the lake as there is no evidence for it. The lake receives an average of about 0.19 m^3/s. Over the whole period of data, significant amount (55%) of the water in Lake Amboseli came from groundwater. This was also confirmed by the Chi-square test results shown in Table 2.3 below. The exception was the heavy local rains (507mm) in 2018 that raised the lake level by almost 0.6 m. The variation in lake water level was in synchrony with the groundwater inflow as indicated by the strong correlation between these variables (Table 2.3; R^2=0.57, p<0.05, t=13.48, n=141).

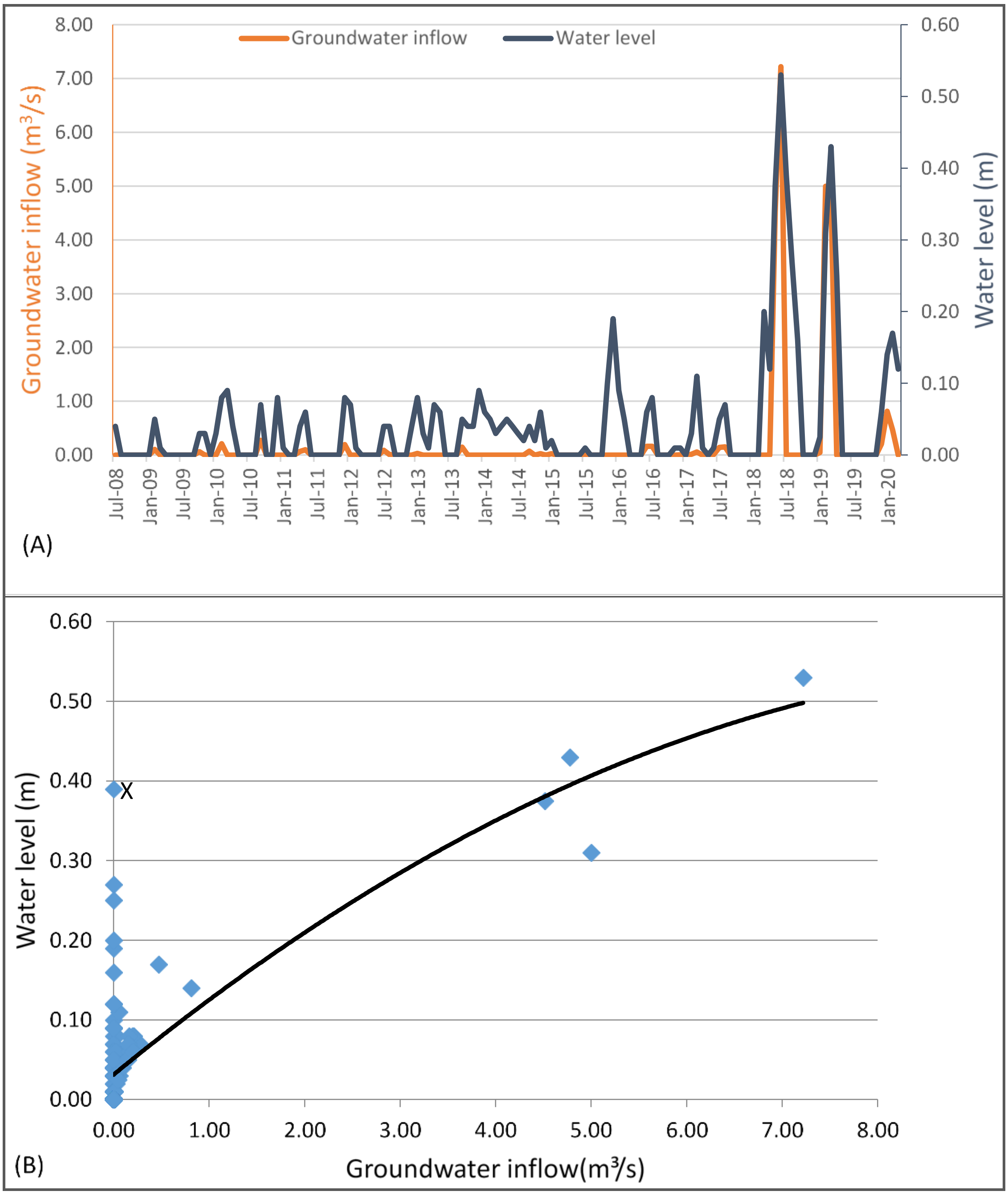

Figure 2. 16: (A) Time series plot of the monthly-averaged groundwater inflow calculated from Equation (1) and the water level of Lake Amboseli; (B) the suggested non-linear relationship between the groundwater inflow and the water level in Lake Amboseli.

The high scatter at small values of the groundwater discharge in Figure 2.16 B is logical because if the lake is at a high level and then a drought occurs, then the groundwater inflow should be zero, as is demonstrated by point X. The high values of the water level and the groundwater inflow occurred during high rainfall resulting in a rising lake water level.

Table 2. 3: Generalised linear model (glm) Analysis of Deviance (Type III Wald chi-square tests) for predictors of Lake Amboseli water level.

	Estimate	Std. Error	LR Chisq	Df	Pr(>Chisq)
Ground inflow	0.4644	0.1623	5.0781	1	0.02423
Rainfall Rongai	-0.2966	3.3196	0.0082	1	0.92806
Rainfall Amboseli	6.7771	8.5164	0.5306	1	0.46635

The NyM is a man-made reservoir and receives water from two main rivers, namely the Kikuletwa River from KINAPA and ANAPA windward side and the Ruvu River, which is the outflow from Lake Jipe and an ungauged small streams on the southeast slopes of KINAPA. The outflow from the NyM reservoir is controlled by a dam, and discharges to the Pangani River. The Kikuletwa River is the main contributor of water to the reservoir, accounting for an average of about 21.35 m^3/s, whereas the Ruvu River accounted for about 13.57 m^3/s. The water level of the NyM reservoir varied in synchrony with river inflow and rainfall as shown in Figure 2.17 and Table 2.4. The trend was slightly negative (R^2=0.006, β=-0.0001 m/month, p>0.05, t=-0.90, n=135) for NyM as the lake level declined especially from 2008 to end of 2012. The peak of the lake level appeared to follow shortly after a peak in rainfall and river inflow, however there were few cases with slight mismatches such as those in between 2013 and 2014. The minimum lake levels of less than 1 m, occurred in 2012, 2013 and 2016. The groundwater inflow to the reservoir was estimated at 0.103 m^3/s (Murashani, 2012), and the contribution from rainfall directly over the reservoir was 2.71 m^3/s. The outflow to the Pangani River has an average of 34.7 m^3/s. So, the total river inflow accounts for more than 90% of all water flowing into the NyM.

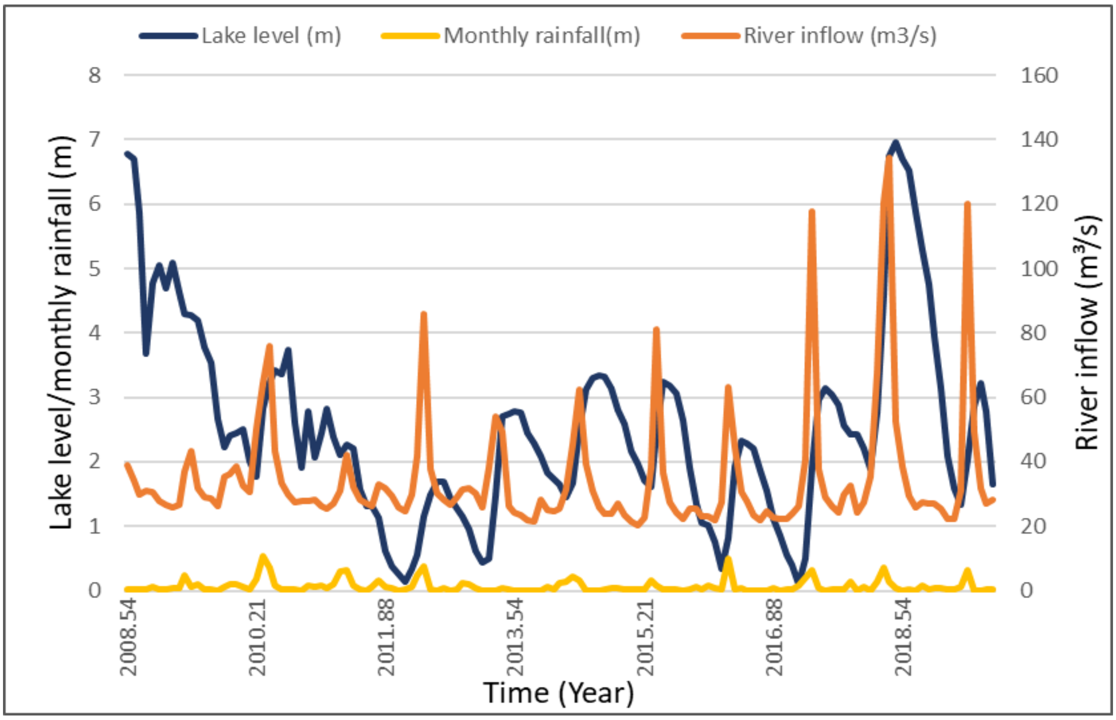

Figure 2. 17: Time series plots of water level, rainfall and river inflow in the Nyumba ya Mungu (NyM) reservoir.

Table 2. 4: Generalised linear model (glm) Analysis of Deviance (Type III Wald chisquare tests) for predictors of lake water level in NyM reservoir.

	Estimate	Std. Error	LR Chisq	Df	Pr(>Chisq)
Rainfall Moshi	-1.694276	0.855223	4.2057	1	0.040288
Rainfall Rongai	-0.59658	0.423811	2.0609	1	0.151124
River inflow	0.010261	0.003148	9.3318	1	0.002252
River outlflow	-0.001264	0.002958	0.1839	1	0.668067

The water level of Lake Jipe (Figure 2.18) generally varied in synchrony with the groundwater inflow rather than rainfall. However, the lake level did not show a strong correlation with either the groundwater inflow (R^2=0.26, p>0.05, t=1.89, n=12) or the rainfall (R^2=0.08, p>0.05, t=-0.96, n=12) for the short duration of the data (1 year). The water level of Lake Jipe varied the least (0.2 m) of all the study lakes. Accordingly, the Ruvu River drains an average of about 9.78 m3/s from the lake. The lake receives surface water mainly from the Lumi River, which drains the northern slopes of Mt. Kilimanjaro, but its contribution was relatively small, estimated at an average of 1.84 m^3/s. Rainfall over the lake contributed an average of 0.57 m^3/s. However, much of the water (78%) in Lake Jipe as calculated from Equation (1) comes from groundwater inflow with an average of 8.61m^3/s draining from Mt. Kilimanjaro. In computing the average groundwater flow, the following variables were taken into account in Equation (1): the temporal change of Lake Jipe storage, the Lumi River inflow, the rainfall over the lake, the water loss by evaporation, and the Ruvu River outflow.

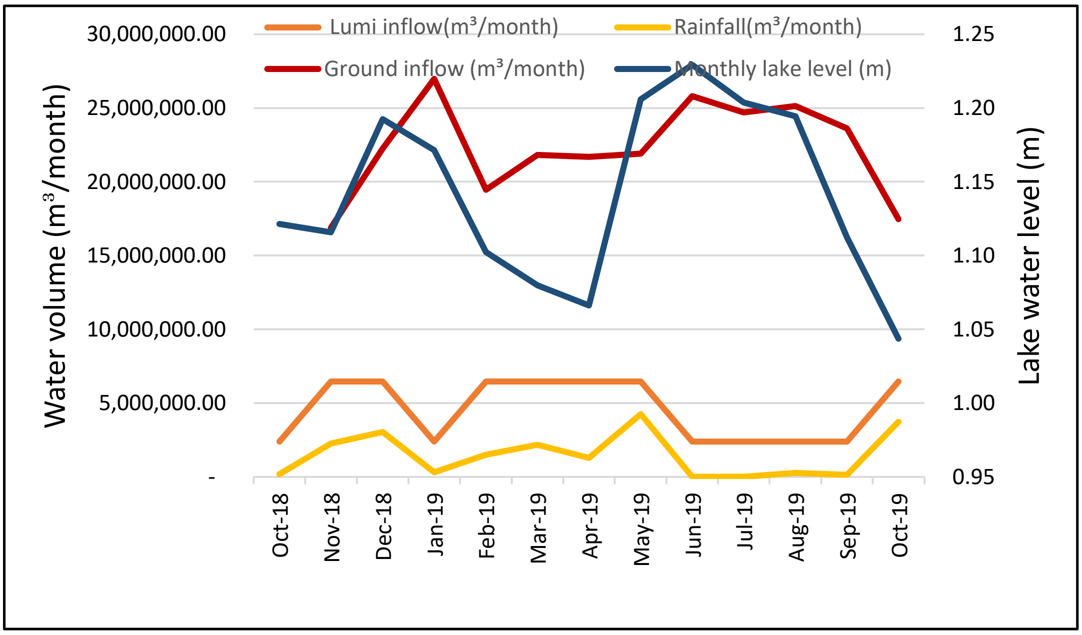

Figure 2. 18: Time series plots of the groundwater inflow, the river inflow, the lake water level and the monthly rainfall in Lake Jipe. Rainfall data source: (TSWENAPA, 2019).

The hydrology data for Lake Chala are shown in Table 2.5 and Figure 2.19. Lake Chala is a crater lake with no river inflow and no river outflow. The lake level fluctuated in time and these fluctuations can only be due to fluctuations in the groundwater inflow, the groundwater outflow and the rainfall. Rainfall data are available from the Rongai station (location: Figure 2.1). However, there were few mismatches. While the lake water level responded positively to an increase in rainfall and groundwater inflow, the lake water level showed a weak increasing inter-annual trend (Table 2.5) and it varied by up to 0.8 m (Figure 2.19). The water level showed an inter-annual and seasonal variation. The lake, which is relatively unexposed to human disturbance, is mainly recharged from groundwater with an average net groundwater discharge, which was calculated from Equation (1) to be about 0.18 m^3/s based on the data from 2011 to 2019. An earlier study by Payne (1970) suggested a net groundwater inflow of 0.138 m^3/s. These two estimates match well with each other, adding confidence to the results.

Table 2. 5: Generalised linear model Analysis of Deviance (Type II Wald chisquare tests) for predictors of water level in Lake Chala.

	Estimate	Std. Error	LR Chisq	Df	Pr(>Chisq)
Groundwater outflow	-1.48E-06	1.79E-06	0.66071	1	0.4163
Groundwater inflow	4.29E-07	4.26E-07	0.9601	1	0.3272
Rainfall	1.91E+00	3.19E+00	0.34579	1	0.5565

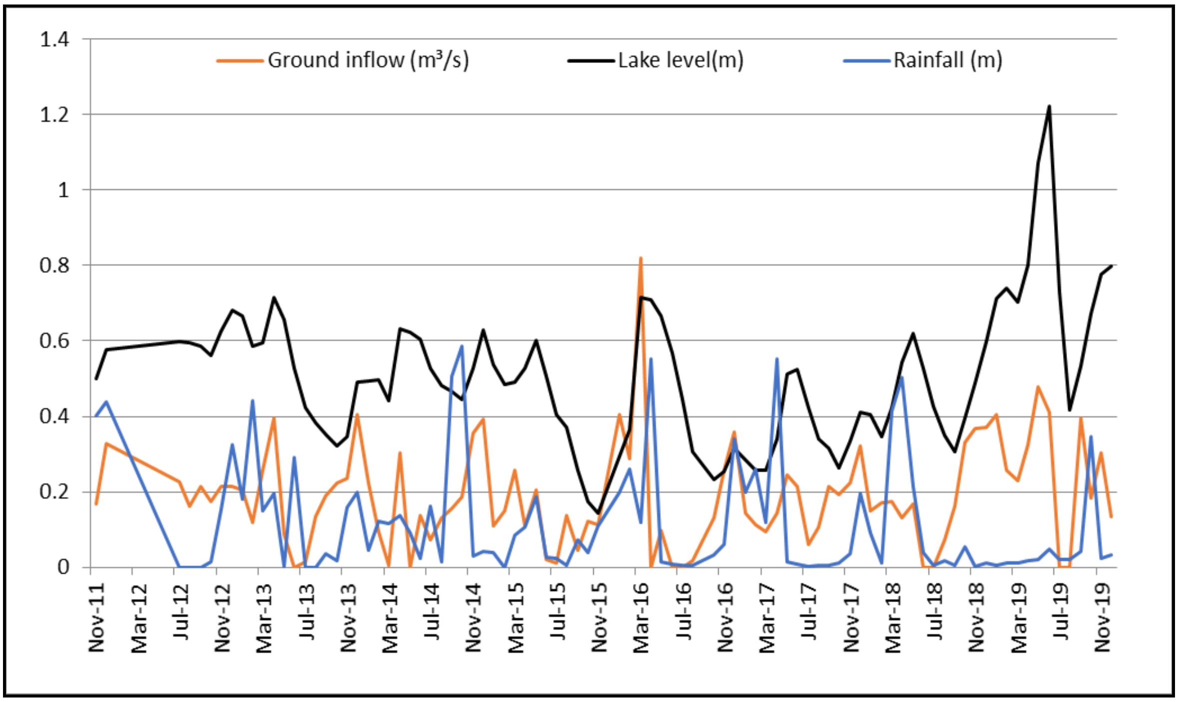

Figure 2. 19: Time series plot for Lake Chala water level, the groundwater inflow and the rainfall at Rongai station in KINAPA between November 2011 and November 2019.

Water holes

Several water holes and cattle troughs have been established in the semi-arid lands, primarily for watering livestock in the dry lands and also for wildlife in areas north of Mt. Kilimanjaro and Mt. Meru (Figure 2.14). These man-made water sources were utilised by wild animals, often at night (particularly those located near human settlements) during which there is minimal or no interferences by livestock or humans. There were at least 13 man-made (excavated) water holes in the dry wildlife areas, and of these, the Sinya water hole (Site H3 in Figure 2.14) near Sinya village contained water (varying from almost 2,000 m^3 in the dry season to 18,000 m^3 in the wet season) throughout the period of the study after being naturally filled during the wet season. Being located within the Enduimet wildlife management area (EWMA; Figure 2.14), the Sinya water holes were generally accessible to wild animals all the time, notwithstanding interruption from livestock during the day. Five of the remaining water holes contained water only intermittently during the dry season as they were supplied by canals that depended on the occasional release by the upstream villages of water from the Simba and Ngarenanyuki Rivers by the upstream villages by means of canals. Two of these water holes were located in Ngereiyani village (see Figure 2.14, Sites H1 and H2) and one in the Madebe area (Site H5), which receive water from the Ngarenanyuki River. The others were located in the Tingatinga village and the Ndarakwai wildlife ranch (sites H4 and H6) respectively and receive water from the Simba River. Basing only on the time when water was available and channelled from the adjacent rivers, the amount of water in these river-fed water holes ranged from almost 100 m^3 during the dry season to almost 2,000 m^3 in the wet season (Table 2.6). The seven remaining water holes (the red dots in Figure 2.14), which depended on direct rainfall and local surface run-off water were dry during most of the dry season. The location of the dry water holes are shown in Figure 2.14 and most of them were located close to the river in the downstream areas, where river channels were heavily sedimented and thus could not retain all their water in the channel during a high discharge. This led to flooding into the adjacent areas including into the waterholes (not fed by canals) during the periods of heavy rain. In addition to the water holes, there were at least five concrete water troughs (sites TI to T5 in Figure 2.14) for watering livestock that were located in villages including Tingatinga, Kitendeni and Irkaswaa all of which bordered the wildlife management area, and two (Kitendeni and Irkaswaa) were in the Kitendeni wildlife corridor. Most of these troughs were established close to the village centre/settlements and depended on the recharge through (flexible or rigid) plastic pipes from extraction points in KINAPA as described above. The troughs contained water in most of the time (>75% of the year) even though in small quantity.

Table 2. 6: Mean volume of water in the man-made water holes in the dry and wet season in the semi-arid West Kilimanjaro area.

Name of water hole	**Source of water**	**Location**	**Dry season volume (m^3)**	**Wet season volume (m^3)**
Ngereiyani-madukani (H1)	Ngarenanyuki River	Ngereiyani village	257	1050
Ngereiyani-Mbonge'eti (H2)	Ngarenanyuki River	Ngereiyani village	393	1886
Sinya waterhole (H3)	Surface run-off	Sinya (EWMA)	7382	11222
Sinya waterhole (H3B)	Surface run-off	Sinya (EWMA)	3987	9497
Tingatinga (H4)	Simba River	Tingatinga village	420	1150
Ngainyamo (H5)	Ngarenanyuki River	Madebe village	135	1200
Ndarakwai (H6)	Leakage from hose pipe	Ndarakwai ranch	715	975

Figure 2.20 shows the time-series from September 2018 to January 2020 of the water volume change of the Sinya animal/livestock water hole (see Figure 2.14, site H3). From visual observations, this pattern was typical for the other water holes not fed by rivers or hose pipe in West Kilimanjaro. This water hole was located further downstream in Enduimet wildlife management area (WMA) and obtained its water largely from surface runoff during the wet season. The water volume declined during the dry season from an initial volume of about 14,000 m^3 at the onset in September 2018 to about 2,000 m^3 in April 2019, before rising slightly following the short rains in May and June, and then rising again from the end of October 2019 to January 2020 when it reached 18,000 m^3. In general, over a period of 17 months, the water hole volume declined by almost 80% within the first 8 months of observation, and from this lowest point it rose again by almost 700% to 18,000 m^3 within a period of 9 months during and following the wet season .

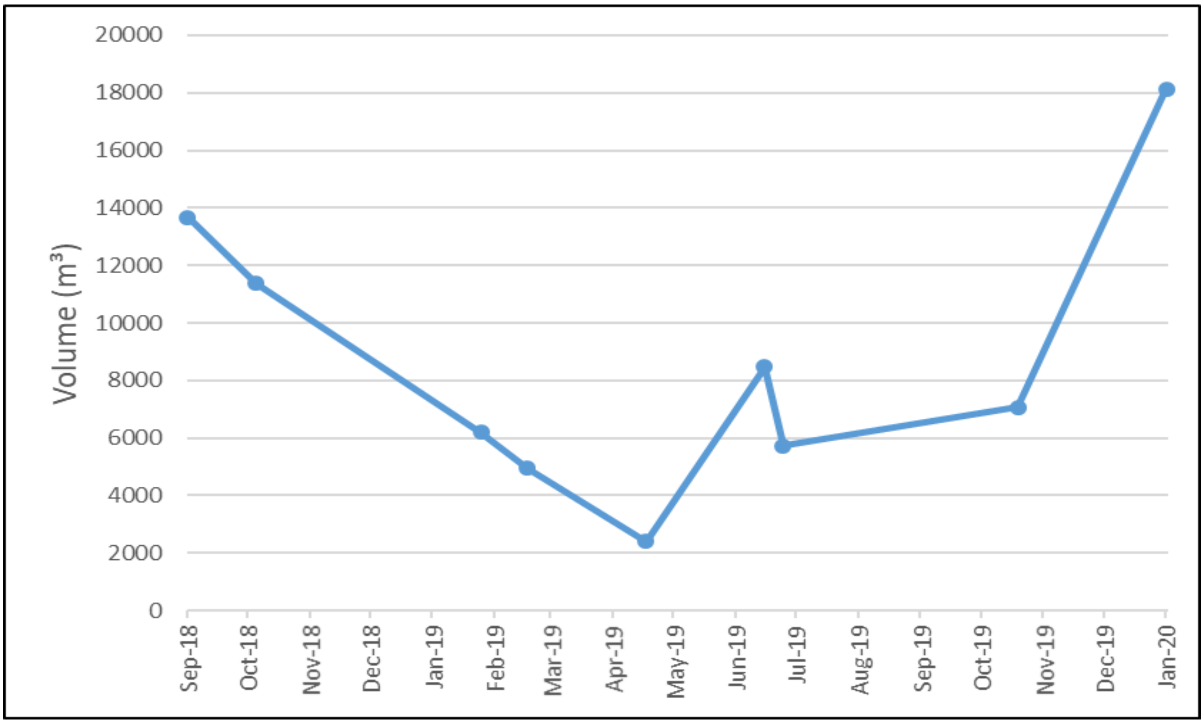

Figure 2. 20: Time series plot of the water volume at Site H3 (Sinya) man-made water hole in Enduimet WMA between September 2018 and January 2020.

2.3.4 Supplementary field observations in the West Kilimanjaro

In the West Kilimanjaro areas, a large amount of water was wasted by irrigation farming, because most of the irrigation canals were unlined and not fitted with control gates to conserve and regulate water flow. For instance, none of the five irrigation canals in the upland villages along the Simba River was fitted with control gates and only two were lined. Only two out of the five irrigation canals in the upland villages along the Ngarenanyuki River were partly lined and fitted with control gates. In addition, water was also wasted in the flood irrigation technique practised in the landscape. Further, the recently established large scale domestic water project that abstracts water from the upstream Simba River and conveys that water to the Longido district was poorly constructed as it lacked in-transit storage tanks, which led to frequent leakage of the conveyance pipe (Chairperson, Mitimirefu village, personal comm.). In addition, river channels were heavily sedimented, due to deforestation, overstocking and poor agricultural management leading to soil erosion in the catchment. In fact, the Simba and Ngarenanyuki River channels in the mid and lower part of the catchments, were so heavily affected by sedimentation that river banks no longer existed, leading to a considerable loss of water as the water did not flow further downstream along the channel but instead flooded laterally into village lands. The areas specifically affected were downstream of site N2, see Figure 2.1), and in the southern parts of Enduimet WMA, i.e. downstream of site S6, see Figure 2.1).

Based on the high resolution Google Earth images, the irrigation area (defined as all areas under irrigated crops farming) in 2018/2019 along Ngarenanyuki and Simba Rivers was estimated at about 90 km^2 and 25 km^2 respectively. Using a run-off coefficient of 0.5 in the irrigated farms as suggested by Ramachandra et al. (2014), and an average annual rainfall of 460 mm in these dry areas, the amount of rainwater retained by the soil in the farms and not returned to the river was estimated at 0.67 m^3/s and 0.19 m^3/s, respectively, for the Ngarenanyuki and Simba Rivers. Because of abstraction, on average, less than 30% of water in the upstream before abstraction in both the dry and wet seasons entered the downstream sites at Ndarakwai and Ngabobo in the Simba and the Ngarenanyuki Rivers (sites S4 and N2, see Figure 2.1). Consequently, the downstream areas which included villages, livestock and the abundant wildlife, were deprived of river water in most of the dry season. Even parts of the mid-downstream areas located immediately downstream of the major extraction sites experienced water scarcity during the peak of the dry season. For instance, there was no flowing water in Simba River at Ndarakwai wildlife ranch from July to September 2019 (pers obs.). As a result, wild animals and livestock drank water from few stagnant pools. Wild animals such as zebras, wildebeests and elephants moved into the neighbouring villages, and upstream the Simba River in search for water during the dry season. However, as the animals moved in search for water they damaged crops, water infrastructures and sometimes directly confronted people. As an ad-hoc way of sharing the available water between the upstream and downstream villages, some of the water in the Ngarenanyuki River was released to flow downstream during the night from Ngabobo. However only rarely did this water reach the downstream wildlife areas during the dry season due to river siltation and the resulting flooding of the riparian area, limited duration of the release and additional on-transit abstraction of water for tomato irrigation farming.

2. 4 Discussion

It is clear that the availability of surface water in the Kilimanjaro landscape varies both temporally and spatially, and that the changes in surface water availability is largely influenced by rainfall and water demands by humans in particular irrigation farming and domestic use.

2.4.1 Impacts of rainfall on surface water availability

Seasonal changes in surface water availability were in phase, with some time lag, with rainfall, reflecting the important role played by seasonal rainfall in water provision. The intensity and duration of the wet season rains were the main factors controlling the availability of surface water in the Kilimanjaro landscape. There was a suggestion of a small, but not statistically significant, long-term declining trend of the rainfall in the southern, windward slopes, and an increase in the northern, leeward slopes. Similar rainfall patterns on Mount Kilimanjaro were reported by Otte et al., (2017). Therefore, in the absence of human impact, variation in the availability of surface water as manifested in the upstream areas of the rivers, was historically largely controlled by the rainfall. This is evident in the pre-extracted upstream areas of the Ngarenanyuki and Simba Rivers whose low and high discharges respectively coincided largely with periods of none/low and high rainfall respectively in the catchment of each river. The few mismatches between rainfall and discharge in the upstream within the parks, were probably due to forest effects such as enhanced rain water infiltration, the steady release of run-off water into the rivers, plus groundwater inflow into the river during the dry season (Robinson et al., 2003; Benegas et al., 2014; Brogna et al., 2017). A study by Peña-arancibia et al. (2019) showed the key role ('sponge effect') played by tropical forests in facilitating a gradual release of run-off water into the rivers that ultimately extends the dry season water flow. In addition, topography and geology may also influence the degree to which any change in rainfall is reflected in the change in surface flow (Hallema et al., 2016).

The inter-annual variability of water availability is also important. During this study, the amount of water available in many streams and rivers in ANAPA was measurably higher than that reported by Elisa et al. (2016) for the period 2012/2013 (Figure 2.9). For instance, Ngongongare 1 which never contained water during the dry season in 2012/2013, recorded a significant amount of water (average 0.3 m^3/min) in 2018/2019 and there was downstream flow throughout the dry season. Such differences are due to the high inter-annual variability of rainfall. Averaging across all the meteorological stations (most of which are located in the wildlife rich areas) within ANAPA over a period of 20 years from 2001 to 2019, the average annual rainfall in the park was almost 1480 mm (ANAPA, 2020). From 2008 to 2013, the average annual rainfall in ANAPA was about 970 mm; for the period from 2014 to 2019, that is during this study, the average annual rainfall was almost 1900 mm which is markedly above the long-term average of 1480 mm (ANAPA, 2020). Thus, the period 2014 to 2019 was water-rich for wildlife as well as human needs. My study may thus overestimate the availability of water. Indeed, only 40% of the 20 years period recorded rainfall above average, and this suggests a high likelihood of water scarcity in the other years. This high inter-annual rainfall variability, evidenced by dry and wet years, is evident also throughout the East African region (Agrawala et al., 2003; Lalika et al., 2015; Mnaya et al., 2021).

2.4.2 Climate change impacts on surface water availability

Surface water availability in the lowlands downstream in the Kilimanjaro landscape depend on evaporation, which in turns depends on the air temperature and the wind. Although the data are limited, they suggest a significant increase in temperature in parts of the landscape. Indeed data in the southern lowland at the Moshi airport indicates that the maximum temperature is rising significantly (0.1°C per year) with time (R^2=0.5, $p<0.05$, t=5.17, n=31).

Since increased temperature also implies an increase in evaporation, surface water bodies especially open shallow lakes, water holes and rivers would be expected to lose water more rapidly. Indeed Lalika et al. (2015) suggested that such an increase in the temperature has caused an increase in evapotranspiration that might have in turn contributed to a decrease in surface run-off, river flow and water levels in the NyM reservoir recently. This evaporation effect is likely to be even important in the semi-arid areas north of Mount Kilimanjaro (KINAPA) and Mount Meru (ANAPA) as demonstrated by Dagg et al. (1970) and Nyingi et al. (2013).

The duration of the wet season varied between 90 and 120 days in any year. However, the inter-annual rainfall variability was enormous both in the northern leeward side and in the southern windward side. There was no significant correlation between the Southern Oscillation Index (SOI) and rainfall in the landscape. This suggests that rainfall and hence surface water availability in this landscape is not controlled by the El Niño-La Nina phenomenon, contrary to the assumptions of Wolff et al. (2011).

There is much debate regarding the magnitude of climate change impacts in the Kilimanjaro landscape (Said et al., 2019). However, in view of the large interannual variability of rainfall, there is no firm evidence for a significant climate change impact on the rainfall (Lalika et al., 2015; this study). Climate change may account for the small long-term trends in the annual rainfall (Figures 2.4 and 2.5) that vary between -5.0679 mm/year and 14.213 mm/year. Several studies have suggested that the rainfall in the Kilimanjaro landscape will be influenced by both local land cover changes and global climate, and that the temperature is expected to rise (Pepin et al., 2010; Otte et al., 2017). This is in agreement with the findings of this study, that ambient maximum temperature in the southern slopes are already increasing. Again, the evidence is unclear because the temperature data in Amboseli National Park since 1977 indicate a declining, but weak, long-term trend. Rainfall is predicted to increase in the future in the entire Kilimanjaro landscape (Otte et al., 2017; Kishiwa et al., 2018). However, in the present era of high human population and a high demand for irrigation water, there is predicted to be an increase in irrigation water deficit (available water minus irrigation water demand) to a value of almost 70% (Kishiwa et al., 2018). Therefore, climate change does not seem to be currently the dominant issue with regard to surface water availability in the Kilimanjaro landscape. Although, the glacier in Mount Kilimanjaro is disappearing, it is unlikely to cause a major impact on the hydrology of the Kilimanjaro landscape as the forest belt taps 90% of the precipitation which is the major source of water supply to springs and rivers in the landscape (Agrawala et al., 2003). Although two rivers are directly connected by very small streams to the glacier and hence a possibility of some surface water contribution (Agrawala et al., 2003), there are no hydrology data and it is not known where the large part of melt water goes and if the eventual disappearance of the glacier may affect groundwater in the Kilimanjaro landscape.

However, the impact of climate change on surface water availability may be significant if the temperature rises in the future as predicted, in synergy with human-induced factors such as excessive water extraction as the human population and its water demands keep increasing.

2.4.3 Impacts of abstraction on surface water availability

Assessment of the quantity of water in rivers and streams has shown that there was an often excessive demand for freshwater for domestic, irrigation and livestock use in the entire Kilimanjaro landscape, so that at times no surface water was available for downstream users, including humans and wildlife, during the dry season, although historically some of these rivers were perennial. The abstraction begins in the National Parks. Indeed, on average, existing extraction in ANAPA and KINAPA took respectively almost 90% and 70% of the available water during the dry season (Figure 2.8). However, no such high extraction of water occurred during the wet season. Water was extracted from both parks to mainly supply domestic water needs of the rapidly growing human population in the communities neighbouring the parks. In addition, some of the water extracted directly from the parks was also used for livestock watering and small-scale irrigation farming.

Outside the parks, the Ngarenanyuki and Simba Rivers, that originate in ANAPA and KINAPA, were strongly impacted by water extraction particularly for irrigation farming, as was documented also by other studies elsewhere (Hinrichsen, 2003; Jury and Vaux, 2007; UNESCO-WWAP, 2012), which have shown that irrigation farming is the largest consumer of extracted freshwater and account for more than 70% of all water used by humans world-wide. In some cases, this proportion is even higher, reaching almost 90% in the least developed countries, most of which are in sub-Saharan Africa (Jury and Vaux, 2007). Such high water for irrigation farming in sub-Saharan Africa, is due to among others low, short and variable rainfall, high temperature, poor management, poor regulations and a lack of law enforcement, a high population growth and poverty (Gommes and Petrassi, 1996; UNEP, 2006, 2010; Conway et al., 2009; Taylor et al., 2009; Wani et al., 2009; Le Quesne et al., 2010; McClain, 2013). While the Ngarenanyuki and Simba Rivers maintained high and relatively stable water discharge upstream from extraction sites within the parks, the discharge declined considerably downstream within the first 15-20 km as noted at Ngabobo and Ndarakwai (respectively at sites N2 and S4, see Figure 2.1). Both rivers were tapped to meet human needs, the Simba River being extracted from within the park for large-scale domestic water projects. Extraction commenced about 5 km into the park, and the river was also subject to substantial extraction further downstream and outside the park in villages for irrigation farming of vegetables, legume and cereals. Although the Ngarenanyuki River is too saline (see Chapter 3) for human consumption, water was still removed for irrigation farming particularly for tomatoes and cabbage production (Figure 2.21) and livestock watering. In common with the Simba River, extraction of water from the Ngarenanyuki River began well within a National Park, approximately 1 km from the park boundary, and in the next 20 km downstream, there were at least four irrigation farming canals and numerous pumps extracting water for irrigation farming within the village lands.

Figure 2. 21:(A) Water abstraction via a canal (arrow) in the Ngarenanyuki River at Ngabobo (N1), (B) Tomato irrigation farming (north of site N2) along the Ngarenanyuki River, (C) Dry season river drying out and siltation in the downstream (north of site N2) reach of the Ngarenanyuki River.

Excessive water abastraction causes the drying out of the downstream sections of rivers and the associated riparian wetlands in ANAPA. In KINAPA the drying out of rivers downstream, forces the wild animals, especially elephants, to damage water infrastructures in searching for drinking water. Downstream drying up of the Simba and Ngarenanyuki Rivers due to excessive up-stream abstraction also cause wildlife, again especially elephants, to search for water in the village lands during the dry season. This leads to human-wildlife conflicts which take various forms including damage of crops, infrastructures, injury and killing of humans, livestock and wildlife (Kikoti, 2009; Mariki et al.,2015; Okello et al., 2016). For instance, there was an incident in 2009 where six elephants that moved into villages were killed by villagers in the West Kilimanjaro (Mariki et al., 2015). Further, excessive water abstraction in the upstream villages causes water shortage in the downstream villages, which inturn results in conflicts between the upstream and downstream water users, including villages along the Ngarenanyuki River e.g. Ngereiyani vs Ngabobo, and the Simba River e.g.Tingatinga vs Mitimirefu (Chairpersons Ngereiyani and Mitimirefu villages, personal comm.).

2.4.4 Water budget in freshwater lakes and man-made water bodies

Water budget evaluation revealed that the amount of water in the lakes and reservoir within the study ecosystems also varied in phase with rainfall, in addition to groundwater discharge and/or surface water discharges from tributaries. Interactions between rainfall, surface- and groundwater in the southern slopes of Mt. Kilimanjaro was also demonstrated by Røhr (2003). His study, which focused on the assessment of hydrological conditions in light of interaction with land use, did not examine the water budget for the freshwater lakes existing in the Mt. Kilimanjaro lowlands. My study however, has explored the water budget in the freshwater lakes. Water levels showed a similar pattern to rainfall in terms of direction (increase/decrease) and variability, which is evident from the long-term rainfall data in the Kilimanjaro landscape. While neither of the water level trend was statistically strong, both Lake Amboseli and Lake Chala in the northern side of Mt. Kilimanjaro showed a rise in water level in phase with an increase in rainfall in the northern side of the mountain (Figure 2.15). Similarly, the water level in the NyM man-made lake (reservoir) (Figure 2.17), which gets its water from the windward side of Mt. Kilimanjaro and Mt. Meru, showed a slightly declining long-term trend matching the negative long-term trend in the rainfall in the windward southern side of Mt. Kilimanjaro and Mt. Meru.

Water level variations and patterns in the NyM reservoir also reflected changes in discharge of the major inflow rivers; Kikuletwa and Ruvu, and which in turn followed the catchment rainfall patterns. A ten years long hydrological data set (Pangani Basin Water Office (PBWO), 2020) indicates that the Kikuletwa River that drains from both Mt. Kilimanjaro and Mt. Meru contributes the largest amount to the reservoir with an average of 21.35 m^3/s, and the Ruvu River that drains Lake Jipe and the north-eastern parts Mt. Kilimanjaro contributes an average of 13.57 m^3/s. Both of these estimates are in agreement with previously reported values by Røhr (2003) who found values ranging between 19 m^3/s and 25 m^3/s for the Kikuletwa River and almost 10 m^3/s for the Ruvu River in the period between 1960s and 2000s.

The NyM reservoir was built in 1965 primarily for hydroelectricity production, but in later years it was also used to support irrigation farming (Pangani Basin Water Office (PBWO), 2008). During construction, it had a maximum depth of 29 m (Lalika et al., 2015). However, the water in the reservoir has been declining (Mulungu et al., 2007; Pangani Basin Water Office (PBWO), 2008). The mean water depth in 12 years period between 2008 and 2019 was 2.5 m (Figure 2.17) which is less than the mean water depth of 6 m reported by Denny (1978). In recent years, the river water flowing to the reservoir was substantially used for irrigation farming. This led to competition for water between irrigation in the upstream and electricity supply in the downstream (Lalika et al.,2015). The seasonal peaks in the NyM water level, always followed a peak in rainfall and also river inflow, and the few mismatches are likely due to water release from the reservoir for hydro-electricity production (Figure 2.17). The progressive fall from 2008 to 2012 of nearly 7 m to almost zero of the NyM water level is most likely due to both below average river inflow and high water release (outflow) for the reservoir operations in an attempt to produce hydroelectricity at the dam site (Murashani, 2012). The reduced surface inflow into NyM reservoir is largely attributed to a decline in rainfall in the southern slopes of Mount Meru and Kilimanjaro, an increase in temperature and hence evaporation, and an increase in water abstraction for irrigation farming in the upstream areas (Lalika et al.,2015). The water supply for the NyM reservoir is now threatened by a number factors especially excessive and illegal water abstraction for irrigation farming in the upstream areas, poor irrigation farming practices that lead to siltation in the inflow rivers, reduced rainfall and mainly increased temperature that enhances water evaporation (IUCN, 2007; Lalika et al., 2015).

Given its limited exposure to local human disturbance (Ruwa et al., 2004), Lake Chala serves as a suitable reference to delineate between anthropogenic and natural impacts on the surface water bodies in the Kilimanjaro landscape. Lake Chala water level showed an increasing but weak trend with time, between 2011 and 2019 (Figure 2.15). Water level in Lake Chala is mainly controlled by the groundwater inflow discharge, as local rainfall and evaporation are roughly in balance, and this was in synchrony with the rainfall at Rongai station on the northern slopes of Mt. Kilimanjaro. However, there were few mismatches (Figure 2.15), suggesting that other processes may occur, such as the lag time for the groundwater from the mountains to reach the lake, the storage of groundwater, and the probability that some of the groundwater might be more influenced by the regional-scale precipitation (Taylor et al., 2012) and thus not be adequately reflected by data from the Rongai rainfall station. Being largely dependent on groundwater, which results from the catchment rainfall infiltration on the mountain, a change in lake level is likely to reflect a change in the amount of catchment rainfall. For instance, as shown in Figure 2.19, the lake level rose within the short rainy season from October to December-2018 (total rainfall 132 mm), and declined as the dry season continued from January to March-2019 (total rainfall 79 mm), before rising again with the rain from April to end of June-2019 (total rainfall 377) (Tanzania Meteorological Agency (TMA)-Moshi, 2020). Lake Chala receives a net groundwater inflow of 0.18 m^3/s based on the most recent 8 years of monthly data. An earlier study by Payne (1970), based on yearly data from 50 years ago, reported net groundwater inflow of 0.138 m^3/s. These two estimates of groundwater inflow 50 years apart are in a pleasing agreement with each other, especially that my estimate used monthly data while the previous estimate used yearly data, and also considering that the rainfall varies inter-annually by nearly a factor of 3 (from 0.455 m/year to 1.276 m/year). How this will change with climate change remains conjectural. We know that the increased temperature due to global warming enhances montane forest fires and melting of the glacier in Mount Kilimanjaro (Agrawala et al., 2003). We also know that deforestation in the lowlands affect the rainfall in the forested highlands and in in turn this reduces the rainfall in the lowlands (Fairman et al., 2011). As the lowland savannah around Mount Kilimanjaro receives groundwater that largely originates from the forest belt (which intercepts fogs and receive higher rainfall) (Agrawala et al.,2003) the increased temperatures and the likely increase in forest fires may affect water availability in the Lake Chala. In addition, Lake Chala may soon be subject to water extraction to meet the increased water demand of a rapidly growing human population both in Tanzania and in Kenya, i.e. the Kenyan government has a proposed plan to abstract water from the Lake for irrigation farming (Ruwa et al., 2004).

Lake Jipe was comparatively stable during my one-year study, as the annual water level variation did not exceed 0.2 m, with an average depth of 1.14 m. Although Lake Jipe has significant surface water (river flow) input, it also showed some similarity to Lake Chala, whose water level fluctuated with rainfall and groundwater discharge. Lake Jipe and Chala are just about 30 km apart, in the dry area under the influence of rainfall in the leeward northern slopes of Mt. Kilimanjaro (Maina, 2019), and both receive a significant amount of groundwater inflow from Mt. Kilimanjaro, though the mechanism of interlinkages of this inflow is only known quantitatively (Payne, 1970; Røhr, 2003; Maina, 2019; this study). Unlike Lake Chala, Lake Jipe is exposed to direct human impacts (Njiriri, 2016), especially through excessive abstraction of the Lumi River for the irrigation farming (Ruwa et al., 2004; Ngugi et al., 2015). The Lake Jipe is said to have declined substantially from its original size of almost 100 km^2 to the current size of about 30 km^2 (Ndetei, 2006). In common with the other lakes under study, Lake Jipe is under serious threat from excessive water abstraction, siltation due to soil loss arising from poor farming practices, livestock overgrazing, and water pollution in the Lumi River basin and the lake shore (Ruwa et al., 2004; IUCN, 2007).

The variation of water level of Lake Amboseli was significantly correlated with the variation in the groundwater discharge, again suggesting that the lake water is mainly controlled by groundwater draining from Mt. Kilimanjaro, (as was also reported but not quantified by

Nyingi et al. (2013)) rather than by local rainfall. Indeed there is no significant correlation between local rainfall and the lake level. Lake Amboseli is exposed to high evaporation and low rainfall (Dagg et al., 1970; Nyingi et al., 2013) and groundwater inflow represents more than 50% of all the water in the lake. Compared to other lakes in the landscape, the Lake Amboseli water level has been relatively steady with the exception of sharp peaks reached during periods of very heavy rain of 2018. Excessive water abstraction during the dry season from the Ngarenanyuki and Simba Rivers possibly may not further threaten Lake Amboseli because, due to this water abstraction, river water already no longer flows to the lake during the dry season. The main threats to Lake Amboseli is then likely to be decreased rainfall on Mount Kilimanjaro as a result of forest fires and deforestation in the (Fairman et al., 2011; MEMR, 2012).

The availability of water in the water holes in the northern semi-arid lowlands, which were used by both livestock and wildlife, varied with time and space but were largely controlled by the amount of rainfall received and the irregular release of river water by upstream irrigation farmers. The water holes may be categorised based on their main source of recharge as: rain-fed (receiving water directly from rainfall or run-off), river-fed (receiving water from intermittent release of river water by upstream users), and tap-fed/water troughs (receiving water through underground pipes from KINAPA (Table 2.5). There were at least 7 rain-fed water holes but most of them lacked water during most of the dry season as received insufficient water to compensate for high evaporation. Of the 13 monitored water holes only two adjacent rain-fed water holes at Sinya retained water in the dry season during the period of this study. These were the water holes which resulted from the abandoned mine site located in the furthest downstream areas at Sinya in Enduimet wildlife management area. Being located in the furthest and relatively flat downstream area, they received much of the run-off water draining from the upstream areas during the wet season. In this mine site, there were also several other water holes but possibly due to their small depth (less than 2 m which is smaller than the annual evaporation loss in the semi-arid areas) they were dry during the dry season. Monitoring results from one of these two water holes that kept some water in the dry season, revealed a high variability in water volume between the dry and the wet seasons, whereby over the dry season the water hole volume declined by almost 80%. This large change in volume is mainly attributed to high evaporation but also high water consumption by the wild animals and livestock. These rain-fed water holes were located in the drier areas of East Africa exposed to a high water demand and high potential evaporation of a magnitude between 1800 mm and 2200 mm annually, which is, on average, more than threefold the annual rainfall (Dagg et al.,1970). This explains why most of the water holes lacked water during the dry season, and those two that retained water barely did so.

In some downstream village areas, river-fed water holes and tap-fed cattle troughs were established for livestock watering. The amount of water in the river-fed waterholes varied seasonally. Based only on those water holes where water was available all year, the volume of water ranged from 100 m^3 in the dry period to almost 2000 m^3 in the wet period depending on the release of water from the Simba and Ngarenanyuki Rivers by the upstream users. Both livestock and wild animals consumed water from these water holes but livestock probably consumed a larger proportion of the water due to their larger numbers and also because human disturbance reduced usage by wild animals for most of the day.

The tap-fed waterholes/troughs generally contained water in all seasons but some of them also experienced intermittent water shortage depending probably on the availability of tap-water from KINAPA and also demand of the water by humans and the livestock. In some cases due to low water supply during the dry season, water was restricted for domestic use only and it was thus unavailable in some of the troughs. The water troughs were also located close to human settlements and therefore they were accessible to wild animals only at night when there was relatively low human interferences. Therefore, human action was the main factor controlling the availability and accessibility of water for wildlife in these man-made water sources. This is also in agreement with the study by Ogutu et al., (2014) in Kenya’s Masai-Mara Reserve and the adjacent livestock ranches who demonstrated human disturbance as one of the factors that limit wild animals’ access to surface water in East African savannah. There is no sustainable availability of water in the existing water holes and tap-fed water holes/troughs. This not only poses a great health risk, especially to the wild animals due to greater risk of transmission of diseases (Ogutu et al., 2010), but it also elevates the risk of increased human-wildlife conflicts, as the wildlife are likely to confront people and to damage water infrastructures in their search for water. As the dry area receives much less rainfall compared to amount of water lost by evaporation, timely release of sufficient water from the upstream Ngarenanyuki and Simba Rivers to the downstream areas including the water holes is essential in sustaining water supply for the wild animals and livestock. However, with the increasing human population and associated water demands, the future of water availability in such shared man-made water sources is threatened. Therefore, a balanced water allocation and water use efficiency should be promoted and enforced to meet both human and animals water needs. Establishment of independent/separate water supply systems for the livestock and wild animals, placed away from human disturbances might also help in alleviating the current water problems facing the animals.

Based on the findings of this study, it is apparent that the water scarcity experienced in the downstream wildlife-rich areas was mainly due to unsustainable water extraction practices to serve various human needs in particular agriculture and domestic use, that are continuously growing due to the rising human population (Mckenzie et al., 2010; Said et al., 2019). In parallel, there is also increased demand for water for cash crops to generate income (Istituto Oikos, 2011). The impact of excessive water abstraction is compounded by unsustainable land use practices downstream of the National Parks ANAPA and KINAPA.

2.4.5 A comparison between the Kilimanjaro landscape and Katavi-Rukwa ecosystem with respect to surface water availability for wildlife

The Katavi-Rukwa ecosystem is located in western Tanzania, where the Katavi National Park (KNP) is one of the richest wildlife areas in Tanzania (Figure 2.22A). The Katavi-Rukwa ecosystem which consists of KNP among other wildlife protected areas, is characterised by miombo woodland. KNP is famous for its abundance of large mammals particulary hippoppotamus, buffalos and elephants, all of which are water dependent species (Caro, 1999). Being located downstream of the perennial, upper section of the Katuma River, the park largely depends on that river which drains the high rainfall Mpanda-Mwese forested ranges in the northwest of KNP (Elisa et al., 2021). Similar to river systems in the Kilimanjaro landscape, the Katuma River discharge entering KNP shows a strong seasonal variability that follows the rainfall seasonal variability (Figure 2.22). Before the river enters KNP, its water is heavily extracted for human activites, especially rice irrigation farming since late 1990s (Elisa et al., 2010 and 2021). The Katuma River provides water in KNP to wildlife rich, floodplain wetlands including Lake Katavi, Katisunga Plains, Lake Chada. The Katuma River finally empties further south into Lake Rukwa, which is a closed lake (Figure 2.22A).

As shown in Figure 2.22B, the Katuma River discharge reaches its largest value during the wet season between February and April. About 50% of water in the upstream Katuma River was extracted for irrigation farming in 2017 (Elisa et al., 2021). The remaining water discharged into KNP and its flow rate decreased rapidly over time during the dry season and ceased in the middle of the dry season. Thus, owing to excessive water abstraction for irrigation, the river which used to flow perennially has since the early 2000s become seasonal and no longer flows during the dry season. As a result the wetlands in KNP are drying up and the wildlife suffers from lack of water, while the water level of Lake Rukwa has decreased by 3-4 m (Elisa et al., 2010 and 2021; Figure 2.22C).

Seasonal variability in river discharge also occurs in the Simba and Ngarenanyuki Rivers in the Kilimanjaro landscape. These rivers were also perennial all the way to the downstream areas before excessive water extraction started in the 1990s. As a result, similar as for the Katuma River, the discharge downstream is now much smaller than upstream, due to water abstraction. In common with the Katuma River, these rivers have ceased flowing downstream of the extraction during the dry season, e.g. there was no water in the downstream of sites S4 and N3 in the Simba and Ngarenanyuki Rivers respectively (Figures 2.1 and 2.12).

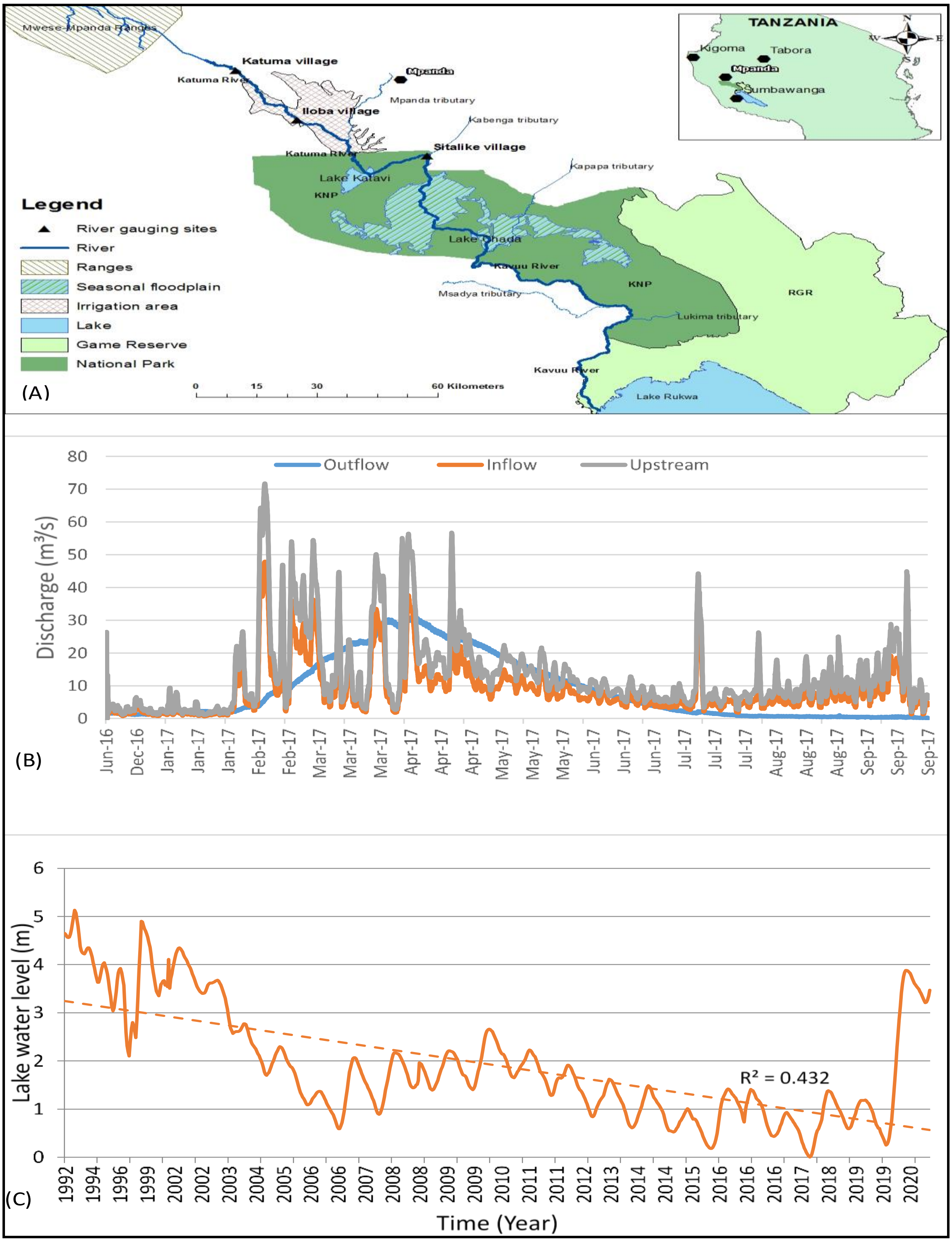

Figure 2. 22: (A) A sketch map of the Katavi ecosystem. (B) Time series plot showing the Katuma River water discharge (labelled 'upstream flow') of the Katuma River upstream of the irrigated areas, the inflow into Lake Katavi downstream of the irrigated areas, and the outflow from Lake Katavi forming the Katuma River entering KNP during 2016-2017. (C) Time series plot of the water level in Lake Rukwa as measured by satellite altimetry. Adapted from Elisa et al. (2021).

Abstraction from the Katuma River resulted not only in the decrease of water level of Lake Rukwa but also of Lake Katavi in KNP which was very shallow or dry in all years except following the exceptionally wet year 2020, when the depth increases to almost 3 m (USDA, 2020; Elisa et al., 2021). Likewise, excessive water abstraction from the Simba and Ngarenanyuki Rivers has resulted in Lake Amboseli not always receiving river water, particularly during the dry season.

In the context of surface water availability particularly in the rivers, the Kilimanjaro landscape and the Katavi ecosystem have similarities but they also differ in certain important respects. The Katavi ecosystem is entirely a downstream wildlife area mainly fed by the Katuma River. The Kilimanjaro landscape is a mix of upstream and downstream wildlife areas where ANAPA and KINAPA are upstream areas from which rivers originate, and the dry West Kilimanjaro area encompasses the downstream wildlife areas, fed mainly by the Simba and the Ngarenanyuki Rivers that receive water from the parks. Further, the river discharges in both ecosystems are strongly influenced by rainfall and therefore the seasonal and inter-annual discharge variability largely reflects the rainfall variability. The rivers in these two ecosystems used to contribute substantial amount of water to the lakes or floodplains located in the downstream areas, with the Katuma River flowing into Lake Rukwa, and the Ngarenanyuki and Simba Rivers in the Kilimanjaro landscape draining to the Amboseli basin. However, none of the rivers now supplies these downstream areas during the dry season and the Ngarenanyuki and the Simba Rivers only reach Lake Amboseli during occasional extreme heavy rains. The reduction/elimination of river discharge has likely contributed to the decline in area and depth of the respective lakes, for instance decline of Lake Jipe and NyM reservoir in the Kilimanjaro landscape (Ndetei, 2006; Lalika et al., 2015), and a decline of Lake Rukwa in the Katavi-Rukwa ecosystem by almost 4 m since 1992 (Elisa et al., 2021).

In terms of vulnerability, both the Kilimanjaro landscape and the Katavi-Rukwa ecosystem are vulnerable to both climate change impacts and water extraction pressure. However, the Katavi-Rukwa ecosystem may be more vulnerable to these factors as the Katuma River is directly and broadly exposed to human-induced impacts from farming over a stretch of more than 120km. In addition, the Katuma River has a larger area under irrigation farming and hence more water is abstracted and retained in the field for crop production. The area under irrigation north of Katavi National Park is almost 500 km^2 (Elisa et al.,2021) compared to less than 150 km^2 under irrigation in both Ngarenanyuki and Simba Rivers. The Katavi-Rukwa ecosystem is further prone to negative impacts from high human and livestock population, resulting from the internal migration of pastoralists and livestock from the western Tanzania (Salerno et al., 2017). The upstream end of Katuma River is in a district forest reserve while the downstream end is Lake Rukwa, part of which falls under the management of the district council. District councils have less capability to provide strict protection of these areas than the National Parks in which both the upstream and the downstream parts of the Simba and Ngarenanyuki Rivers are located. Being located in the Amboseli National Park, Lake Amboseli is relatively well protected.

Thus in all these three catchments (Katuma, Ngarenanyuki and Simba Rivers) the downstream-most areas have most suffered from excessive water abstraction upstream. Clearly water resources management at the catchment scale is much needed for these three rivers; these rivers are located in different regions of Tanzania but their water problems are similar, and the resulting water crisis indicate a malaise of non/miss-governance of water resources in Tanzania.

This study aimed to assess the change in surface water in term of its availability, and evaluate how human and natural factors are contributing to such changes, with a main focus on the wildlife-rich areas in the Kilimanjaro landscape. While the aim has been addressed, there are several limitations including: (i) The short and unexpected seasonal patterns of rainfall, in particular the timing and extended length of the dry season. The resulting shortening of the wet season led to low coverage of the wet season; (ii) Not all water sources in KINAPA could be studied due to limited resources given the large expanse and difficult terrain of the park. However, the selected sites appear to be sufficiently representative (from local knowledge of the park rangers), and include local wildlife rich areas, especially those on the leeward side, which has until recently received minimal attention; (iii) Difficulty in quantifying the change in river-fed water holes which were fully under human control and operated on an intermittent basis only; (iv) A lack of long-term water level data limited the valuation of water budget and changes in water availability in the freshwater lakes to a short time period.

2.5 Conclusion

This study aimed to assess the change in surface water in term of its availability, and evaluate how human and natural factors are contributing to such changes, with a main focus on the wildlife-rich areas in the Kilimanjaro landscape. While the aim has been addressed, there are several limitations including: (i) The unexpected seasonal patterns of rainfall, in particular the timing and the duration of the dry season. The resulting shortening of the wet season led to low coverage of the wet season; (ii) Not all water sources in KINAPA could be studied due to limited resources given the large expanse and difficult terrain of the park. However, the selected sites appear to be sufficiently representative (from local knowledge of the park rangers), and include local wildlife rich areas, especially those on the leeward side, which has until recently received minimal attention; (iii) Difficulty in quantifying the change in river-fed water holes which were fully under human control and operated on an intermittent basis only; (iv) A lack of long-term water level data limited the valuation of water budget and changes in water availability in the freshwater lakes to a short time period.

This study has demonstrated that spatial and temporal changes in the availability of surface water in the Kilimanjaro landscape are dependent on both natural and anthropogenic factors but their relative importance varied. Patterns of the change in the freshwater lakes reflected the different rainfall patterns in the northern and southern slopes of Mt. Kilimanjaro and Mt. Meru. There was no evidence that the mean annual rainfall changed significantly since the 1970s, nor that it led to significant changes in the availability of surface water. This is also in agreement with a recent study by Lalika et al. (2015), who found no empirical evidence to associate climate change with the decline in rainfall and surface water availability in the Upper Pangani River Basin whose water largely originates from the Kilimanjaro landscape. As climate change has not to date resulted in a statistically significant change in rainfall duration and intensity, in view of its large interannual variability, its impact of surface water is therefore minimal so far. Anthropogenic impacts did however result in a marked decrease in surface water availability downstream of abstraction sites within and outside the forested National Parks. There was a significant reduction in the river discharge with distance downstream of the parks and this decrease was due to water abstraction to meet human needs. This abstraction was quantified in rivers and streams within the parks and outside the parks for the Ngarenanyuki and Simba Rivers located in the lowland drier areas north of Mt. Meru and Mt. Kilimanjaro. The existing water abstraction was mainly for irrigation farming, which is also the largest consumer of freshwater elsewhere in Africa (Hinrichsen, 2003; Jury and Vaux, 2007; UNESCO-WWAP, 2012). This water abstraction was excessive and unsustainable as it caused serious deprivation of water to the downstream areas populated by people, wildlife and livestock. This was particularly evident in the dry season in the Simba and Ngarenanyuki Rivers. As water demands and unsustainable use continue to grow in the landscape, so is the water crisis, which will ultimately evolve into a wider environmental, and social-eoconomic crisis. It is known that unsustainable abstraction of water has far reaching social-ecological and economic impacts as reported for other river basins in Tanzania, such as the Katavi-Rukwa ecosystem, and for East Africa in general (Gichuki, 2002; Liniger et al., 2005; Mtahiko et al., 2006; Elisa et al., 2010 and 2021; Stommel, 2016). Clearly, there is an urgent need for establishing ecologically sustainable water resources management plan and practices at the watershed scale in the entire Kilimanjaro landscape.

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Chapter 3: Assessment of the current water quality status in the Kilimanjaro landscape

Abstract

Global freshwater resources of high quality are continually declining in quantity due to unsustainable use as the rapidly growing human population continues to over-abstract and pollute freshwater to meet socio-economic demands, and result in adverse impacts on natural ecosystems. This chapter assesses the changes in surface water quality due to natural or anthropogenic factors in the Kilimanjaro landscape. The status of water quality in the ecosystem is evaluated with reference to guidelines, in particular with respect to what is tolerated by wildlife. Impact of water quality at the landscape level was also examined, with the emphasis on fluoride and salinity in the light of previous studies revealing elevated levels of fluoride and other dissolved salts in Arusha National Park and the neighbouring areas (Kilham and Hecky, 1973; Elisa et al., 2016; Malago et al., 2017). Water quality, specifically salinity, DO, pH, temperature, fluoride, nutrients, water hardness and heavy metals, was assessed at ecologically important sites and upstream points supplying these sites. Water quality results were compared with quality guidelines and the literature to determine whether they may be detrimental to wildlife and the wider ecosystem.

While water varied across space and time, water across the landscape was largely of good quality, including for wildlife use. In general, the concentration of physicochemical parameters increased with the dry season for all parts of the landscape. The quality of water in the National Parks did not seem to be significantly impacted by anthropogenic water abstraction or pollution. However, water quality in the areas outside the National Parks, particularly in the low lying and dry community wildlife areas was adversely affected by both pollution and excessive water abstraction, which reduced the effect of dilution and hence increased mineral concentration. However, increase in the dry season, of the mineral concentration in waterholes located in the semi-arid areas, was further exacerbated by more rapid evaporation in these waterholes, which had large surface areas to volume ratio. During the dry season, the metals, salinity and nutrients in some surface waters of the semi-arid areas were likely too high to support large mammals and biodiversity health. In

particular, high metal concentration may be toxic and adversely affect ecological health especially over a prolonged exposure (Alloway, 2012). Therefore, an urgent management action aimed at controlling water extraction and reducing pollution is required to address the continuing threat of dry season poor quality water in the semi-arid wildlife rich areas.

3. 1 Introduction

Good water quality is essential for a healthy ecosystem and improved human well-being (Ward, 1998; Vörösmarty et al., 2000; Dudgeon et al., 2006; Mallya, 2007; UNEP, 2010; Grafton et al., 2013; McClain, 2013; Richter et al., 2015; WHO & UNICEF, 2015). However, water pollution is increasingly pervasive, in addition to being the number one cause of human deaths world-wide (Jury and Vaux, 2007), it is also adversely impacting on wild animals and biodiversity at large (Stewart et al., 2008; UNEP, 2008). Freshwater pollution, especially through direct solid waste (mainly human and agricultural waste) disposal into streams, rivers and lakes, is most prevalent in sub-Saharan Africa (UNEP, 2006). Anthropogenic impacts on surface water is pervasive in Tanzania, leading to deterioration of quality and resulting impacts on biodiversity (Mohammed, 2017). The suspended solid load in water is increased as a result of run-off due to poor land management. Over-abstraction of surface freshwater also leads to poor water quality due to reduction in water flow that in turn leads to a reduction in dilution of pollutants. For instance, Stommel et al. (2016) found that abstraction of the Great Ruaha River for irrigation during the dry season caused a marked reduction in river flow, leading to poor water quality, including through increased salinity and bacterial load in the wildlife rich ecosystem in Ruaha National Park, Tanzania. Poor quality water affect wildlife behaviour, distribution, and species throughout sub-Saharan Africa. Water abstraction and damming of the Olifants River for irrigation farming, and mining activities have led to reduced water flow and increased pollution, including metal contamination, that adversely affected tiger fish ecology and the wider aquatic ecosystem in Kruger National Park in South Africa (Smit et al., 2013). Increased nutrient concencentration which is partly caused by reduced flow, has provided a conducive environment for encroachment by invasive plant species in Lake Jipe (Ndetei, 2006; MEMR, 2012). In the Serengeti and Ruaha National Parks, Tanzania, the wild animals are known to avoid poor quality water such as water with high salinity concentration and/or bacterial load (Wolanski et al., 1999; Strauch, 2013; Stommel et al., 2016). Following a marked reduction in the Great Ruaha River flow, and the resulting poor quality water in the Ruaha National Park, animals such as African elephant (*Loxodonta africana*), plains zebra (*Equus quagga*) and warthog (*Phacochoerus africanus*) have developed a behaviour of actively digging water holes in search of alternative good quality drinking water and which is then available to other wildlife (Stommel et al.,2016; Mnaya et al., 2021).

While surface freshwater has been abstracted to meet socio-economic needs for many years in the Kilimanjaro landscape, little is known about the impact of abstraction on the water quality at the ecosystem level. Several rivers and streams are abstracted for domestic use in Arusha National Park (ANAPA) and Kilimanjaro National Park (KINAPA), and for irrigation farming as they traverse the neighbouring community lands before entering the downstream wildlife-rich semi-arid areas, and such abstraction and associated activities are likely to result into a reduced water quality. In addition to human impacts, natural factors such as rainfall, evaporation, floods, and rock weathering are known to influence water quality (Espinoza-quiñones et al., 2005; Shanbehzadeh et al., 2014), and therefore, they may also be altering the quality of water in the Kilimanjaro landscape. Very little is known about the changes in freshwater quality in the wildlife-rich areas on the northern slopes of Mt. Meru and Mt. Kilimanjaro and the semi-arid areas further downstream (Mohammed, 2017; Said et al., 2019). The few existing water quality studies mainly targeting impacts on human health have focused on the more urbanised and densely populated windward southern slopes of Mt. Kilimanjaro and Mt. Meru (Røhr, 2003; Kaseva and Moirana, 2010; Mckenzie et al., 2010; Ndalilo et al., 2020). The shortage of comprehensive studies on water quality in Tanzania is acknowledged (Hellar-kihampa, 2017). Such deficit is even more common in the wildlife protected areas especially those in the upper catchment as noted in a recent study by Elisa et al. (2016). This study, though with a limited spatial and temporal scope, sheds some light on the status of water quality in the perspective of human activity and biodiversity in ANAPA.

This study aims to assess temporal and spatial changes in surface water quality due to natural and anthropogenic factors focusing mainly on water abstraction, and their effect on biodiversity health at the ecosystem scale in the Kilimanjaro landscape with particular reference to wildlife. The main focus in the lowland drier areas was the Ngarenanyuki and Simba Rivers which are the key perennial rivers that supply water to the dry wildlife-rich and community areas. These rivers provide water for domestic, and livestock use and are excessively abstracted for irrigation farming such that water is rarely available to the entire downstream wildlife areas during the dry season. Water quality is defined in terms of the levels of physical and chemical properties and their current acceptable limits and guidelines (UNEP, 2008). The findings of this study will contribute to the much-needed improvements in water resource management at the watershed scale for both human and ecological needs in Tanzania and elsewhere in sub-Saharan Africa.

3.2. Methodology

Sites, parameters selection and sampling

Key water sources in the study ecosystem, were identified, and mapped using GPS unit and ArcGIS software. The selection of the study sites took into account spatial coverage, water extraction status (degree of extraction/no extraction), possible point sources of pollution, and the ecological importance of the water source to wildlife. Most of the sites selected for water quantity assessment in Chapter 2 were also assessed for water quality and, again included both sites subject to water extraction and those currently not subject to water extraction. A total of 57 surface water sites (Figures 3.1 and 3.2) in KINAPA, ANAPA and in the lowland drier areas largely north of Mt. Meru and Mt. Kilimanjaro were chosen and sampled for water quality assessment. The focus of the study was mainly on the dry season when water is scarce and thus both a limiting ecological factor (see Chapter 2) and has a greater potential for poor water quality.

Sampling occurred from September 2018 to February 2020. The focus was on parameters that were likely to have an impact on wildlife, and hence to facilitate informed decision making to foster ecologically sustainable water resource and biodiversity management. Several other sites in the Ngarenanyuki and Simba Rivers, including one site located in the upstream within the park, were monitored for water quality (Figure 3.1): Ngarenanyuki

River sites N1, N2, & N3, and six sites ; S1, S2, S3, S4 , S5 and S6 on the Simba River. In addition, water holes (sites HI, H2, & H3) located in the downstream community wildlife areas (Figure 3.1) were assessed as they form an important source of water for the wildlife and livestock especially in the dry season when river water seldom reaches the semi-arid downstream areas due to over-abstraction of water in the upstream areas (see Chapter 2). Further, sites in rivers and streams were assessed in the National parks, where sites A1 to A33 were located in ANAPA, while sites K1 to K4 were in KINAPA (Figure 3.2).

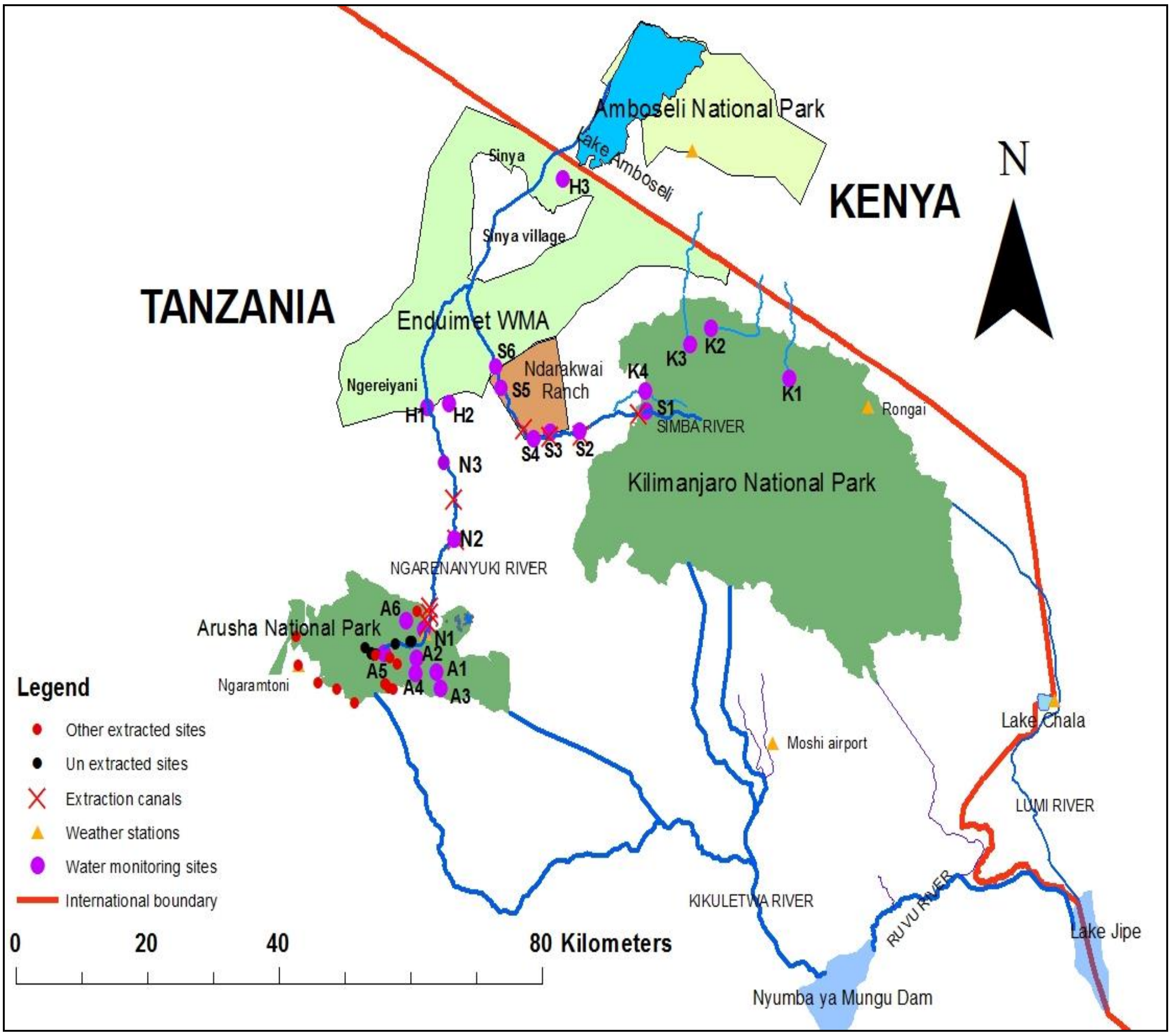

Figure 3. 1: Location of the water quality monitoring sites in the Kilimanjaro landscape.

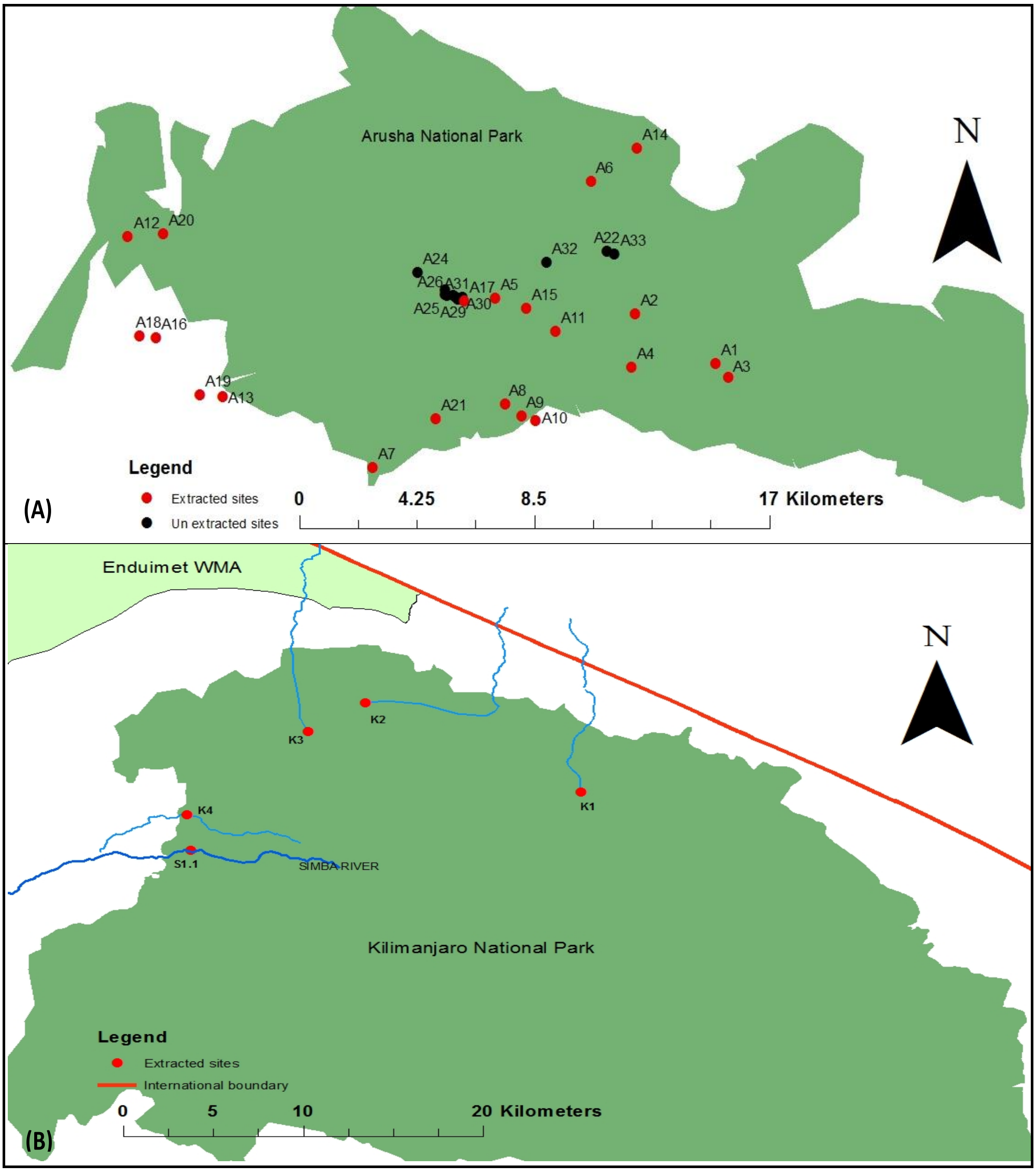

Figure 3. 2: Enlargement of the area encompassing the water quality sampling sites in (A) ANAPA and (B) KINAPA. In ANAPA, a few sites are in the adjacent forest reserve but are all counted as being in ANAPA.

3.2.1 Data collection

Salinity, DO, pH, and temperature were directly measured on site. Other key water quality parameters were measured in the laboratory; specifically, fluoride, water hardness (calcium and magnesium), nutrients (nitrate and phosphate) and heavy metals.

For each extracted water source, water quality parameters (those measured onsite) were measured, on average, once per month mainly during the dry season in the upstream above extraction site and downstream after extraction at 25 sites on the rivers and streams to evaluate the impact of water extraction on water quality within the National Parks. Measurements were also taken in the wet season to compare water quality during the period of maximum dilution. Out of the monitored 57 sites, 11 sites were un-extracted rivers and streams in ANAPA, which were selected to provide an indication of water bodies whose water quality would be suitable for wildlife as such information would be useful in water resources and biodiversity planning and management, and such additional sources would assist in reconciling the conflicting demands on water resources. To examine heavy metals, water hardness and fluoride, freshwater samples were collected from the study sites in the Kilimanjaro landscape and analysed in the laboratory. Sites were sampled once in the dry and wet season, resulting in 37 dry season sites and 19 wet season sites.

A portable hand-held device (Extech EC400 Waterproof ExStik II) was used for onsite measurement of salinity, DO, pH, and temperature, and was in accordance with the sampling strategy in Elisa et al. (2016). At these sites, water samples were collected and analysed at the Ngurdoto Defluoridation Research Station laboratory for fluoride (F), nitrate (NO_3), phosphate (PO_3), and at the University of Manchester for trace metals, and water hardness. Water samples for F, NO_3, and PO_3 were collected in labelled polythene/plastic bottles, and analysed in the laboratory using an ion-selective electrode (Milham et al., 1970; Tokalioglu et al., 2004). Water samples for heavy metal analysis were collected in 20ml tubes, labelled and immediately acidified to pH 2.0 with ultrapure nitric acid. The samples were filtered (0.45 micron) in the lab and transferred into 15 ml labelled plastic tubes. The samples were analysed for Aluminium (Al), Iron (Fe), Manganese (Mn), Zinc (Zn), Copper (Cu), Cadmium (Cd), Arsenic (As) and Lead (Pb) using Inductively Coupled Plasma Mass Spectrometry (ICP-MS). ICP-MS is the best analytical technique with a high precision and accuracy for identifying and quantifying trace elements (Ammann, 2007). For the analysis of water hardness (Calcium and Magnesium), Inductively Coupled Plasma-Optical Emission Spectroscopy (ICP-OES) was used. ICP-OES has the ability to identify multiple elements and their ratio in complex samples (de Oliveira Souza et al., 2015). Some water samples were measured twice (proof-measuring) using the same method and operator and on the same day (Bhattacharya et al., 2017). To ensure quality control, a sample of known water quality as well as paired samples (with different identification code) taken from the same source and time were included among the set of other samples to verify the validity and the reliability of the laboratory analyses. None of the samples subjected in this quality procedure resulted in unexpected values.

3.2.2 Data analysis

Data on water quality were subjected to descriptive analysis to compute measures of central tendency (mean) and dispersion (standard deviation and errors) for various water quality parameters. In addition, I used regression analysis in R software to elucidate trends and relationships among the parameters over space and time. Statistical modelling took into account the most important variables that had a high likelihood of influencing temporal and spatial changes in water quality parameters across the landscape. Variables taken into account were also those which were known or could be clearly defined and/or measured in this study. These variables were; location of water point (as either inside or outside the park), type of water source (river/stream or water hole/standing water), season (as either dry or wet season) and concentrations of key water quality parameters plus pH. A knowledge of pH is particualry important in the analysis of heavy metals whose concentration in the environmemt is often pH dependent. There was a number of other variables such as geology/soil type, which might affect the concentration of water quality parameters such as fluoride; however they were not included in the statistical modelling as the data is not available and time and resources precluded direct analysis during the course of this study. Regression was mainly used to explore the nature and strength of the relationships among various parameters, whereas a t-test and coefficient of variation (CV) were used to the explore variations among the parameters. To address the issue of data auto-correlation, e.g. in the case where data had less spatial or temporal independency, the data were averaged and analysis carried out on the means (Crawley, 2005). The Accumulation Factor (AF) was used to provide an indication of the magnitude of the human impact on water quality between upstream and downstream sites in the Simba and Ngarenanyuki Rivers as used by Bhat et al., (2014). AF was calculated as the ratio of the mean value of a given parameter in the downstream (after human impact) to the mean value of the same parameter in the upstream (before human impact) (Fakayode, 2005). To establish the capacity of the rivers' self-purification, the degree of River Recovery Capacity (RRC) was computed for the Simba and Ngarenanyuki Rivers using the following equation as applied by Fakayode,2005 and Bhat et al., (2014) :

$$\frac{S_0 - S_1}{S_0} \text{ x } 100 = \text{RRC}.$$

Where, S_O is the value parameter downstream after abstraction, and S_1 is the corresponding average value of parameter in the upstream before abstraction. The RRC is expressed in percentage.

Due to limited availability of water quality guidelines for the wildlife use, concentrations of water quality parameters were assessed against the range of acceptable levels in drinking water for humans and livestock to determine their suitability for the wildlife and also human use. Several water quality guidelines including WHO guidelines for human drinking water (WHO, 2003b, 2004a, 2004b, 2010, 2011), and various state guidelines for livestock drinking water (DWAF, 1996b, 1996a; Bagley et al., 1997) were used.

3.3 Results

Water quality data from September 2018 to February 2020 revealed spatial and temporal variations in all water quality parameters across extracted and non-extracted water sources in the landscape. In this section, I first provide a summary of results based on the major findings on the assessed water quality parameters, and then I provide a detailed presentation on the findings for each water quality parameter.

3.3.1 Summary of water quality results

DO and Temperature

In the water extraction sites within the National Parks, DO ranged from 3 mg/l to almost 8 mg/l. Ngongongare 1 in ANAPA (site A1 in Figure 3.2A) showed a significant difference ($p<0.001$, $t=-20$, $n=6$) in DO between up-and downstream during the dry season. Overall however, there was no significant difference ($p>0.05$, $t=0.87$, $n=25$) in DO between up- and downstream sites of the same watercourse and none of the recorded differences exceeded 2 mg/l in ANAPA and KINAPA. During the rainy period, the rivers were fully oxygenated throughout with DO ranging from 8 mg/l (114% saturation) on the Simba River entry to WMA to about 10 mg/l (140% saturation) on the Ngarenanyuki River at Madebe. Water temperature in these rivers increased almost linearly with increasing distance downstream and the values ranged from 14°C to 26 °C. Water holes in the semi-arid areas recorded comparatively low dry season DO concentration of less than 5 mg/l with Ngereiyani water holes having the lowest values, with an average of 2.2 mg/l at site H1, followed by site H2 with 4.5mg/l, and then site H3 which measured 4.87 mg/l. As would be expected, temperature in the water holes and other standing waters was higher than the running waters, especially during the dry season, ranging between 20°C and 31°C.

Salinity

Overall, ANAPA recorded higher salinity values than KINAPA, ranging from as low as 20 ppm to almost 340 ppm in the water sources extracted mainly for domestic use. There was no difference ($p>0.05$, $t=0.76$, $n=25$) in salinity between up-and downstream sites across all sites within ANAPA and KINAPA. However, for the Simba and Ngarenanyuki Rivers, salinity showed an increase downstream in the areas outside the parks. Salinity ranged from almost 20 ppm inside the park to 60 ppm outside the park in the Simba River, and nearly 800 ppm inside the park to 1000 ppm outside the park in the Ngarenanyuki River. There was a larger increase in salinity concentration in the Simba River than the Ngarenanyuki River in the downstream areas as reflected by accumulation factor values of 3 and 1 at sites S6 and N3 respectively (Figure 3.1). As expected, the highest salinity concentrations in the landscape were recorded in the water holes (Sites H1, H2, H3 & H5) of the semi-arid areas. Salinity was higher particularly during the dry season. On average, dry season salinity values ranged from almost 1,300 ppm at Ngereiyani water hole (Site H2) to almost 4,000 ppm at Sinya water hole (Site H3).

pH

Generally, the pH level in the landscape was alkaline. Within the National Parks and along the Simba River, pH ranged between 7.5 and 9.0, whereas the pH in the Ngarenanyuki River ranged from almost 9.37 to 9.80. The water holes (Sites H1, H2 & H3), and the other standing waters recorded a distinctly alkaline pH of between 9.0 and 10.0. There was no significant difference ($p>0.05$, $t=1.7$, $n=25$) in pH value between up-and downstream of the extraction sites in the National Parks and did not exceed 0.7. pH also did not change between up-and downstream of extraction sites in the Ngarenanyuki ($p>0.05$, $t=0.91$, $n=15$) and Simba ($p>0.05$, $t=0.26$, $n=13$) Rivers.

Fluoride

Fluoride concentration varied in parallel with other physicochemical parameters, in particular salinity ($R=0.54$, $p<0.05$, $n=20$). At the water extraction sites within the National Parks, fluoride values ranged from less than 0.3 to 9 mg/l. KINAPA had the lowest fluoride concentration with values not exceeding 0.3 mg/l. The concentration of fluoride was also significantly ($p<0.001$, $Z=5.66$, $n=35$) higher in the areas outside the National Parks. Fluoride concentration increased downstream in the Simba and Ngarenanyuki Rivers with values ranging from 0.2 to ~30 mg/l. The Ngarenanyuki River had far higher levels of fluoride reaching almost 30 mg/l in the dry season, whereas Simba River measured between 0.22 to 0.48 mg/l. The waterholes contained larger amounts of fluoride than the rivers during the dry season, ranging from 10 to almost 40 mg/l in Sinya (Site H3) and Ngereiyani waterhole (Site H1) respectively.

Nutrients

Most water sources measured significantly ($p<0.001$, $t=5.95$, $n=36$) higher concentration of nitrate than phosphate. The concentrations of both nitrate and phosphate were also higher in the water holes than in the rest of water sources in the landscape. In both parks, nitrate concentration ranged from 0.4 mg/l to 9 mg/l, while the maximum phosphate was 3 mg/l at Narok (A13), in ANAPA. With the exception of Sambasha (A18) in ANAPA, which measured a nitrate of almost 9 mg/l, the level of nutrients in the extraction sites within the parks was comparatively low compared to running water sites outside the parks such as the Simba and

Ngarenanyuki Rivers. Dry season nitrate concentration along both rivers ranged from almost 2 mg/l to about 4 mg/l and phosphate from 0.13 mg/l to 1.58 mg/l. While water sources outside the parks recorded higher dry season nitrate concentration, an opposite situation was observed downstream the Simba River which recorded higher wet season concentration ranging from 1.44 mg/l to almost 4 mg/l. Nitrate ranged from 7mg/l at Sinya water hole (H3) to almost 480 mg/l at Ngereiyani water hole (H1), and Phosphate from 1 to almost 6 mg/l respectively.

Heavy metals

Heavy metals (Al, Mn, Fe, Cu, Zn, As, Cd and Pb) concentrations varied both temporally and spatially across the landscape. As would be expected given their higher concentrations in the lithosphere, aluminium, iron and manganese recorded the highest values. Iron and aluminium concentrations significantly declined during the wet season ($p<0.001$, $z=-9.83$, $n=56$ and $p<0.001$, $z=-14.65$, $n=56$ resppectively), and their concentrations were significantly ($p<0.001$, $z=13.23$, $n=56$ and $p<0.001$, $z=13.91$, $n=56$ respectively) higher outside than inside the parks. Iron and aluminium increased during the dry season with the highest values of 3 mg/l and 2 mg/l respectively in KINAPA. Generally, the level of heavy metals in most of the upstream of water extraction sites in the National Parks were within recommended guideline values for both animal and human consumption. Cu, Zn, As, Cd and Pb concentrations increased during the dry season. Water sources in the semi-arid areas, i.e. water holes and the downstream reach of the Simba River, recorded relatively high metal concentrations compared to the values in the National Parks, and some of these were above recommended limits for human and livestock use. On average, metal concentration in the downstream Simba River was almost 20 times larger than concentration recorded in the upstream site S1 which is located 20-30 km from the downstream sites. Aluminium and iron had relatively higher values and their concentrations increased downstream the river with values ranging from 10 mg/l and increasing to almost 40 mg/l and 37 mg/l respectively, which implies almost 70 and 66 times greater downstream concentration than upstream. Water holes contained the highest metal concentration of any sites, e.g. aluminium and iron were at least thrice the maximum values measured in the Simba River. The Ngereiyani-madukani (H1) and Ngainyamo (H5) water holes had the highest aluminium concentration recording almost 200 mg/l and 150 mg/l respectively during the dry season.

Water hardness

Most of surface water in the landscape is considered soft water due to the low levels of calcium and magnesium, which on equivalency is less than 60 mg/l of calcium carbonate (WHO, 2011). In most sources, calcium and magnesium concentrations were less than 32 mg/l, except in a few water holes whose dry season concentration were above 60 mg/l, reaching a maximum of nearly 70 mg/l. In both wet and dry season, calcium concentration was significantly higher ($p<0.01$, $t=3.22$, $n=56$), than magnesium concentration in the Kilimanjaro landscape. In addition, calcium and magnesium concentrations were significantly higher ($p<0.001$, $z=13.62$, $n=56$ and $p<0.001$, $z=7.96$, $n=56$ respectively) outside than inside the parks. Calcium and magnesium also decreased significantly in wet season ($p<0.001$, $z=-6.69$, $n=56$ and $p<0.01$, $z=-2.67$, $n=56$ respectively). Water holes in the semi-arid areas measured the highest total water hardness and like the heavy metals, showed a similar spatial and temporal pattern.

3.3.2 Spatial and temporal changes in water quality parameters

DO, Temperature, pH and Salinity in flowing waters

Figure 3.3, shows the level of DO in the extracted water sources between upstream and downstream of extractions. DO varied across sites within ANAPA, ranging from as low as 3 mg/l (41.3 % saturation) at Ngongongare 1 (Site A1) which was the only site with DO concentration below 5 mg/l (54 % saturation), to almost 8 mg/l (110 % saturation) at Kira hill (Site A9). These results align well with the past findings reported by Elisa et al. (2016), whereby DO concentration in the park ranged between 5 and 11 mg/l (54% and 113% saturation respectively) and showed no consistent spatial gradients. Overall, there was no significant difference ($p>0.05$, $t=0.87$, $n=25$) in DO between up- and downstream sites of the same watercourse and none of the recorded differences exceeded 2 mg/l in ANAPA and KINAPA. However, Ngongongare 1 in ANAPA (Site A1 in Figure 3.2A) showed a significant difference ($p<0.001$, $t=-20$, $n=6$) in DO between up-and downstream during the dry season.

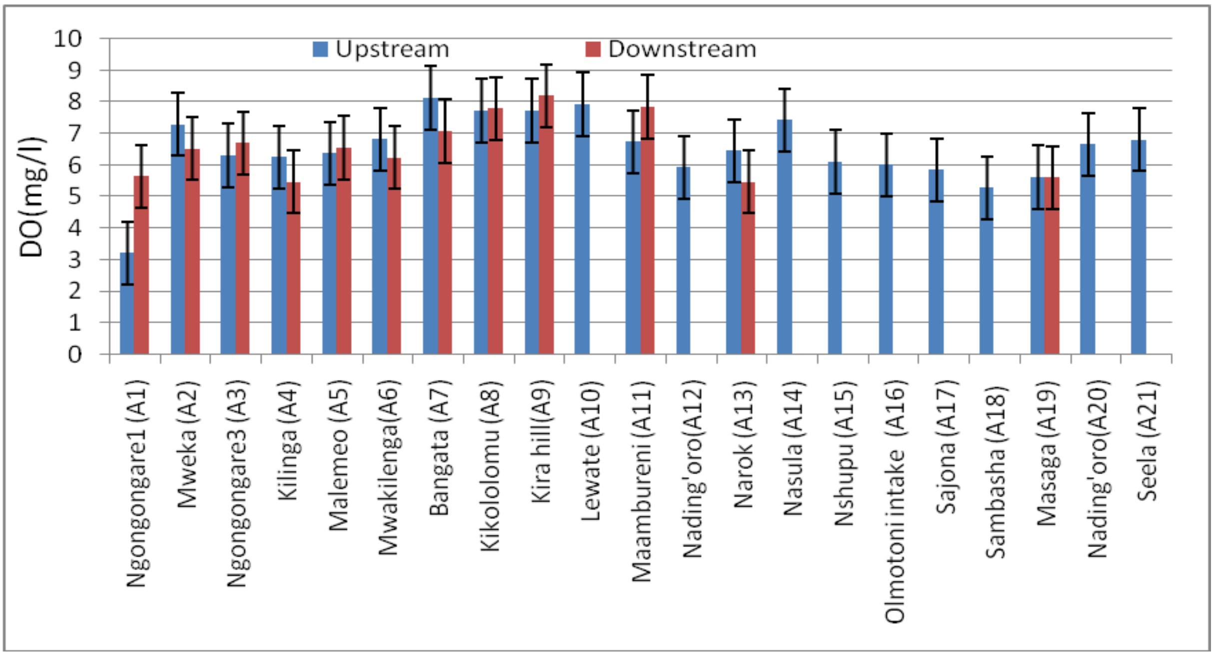

Figure 3. 3: Mean (±SE, n=4) dissolved oxygen (DO) in the extracted sites up- and downstream in ANAPA during the dry season.

In most cases, water temperature was slightly higher in the downstream sites than in the upstream in both ANAPA and KINAPA (Figure 3.4); however, the difference in both areas was small. Some of the DOs declined downstream, with the increasing temperature in both parks. Water temperature was slightly higher in the ANAPA sources, ranging between 12°C and 26°C whereas KINAPA recorded values between 12 °C and 16 °C. Upstream, Lerangwa (Site K3) in KINAPA recorded the highest DO of about 7.5 mg/l (98.9 % saturation) and lowest average temperature of about 12˚C. Downstream Kitendeni (Site K2) had the highest temperature of 16 ˚C although the lowest DO concentration (6 mg/l) recorded at this site was not due to temperature as represents 78 % saturation.

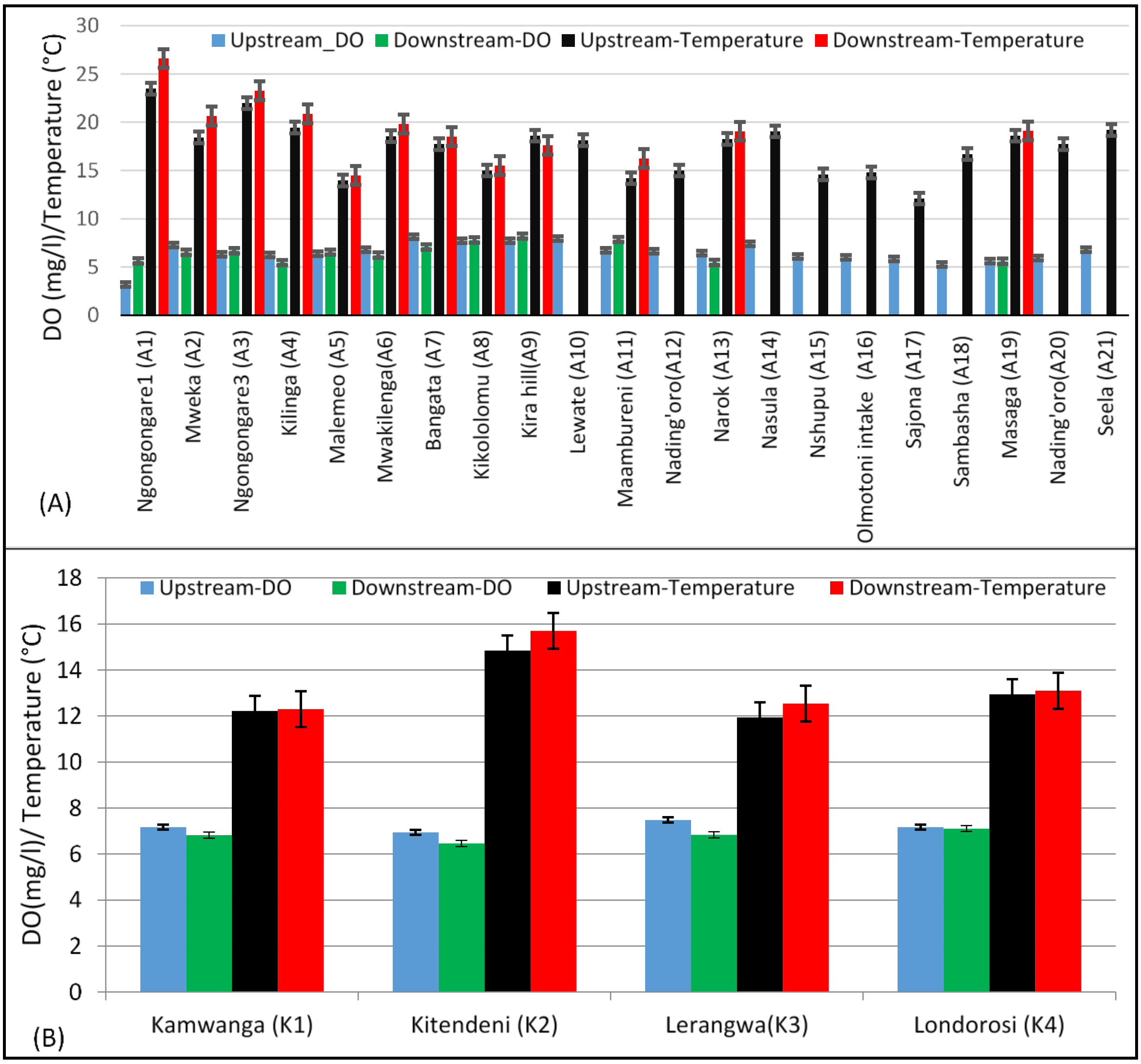

Figure 3. 4: Mean DO concentration and water temperature in sites up-and downstream during the dry season in (A) ANAPA (±SE, n=4), and (B) KINAPA (±SE, n=7).

Generally, the pH declined downstream as evidenced in most of the sites where there was not complete water abstraction in both ANAPA and KINAPA (Figure 3.5), however a t-test showed no significant difference ($p>0.05$, $t=1.7$, $n=25$) in pH between up-and downstream sites.

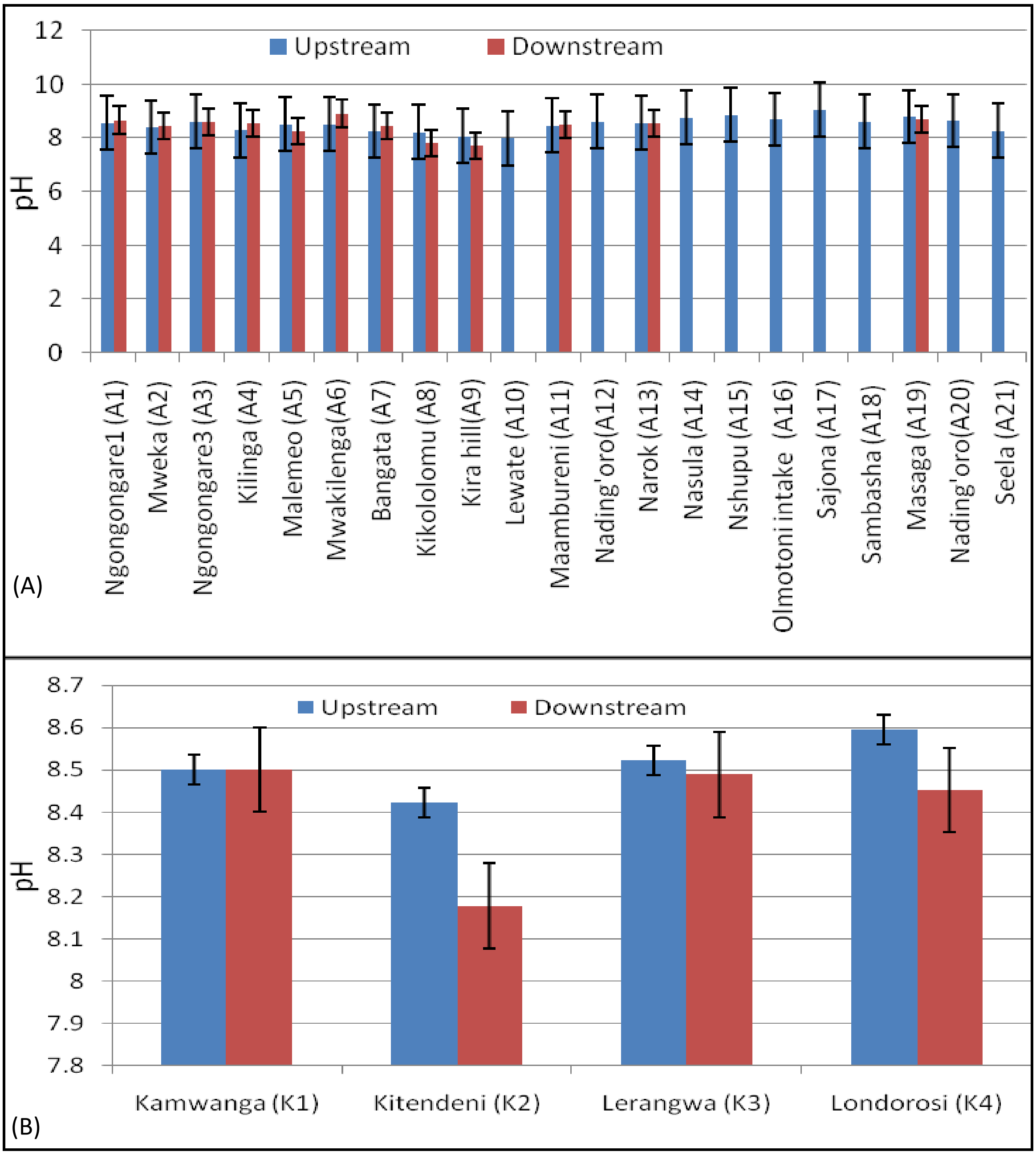

Figure 3. 5: (A) Mean pH up- and downstream of abstracted sites during dry season in (A) ANAPA (±SE, n=4) , and (B) KINAPA (±SE, n=7).

Figure 3.6 shows the average salinity concentration upstream and downstream of extraction sites in ANAPA and KINAPA. There were apparent variations in salinity concentration, though not significant ($p>0.05$, $t=0.76$, $n=25$), between upstream and downstream of the extraction sites within the same watercourse in both ANAPA and KINAPA. The difference in salinity among different extracted sources in ANAPA was relatively high, with sources located to the east as well as those located in the west, recording relatively higher concentration values than sites in the central areas of the park.

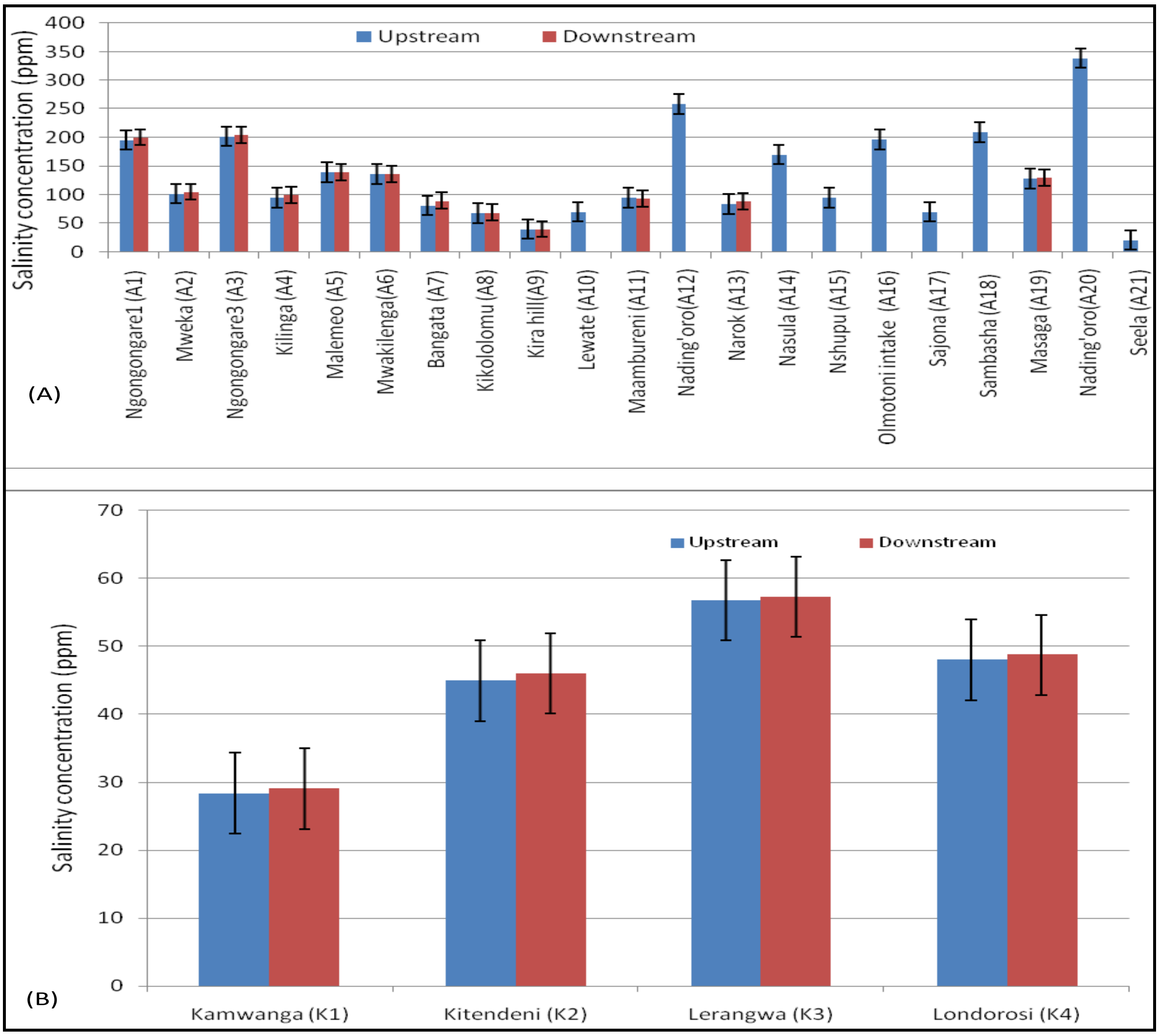

Figure 3. 6: Mean salinity concentration between up-and downstream of abstracted sites during dry season in (A) ANAPA (±SE, n=4), and (B) KINAPA (±SE, n=7).

Table 3.1 shows the average values and respective coefficient of variations for various water physicochemical parameters as measured upstream and downstream along the Simba and Ngarenanyuki Rivers during the dry and wet seasons. Spatial and temporal variations were apparent for all measured parameters. In general, salinity and water temperature increased downstream during the dry season whereas DO and pH decreased. There was a significant difference in dry season salinity between upstream (Site S1) and downstream site (Site S4) in the Simba River ($p < 0.01$, t=7.86, n=6). However, such difference was not observed in the Ngarenanyuki River for the corresponding up-and downstream sites (Sites N1 and N2).

Instead, upstream salinity and the DO concentrations in the Ngarenanyuki River, each showed a significant difference ($p < 0.05$, $t=2.97$, $n=7$ and $p<0.05$, $t=3.18$, $n=7$ respectively) between dry and wet season. Compared to other parameters, pH was relatively stable with a coefficient of variation of less than 4 %. Generally, in the dry season, there was no water flowing into the downstream areas beyond Ngabobo (Site N2) and Ndarakwai (Site S4) in the Ngarenanyuki and Simba Rivers respectively, and as such water quality assessment could not be conducted further downstream.

Table 3. 1: Physicochemical characteristics of the water in the Simba and Ngarenanyuki Rivers.

Simba River						
	Season	Stat	DO (mg/l)	Temp (˚C)	pH	Salinity (ppm)
Upstream	Dry	Average	6.95 ± 0.48	14.62 ± 0.32	8.54 ± 0.07	29 ± 0.75
		CV (%)	20.59	6.52	2.3	7.8
	Wet	Average	8.36 ± 0.60	13.63±0.34	8.49±0.07	22.03±1.71
		CV (%)	20.45	7.11	2.31	21.93
Ndarakwai	Dry	Average	5.32 ± 0.51	23.64±0.21	8.51±0.05	57.24±3.51
		CV (%)	23.26	2.18	1.56	15.02
	Wet	Average	9.19 ± 0.49	18.41±0.56	8.48±0.11	45.86±9.24
		CV (%)	15.22	8.6	3.59	56.99
Tingatinga	Wet	Average	8.45±0.75	26.83±1.19	8.52±0.03	55.97±2.03
		CV (%)	8.84	7.66	0.65	6.29
WMA	Wet	Average	0.62±0.36	26±2.10	8.53±0.06	61.2±1.68
		CV (%)	7.8	13.99	1.29	4.76
Ngarenanyuki River						
	Season	Stat	DO (mg/l)	Temp (˚C)	pH	Salinity(ppm)
Upstream	Dry	Average	7.24±0.39	20.91±0.75	9.79±0.05	1037.60±26.91
		CV(%)	17.02	11.3	1.68	8.2
	Wet	Average	9.05±0.27	19.25±0.48	9.38±0.10	806.86±46.70
		CV(%)	7.84	6.55	2.9	15.31
Ngabobo	Dry	Average	7.17±0.35	23.11±1.01	9.67±0.06	1079.80±25.31
		CV(%)	15.49	13.87	2.09	7.41
	Wet	Average	8.69±0.20	22.53±0.66	9.41±0.10	944.71±42.31
		CV(%)	6.03	7.77	2.88	11.85
Madebe	Wet	Average	8.51±0.63	23.67±2.5	9.37±0.07	957.67±37.40
		CV(%)	12.77	18.32	1.21	6.76

As shown in the Table 3.1 above, during the wet season, DO concentration in the Simba and Ngarenanyuki Rivers, remained nearly steady from up- to downstream with a range from 8 mg/l on the Simba River entry to the WMA (Site S6) to 10 mg/l on the Ngarenanyuki River at Madebe (Site N3). Water temperature increased almost linearly with increasing distance downstream and the values ranged from 14°C to 26 °C.

Likewise, in the rainy period, salinity concentration increased downstream where its values from up- to downstream ranged from almost 20 ppm to about 60 ppm in the Simba River and from 800 ppm to almost 1000 ppm in the Ngarenanyuki River. While both rivers experienced temporal and spatial variations, the Simba River recorded comparatively very low levels of salinity where the maximum average value was about 60 ppm downstream at the entry to WMA (Site S6). Upstream the Simba River (Site S1 in Figure 3.1) was consistently characterised by low salinity, and differed significantly ($p < 0.05$, $t=2.54$, $n=8$) from the high values measured in the downstream areas (e.g. Site S4). Similarly, the Ngarenanyuki River also measured significantly ($p<0.01$, $t=6.81$, $n=4$) higher salinity concentration at the downstream (Site N3) than at the upstream (Site N1). Discharge declined significantly downstream in both rivers (see Chapter 2). For instance, the Simba River discharge was significantly ($p<0.01$, $t=5.65$, $n=17$) lower in the downstream areas (Site S4) than in the upstream (Site S1) in both dry and wet seasons. Likewise, the Ngarenanyuki River discharge was significantly ($p<0.01$, $t=17.9$, $n=9$) lower in the downstream areas (Site N3) than in the upstream (Site N1) in both dry and wet seasons.

In the Simba and Ngarenanyuki Rivers, the salinity at the upstream sites S1 and N1 varied negatively to the river discharge (Figure 3.7A and B). The discharge and salinity concentration in the Ngarenanyuki and Simba Rivers were strongly negatively correlated at $R=-0.75$, $p<0.01$ and $R=-0.65$, $p<0.01$ respectively. Salinity concentration at the upstream sites decreased with the increase in river discharge, and both parameters appeared to be under the influence of rainfall patterns. Further, compared to wet season, salinity concentration in the upstream for Ngarenanyuki and Simba Rivers increased in the dry season respectively by almost 30% and 50%.

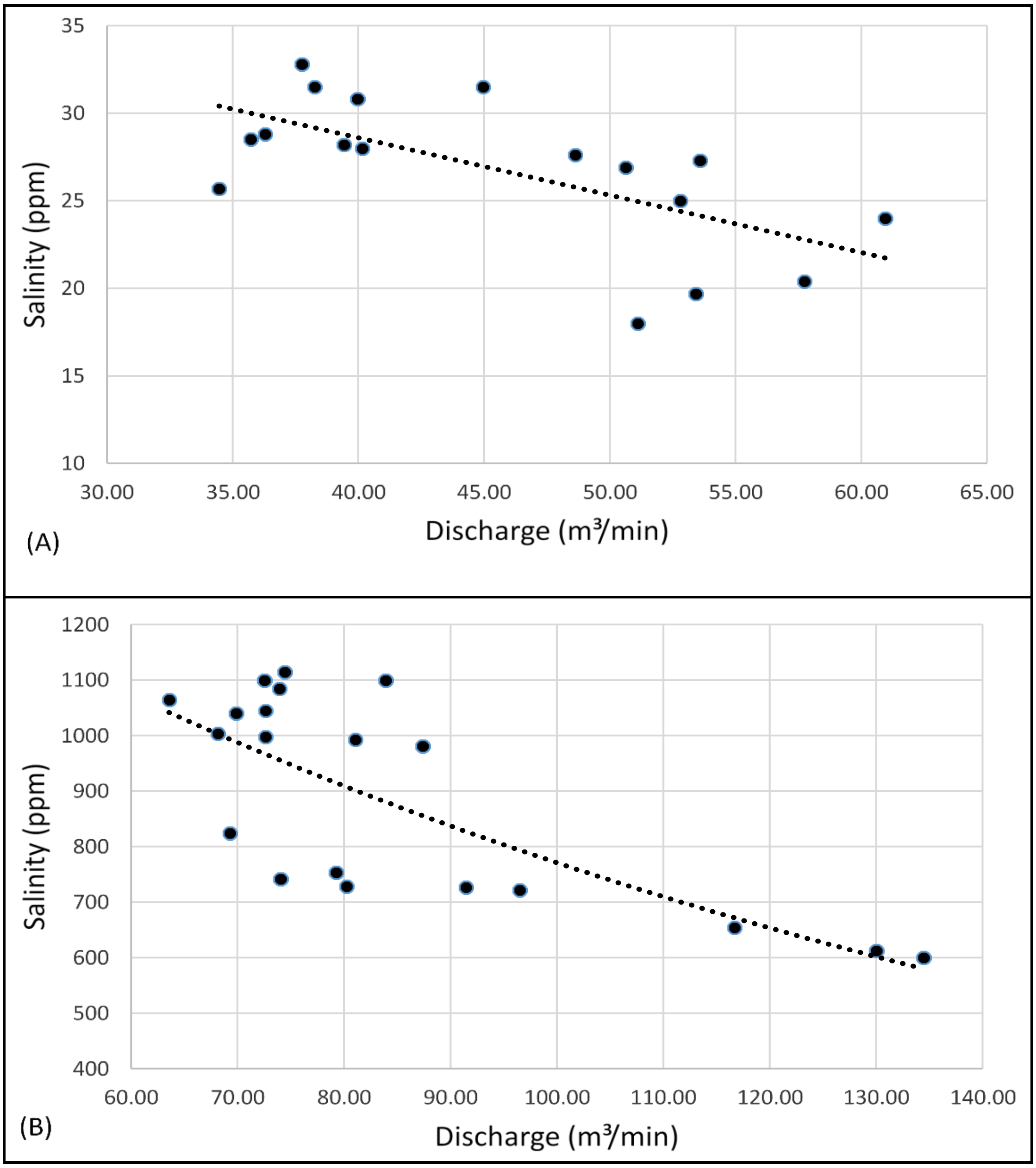

Figure 3. 7: Scatter plot of discharge and salinity concentration in the upstream (A) Simba River and (B) Ngarenanyuki River from September 2018 to May 2020.

Values for several water quality parameters showed a likely human impact as reflected in the accumulation factor (AF) (which may also be referred as human impact factor, e.g.from agricultural effluent) as the Simba and Ngarenanyuki Rivers flowed downstream (Figure 3.8). In the dry season, the AF values for water temperature and salinty were about 1.6 and 1.9 respectively for Simba River at Ndarakwai. Simba River recorded higher AF values with salinity reaching almost 2, whereas the Ngarenanyuki River AF values were generally about 1. AF increased during the rainy period as indicated by relatively high values of water temperature, and salinity in the Simba River, and such values showed a spatial pattern by increasing downstream by almost 2 and 2.5 respectively.

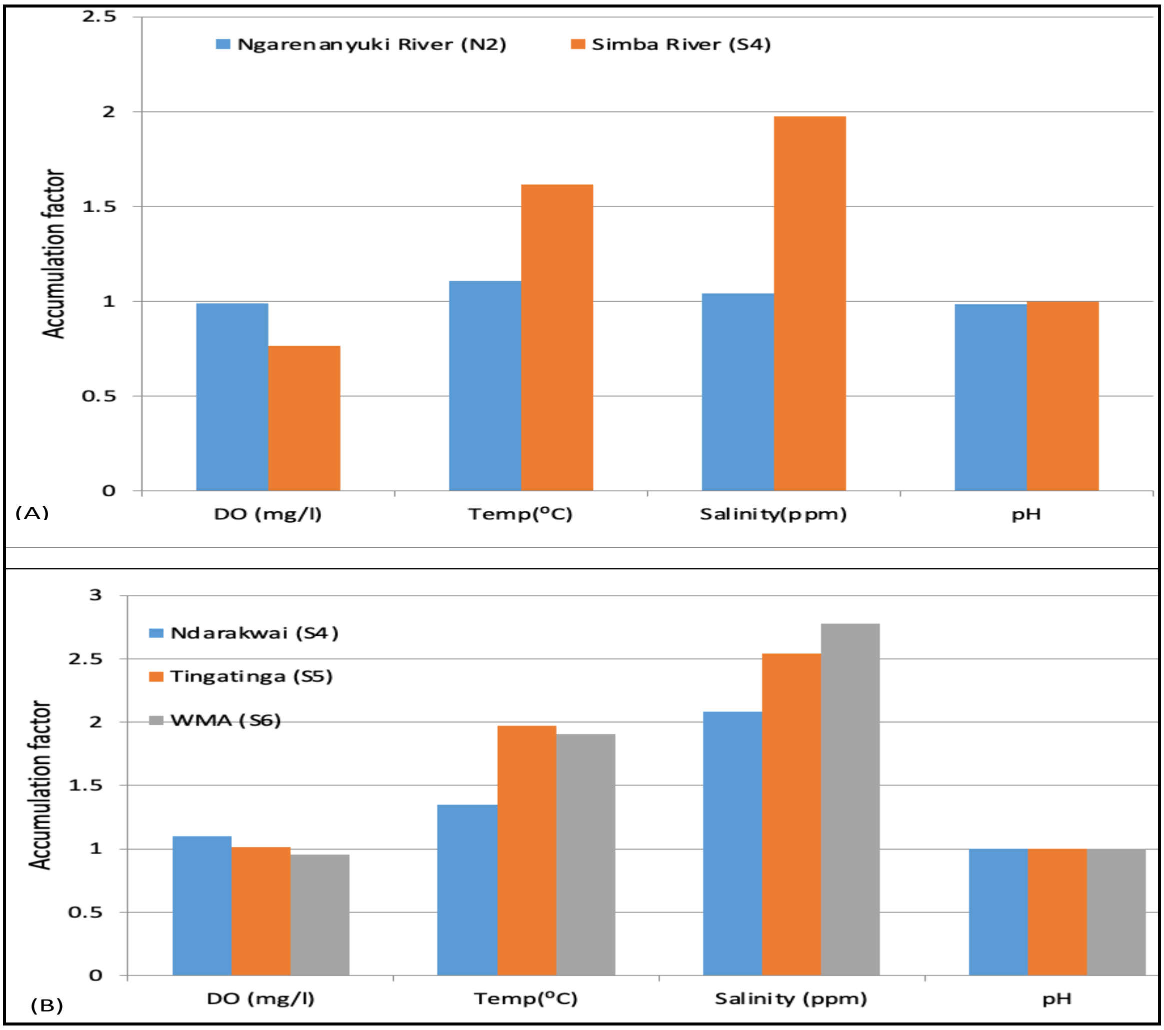

Figure 3. 8: Accumulation factor for physicochemical parameters downstream in (A) Simba and Ngarenanyuki Rivers respectively at Ndarakwai and Ngabobo in dry season, (B) Simba River in the downstream areas during the rainy season.

DO, Temperature, pH and Salinity in standing waters

Table 3.2 shows that there were variations in the values of various physicochemical paramaters, particularly DO and salinity, across water holes and between the dry and wet seasons. During the dry season, all water holes (Sites H1, H2 and H3) that were relatively more accessible to wildlife in the lowland semi-arid areas recorded low DO concentration (< 5 mg/l). The Ngereiyani water holes, particularly site H1, recorded the lowest average DO value of 2.2 mg/l, followed by site H2 with 4.5 mg/l, and then site H3 with a mean DO of 4.9 mg/l. Salinity levels were also higher in the dry season, with the salinity at the Sinya water hole (Site H3) recording the highest average of 3773.6 ppm. The pH was less variable and generally more alkaline than the rest of the sites in the landscape with values between 9 and 10.

Table 3 2: Mean values of various physicochemical parameters in the dry (±SE, n=8) and rainy season (±SE, n=5) in the Sinya (H3) and Ngereiyani (H1 and H2) waterholes.

Dry season				
Water source	DO (mg/l)	Temp (°C)	pH	Salinity(ppm)
Ngereiyani (H1)	2.20	27.86	8.91	1678.00
Ngereiyani (H2)	4.54	24.78	9.22	1292.00
Sinya (H3)	4.87	25.16	9.81	3773.63
Wet season				
Ngereiyani (H1)	8.24	24.13	9.30	1016.25
Ngereiyani (H2)	6.44	24.57	9.11	869.67
Sinya (H3)	10.06	29.42	9.45	529.80

Table 3.3 shows results of a single sampling in other temporary and permanent surface water bodies during the dry season. The values of various physicochemical parameters were within the range of values measured in other sites within the landscape, and thus deemed representative. Madebe water hole recorded the highest salinity concetration of almost 1500 ppm. However, comparing the values between these other water bodies and the monitoring sites, it is apparent that the most saline (> 3000 ppm) water in the landscape is the Sinya water hole (H3 in Table3.2) located in the abandoned Sinya mining area within the Enduimet wildlife management area. The Sinya water hole (H3) and the adjacent Sinya water hole (H3B) in Table 3.3 are different year round water holes, which however are both situated in Sinya area within EWMA. In general, the semi-arid standing waters (Table 3.2 and 3.3) were the most alkaline in the Kilimanjaro landscape with pH values ranging from almost 9 to 10.

Table 3. 3: Single dry season sampling value for other surface water bodies in the Kilimanjaro landscape.

Name of water body	Location	DO (mg/l)	Temp (°C)	pH	Salinity(ppm)
Lake Chala	1021377.29 9633083.46	5.64	29.7	9.02	177
Lake Jipe	1030982.11 9599092.79	6.72	29.2	9.62	483
Sinya water hole (H3B)	949497.35 9698750.41	5.35	26.6	9.69	1260
Tingatinga water hole	272802.91 9673008.70	7.82	30.8	9	50.7
Madebe water hole	929697.34 9662732.97	5.64	23.8	10.05	1490
Ndarakwai water hole	944560.26 9666569.86	15.7	22.5	8.7	188
Tingatinga trough	272802.98 9673008.77	2.25	19.2	8.83	54.5
Kitendeni trough	306332 9684196	10.5	21.8	9.54	52.8
Kitendeni trough 2	306475 9686402	7.1	21.4	8.5	51.3

Both the salinity and the water volume varied markedly in time (Figure 3.9A), and in opposite directions (Figure 3.9B) because of dilution mainly by rainwater and resultant surface run-off, at the Sinya water hole (Site H3 in Figure 3.1). The Sinya water hole (H3) was not supplied by a river but by surface run-off, and hence it clearly reflected changes in rainfall. The highest salinity value of more than 7500 ppm was reached at the Sinya water hole (H3) in April 2019 during the dry season when water volume was 13 % of the maximum, and the lowest values of about 200 ppm was reached in January 2020 during the rainy season.

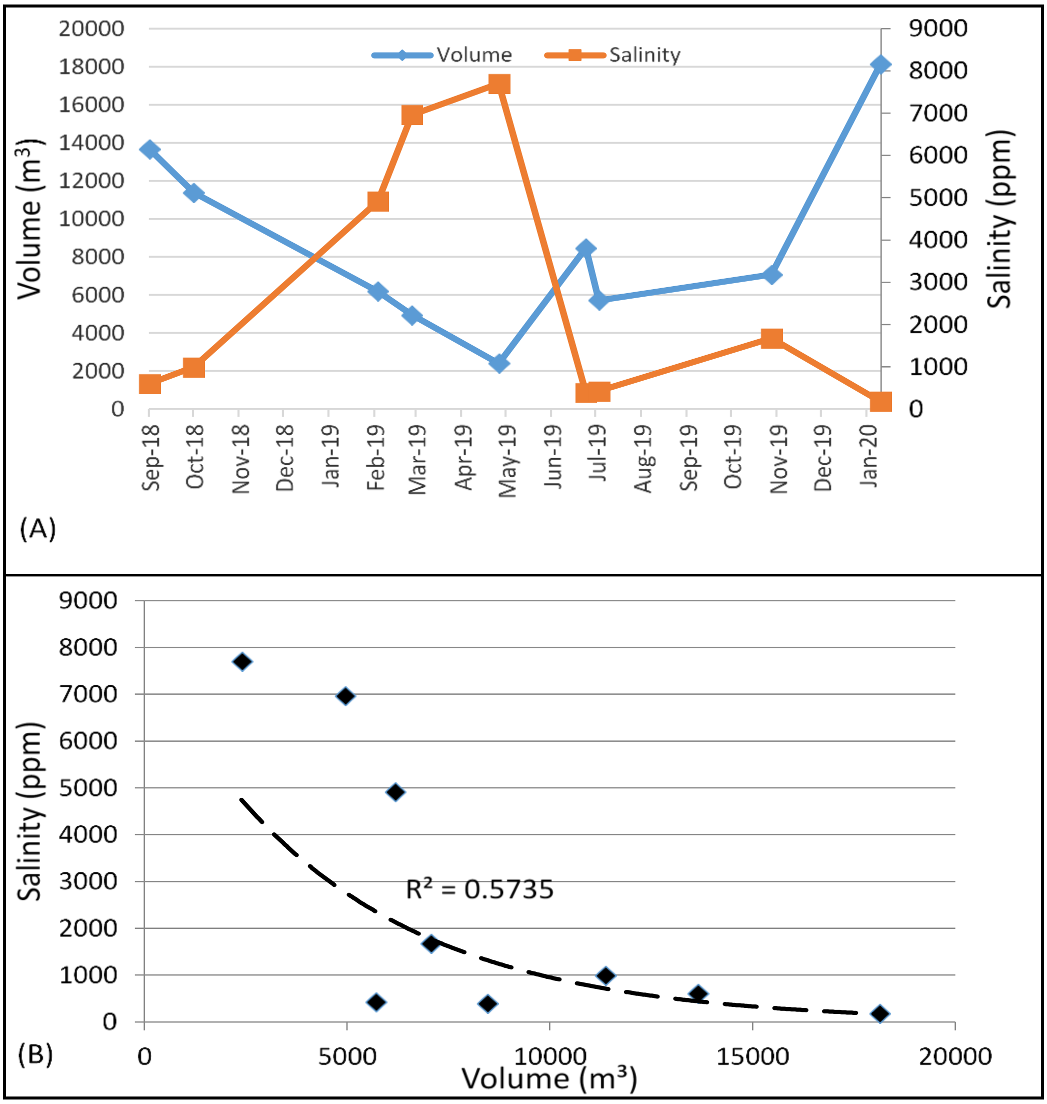

Figure 3. 9 : (A) Time series plot showing changes in salinity with volume and (B) Scatter plot of salinity vs water volume in the Sinya water hole (H3) in the Enduimet WMA.

Fluoride and Nutrients

Like others physicochemical parameters, fluoride varied across space and time in the Kilimananjaro landscape (Figure 3.10). The fluoride concentration in ANAPA had highest values occurring in water bodies in the eastern and western parts as compared to the central areas of the park.

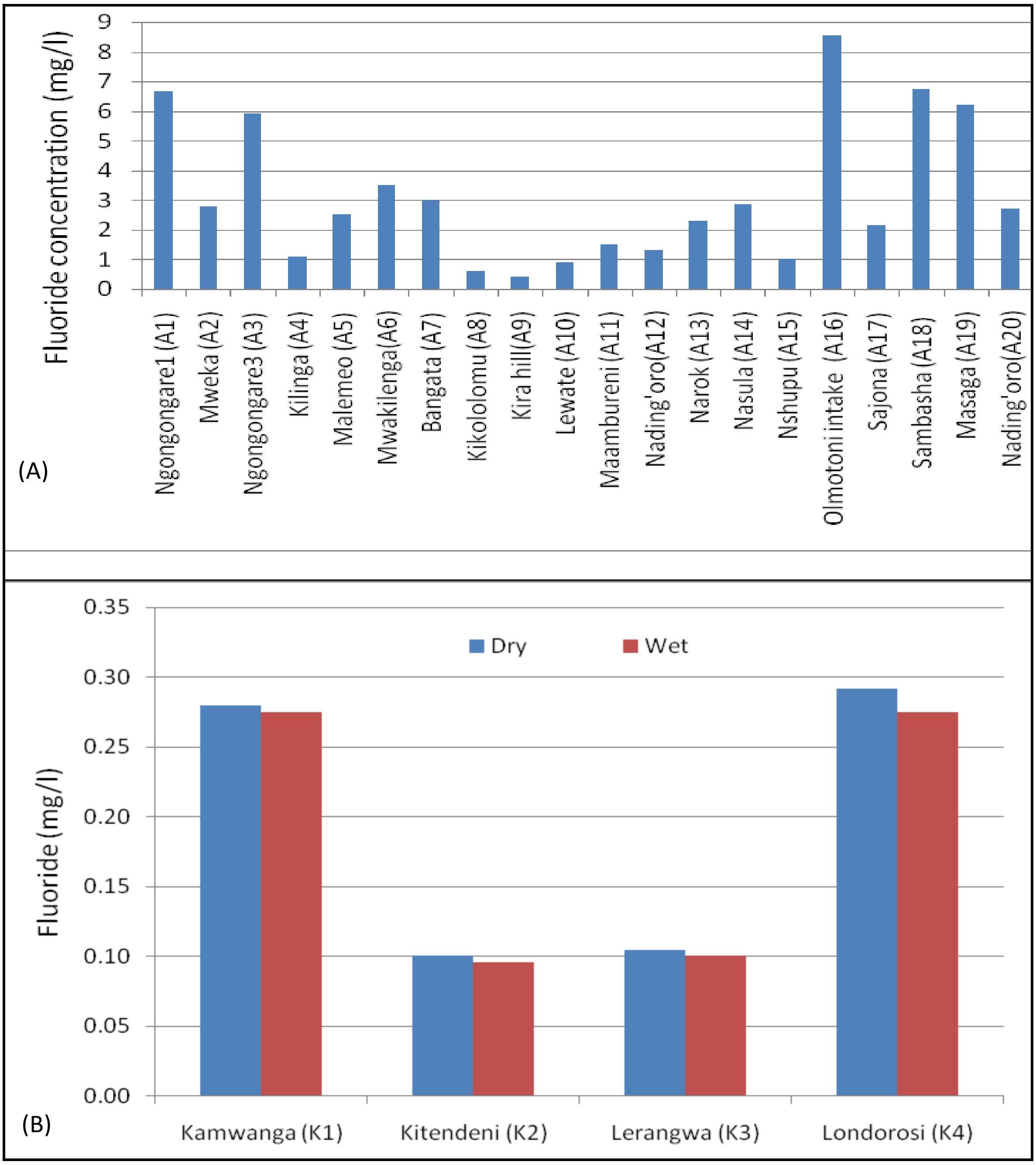

Figure 3. 10: Dry season fluoride concentration in the extracted sites in (A) ANAPA and (B) dry and wet season fluoride concentrations at extracted sites in KINAPA.

The concentration of nitrate was significantly higher ($p<0.05$, $t=9.49$, $n=24$) than that of phosphate in all of the water extraction sites in the two National Parks (Figure 3.11). All sites in KINAPA except Kamwanga (K1), had higher nitrate in the wet season. In KINAPA, nitrate ranged from almost 1.3 mg/l at Kamwanga intake (K1) to more than 5 mg/l at Lerangwa (K3). Nitrate levels in ANAPA sources ranged from almost 0.4 mg/l at Nading'oro

(A12) to about 9 mg/l at Sambasha (A18). Maximum recorded phosphate in ANAPA was about 3 mg/l at Narok (A13), whereas the maximum phosphate in KINAPA was 0.5 mg/l at Londorosi (K4).

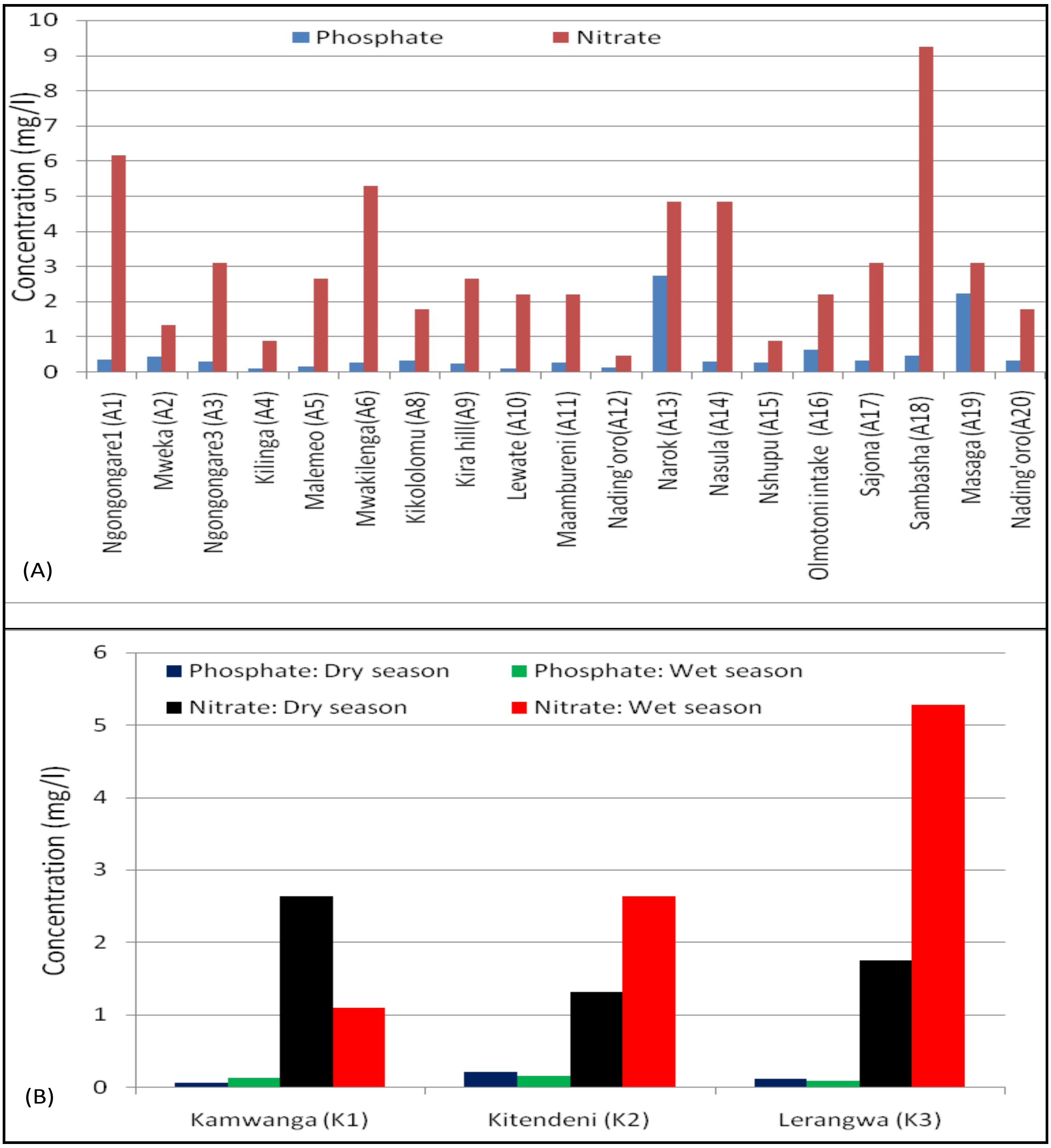

Figure 3. 11: Concentration of nutrients in the extracted sites in (A) ANAPA during dry season, (B) KINAPA in dry and wet season.

Results from generalised linear model indicated variations in fluoride and nutrient concntrations across the landscape. The model took into account some of the factors that might influence the surface water concentration of fluoride, nitrate and phosphate. The factors considered were type of water source, location of water source as either inside or outside the park and season, which included both dry and wet season. The findings (Table 3.4) show that, fluoride concentration significantly ($p<0.001$) declined in the wet compared to the dry season, and measured higher ($p<0.001$) values in the areas outside the parks especially in the water holes in the lowland semi-arid areas. Likewise, nitrate and phosphate concentrations were respectively higher ($p<0.01$ and $p<0.05$) in the water holes (Table 3.4). Note that other factors such as geology/soil, might also affect the concentrations of fluoride and the nutrients in surface water but could not be examined.

Table 3. 4: Generalised linear model outputs on fluoride and nutrient concentration in the Kilimanjaro landscape.

Fluoride				
Factor	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	1.59	0.11	13.95	< 2e-16
Source: Water hole	0.56	0.17	3.35	0.000811
Season: Wet season	-0.52	0.13	-4.05	5.11E-05
Location: Area outside the park	0.96	0.17	5.66	1.53E-08
Nitrate				
(Intercept)	1.26	0.14	8.93	< 2e-16
Source: Water hole	0.95	0.33	2.90	0.00368
Season: Wet season	-0.33	0.19	-1.77	0.07608
Location: Area outside the park	-0.29	0.30	-0.94	0.34629
Phosphate				
(Intercept)	-1.24	0.46	-2.71	0.00668
Source: Water hole	1.33	0.66	2.01	0.04465
Season: Wet season	-0.47	0.46	-1.03	0.30166
Location: Area outside the park	0.79	0.73	1.09	0.27345

An increase in fluoride concentration downstream was recorded in the Ngarenanyuki River from almost 29 mg/l at the upstream site N1 to 30.35 mg/l downstream at Ngabobo (N2) (Table 3.5). Fluoride concentration in the Simba River also increased downstream, from 0.22 mg/l at upstream site S1 to almost 0.5 mg/l in the downstream site S4 at Ndarakwai wildlife ranch. The Ngarenanyuki River-fed water hole in Ngereiyani (Site H1) recorded the highest fluoride level of 36.5 mgl/l during the dry season. The Sinya water hole (Site H3 in Figure 3.1) in the Enduimet WMA recorded a fluoride level of 10 mg/l. In comparison and as was the case with salinity levels; the Simba River recorded the lowest fluoride concentration while the Ngerenanyuki River system recorded the highest concentration.

Table 3. 5: Fluoride and nutrient concentration in the water holes, the Ngarenanyuki and Simba Rivers.

Ngarenanyuki River						
Water source	Dry season			Wet season		
	Phosphate (mg/l)	Fluoride (mg/l)	Nitrate (mg/l)	Phosphate (mg/l)	Fluoride (mg/l)	Nitrate (mg/l)
Upstream ngarenanyuki (N1)	0.28	28.59	3.96	0.32	19.87	0.88
Ngabobo (N2)	0.29	30.35	2.2	0.37	24.09	2.64
Simba River						
	Phosphate (mg/l)	Fluoride (mg/l)	Nitrate (mg/l)	Phosphate (mg/l)	Fluoride (mg/l)	Nitrate (mg/l)
Simba upstream(S1)	0.13	0.22	2.64	0.13	0.19	2.64
Ndarakwai(S2)	0.12	0.48	2.2	1.58	0.23	3.96
WMA (S6)				0.28	0.56	1.44
Enduimeti WMA						
	Phosphate (mg/l)	Fluoride (mg/l)	Nitrate (mg/l)	Phosphate (mg/l)	Fluoride (mg/l)	Nitrate (mg/l)
Sinya (H3)	1.1	10	7.04	0.52	2.22	1.23
Sinya water hole (H3B)				1.02	2.12	0.88
Ngereiyani						
	Phosphate (mg/l)	Fluoride (mg/l)	Nitrate (mg/l)	Phosphate (mg/l)	Fluoride (mg/l)	Nitrate (mg/l)
Water hole1(H1)	5.5	36.52	478	1.92	26.2	7.04
Water hole2(H2)				0.85	21.25	2.2

As is commonly the case, nitrate concentrations were consistently higher ($p<0.001$, $t=5.96$, $n=15$) than phosphate in all sites. During the dry season, the Ngereiyani water hole (Site H1 in Figure 3.1) measured up 478 mg/l, followed by 7 mg/l recorded at the Sinya water hole (Site H3 in Figure 3.1). The highest amount of phosphate was 5.5 mg/l again in the Ngereiyani waterhole (Site H1 in Figure 3.1) which received its water from the Ngarenanyuki River.

Metals

It is evident from Figure 3.12A that aluminium, iron and manganese recorded higher concentration than the other metals in the Simba River. Relatively, aluminium measured higher concentration both in the upstream and downstream areas. Within the river, iron and aluminium respectively reached a maximum concentration of almost 35 mg/l and 40 mg/l in the downstream areas. These concentrations far exceed the guideline values of water quality for livestock use, which are 0-5 mg/l and 0-10 mg/l of Al and Fe respectively (DWAF, 1996a). As shown in Figure 3.12A and B, generally all heavy metals increased downstream and measured higher values in dry season compared to wet season. Cadmium, and arsenic, each measured concentrations of less than 0.02 mg/l in both dry and wet seasons as shown in Figure 3.12C. However, while lead was well below the guideline value for livestock use, dry season concentrations were above the guideline limit (0.01 mg/l) for human drinking water downstream the Simba River (Figure 3.12C; WHO 2004). With the exception of aluminium and iron, the rest of heavy metals measured concentration less than those stipulated in the standard guideline values of water quality for livestock use.

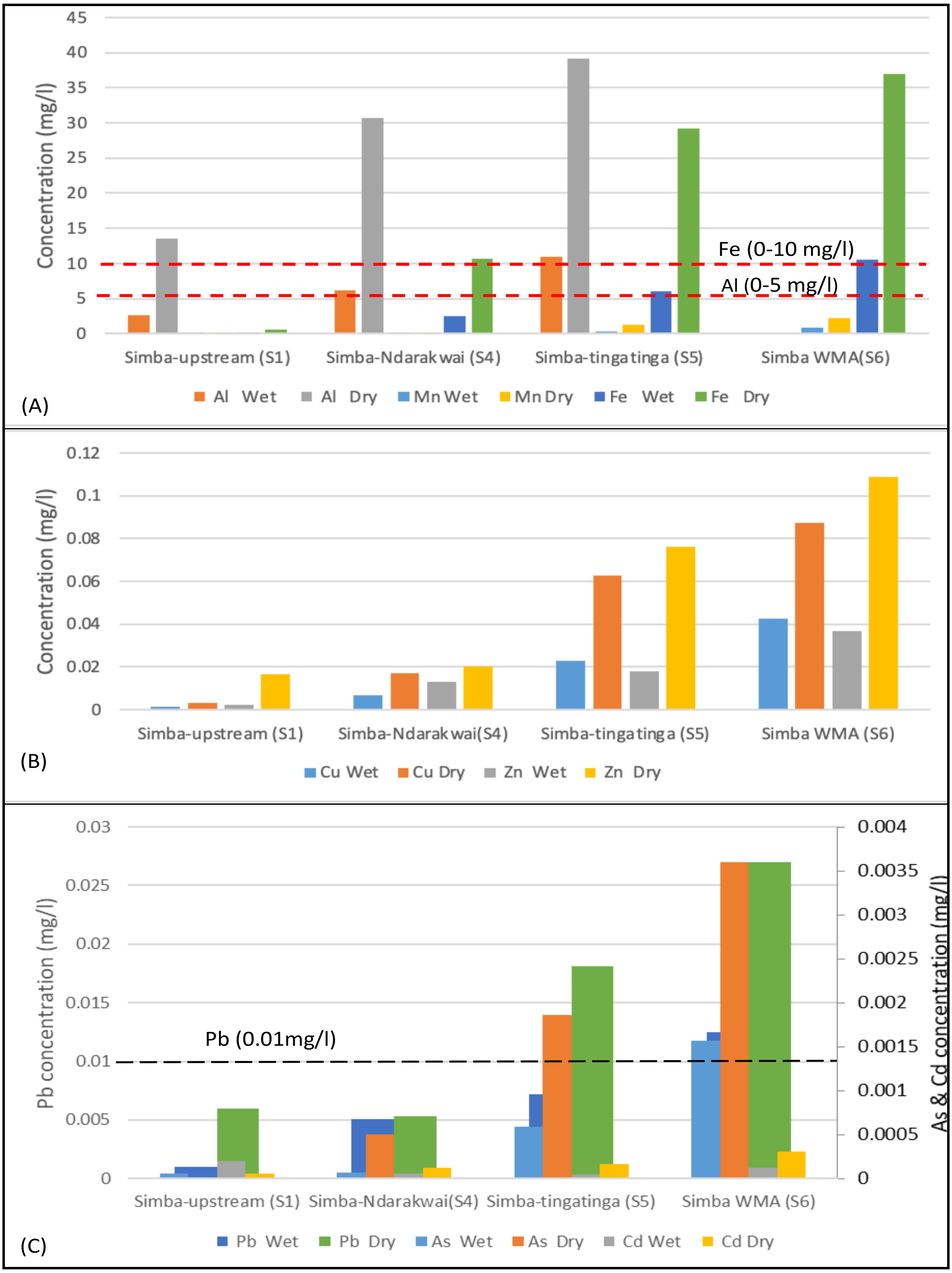

Figure 3. 12: Spatial and temporal concentration of (A) aluminium, manganese, and iron; (B) copper, zinc; (C) arsenic, cadmium, and lead in the Simba River from upstream to downstream areas. Dashed lines represent guideline values of water quality for human use (black coloured) and livestock use (red coloured). Source: (DWAF, 1996a; WHO, 2004).

As shown in Table 3.6, the concentration of most of the heavy metals increased markedly downstream as measured in the Simba River at the boundary of the Enduimet WMA (EWMA) site S6, where manganese was 134 times higher than the concentration measured upstream in KINAPA at site S1 before water extraction. Arsenic recorded a value of less than 0.0005 mg/l and 0.004 mg/l respectively in upstream and downstream.

Table 3. 6: Change in concentration of heavy metals as indicated by the Accumulation Factor (AF) and River Recovery Capacity (RRC) along the Simba River between the upstream site S1 and downstream site S6 during the dry season.

	Al	Mn	Fe	Cu	Zn	As	Cd	Pb
AF	69.56	134.25	65.91	30.05	6.52	0	5.63	4.49
RRC (%)	98.56	99.26	98.48	96.67	84.65	100	82.24	77.76

Iron and aluminium recorded the highest concentration of almost 0.5 mg/l in the upstream site of the Ngarenanyuki River (site N1), and then declined downstream during the dry season (Figure 3.13A). The concentration of zinc, arsenic and lead did not exceed 0.008 mg/l and decreased downstream (Figure 3.13B). Copper however increased downstream and attained comparatively high AF value of 1.13 (Table 3.7) but cadmium remained relatively low and stable. The Ngarenanyuki River recovery capacity was generally low for all metals and decreasing for most of the heavy metals except for copper and cadmium (Table 3.7).

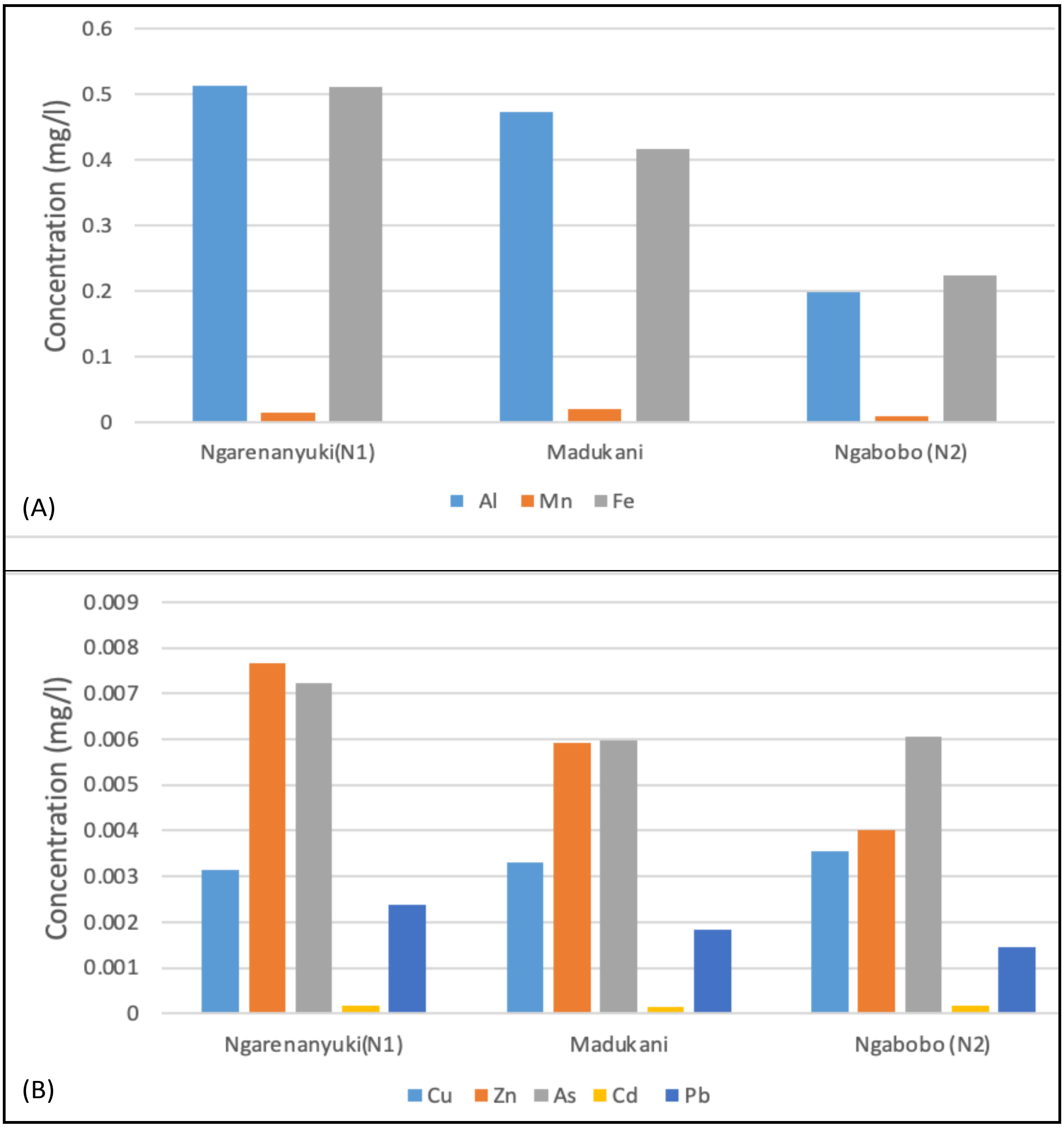

Figure 3. 13: Spatial concentration of (A) aluminium, manganese and iron; (B) copper, zinc, arsenic, cadmium and lead, during the dry season in the Ngarenanyuki River.

Table 3. 7: Change in concentration of heavy metals as indicated by the Accumulation Factor (AF) and River Recovery Capacity (RRC) along the Ngarenanyuki River between the upstream site N1 and downstream site N3 during the dry season.

	Al	Mn	Fe	Cu	Zn	As	Cd	Pb
AF	0.39	0.62	0.44	1.13	0.52	0.84	1.03	0.61
RRC (%)	-158.22	-61.02	-127.73	11.38	-90.8	-19.26	3.17	-63.55

Compared to other water sources, water holes measured relatively higher concentration of heavy metals particularly aluminium and iron, which were respectively up to 5- and almost 3-fold greater than the maximum values recorded in Simba River (Figure 3.14). These metal concentrations were far higher than the guideline values of water quality for livestock consumption. The highest concentrations were recorded during the dry season (Figure 3.14A). For instance, in the water holes, percentage change in metal concentration between wet and dry season ranged on average from 40 to 70%. Ngereiyani-madukani (Site H1) and Ngainyamo (Site H5) water holes had the highest aluminium concentration recording almost 200 mg/l and 150 mg/l respectively during the dry season. Iron concentration marked the second highest value ranging from almost 50 mg/l in Ngainyamo water hole to almost 100 mg/l at Ngereiyani-madukani water hole.

With regard to water sources within the National Parks, iron and aluminium recorded the highest dry season concentrations of almost 3 mg/l and 2 mg/l respectively in Lerangwa intake of KINAPA (Figure 3.14 B). Ngongongare 1 of ANAPA whose iron and aluminium concentrations were almost 1 mg/l followed this. While iron concentration is likely within the recommended limits (1-3 mg/l), aluminium concentration of 2 mg/l exceeds human drinking water values recommended by WHO (2010) of 0.9 mg/l, and therefore this may be a problem to human health. The extracted sites recorded dry season concentration of less than 0.015 mg/l for copper, arsenic, cadmium and lead, and a concentration of less than 0.04 mg/l for zinc. All of these metal concentrations are well below the guideline values for human drinking water as recommended by WHO (2004a; 2004b; 2003b; 2003a), which are: copper (2 mg/l) ; arsenic (0.01 mg/l); cadmium (0.003 mg/l); lead (0.01 mg/l; and Zinc (3mg/l).

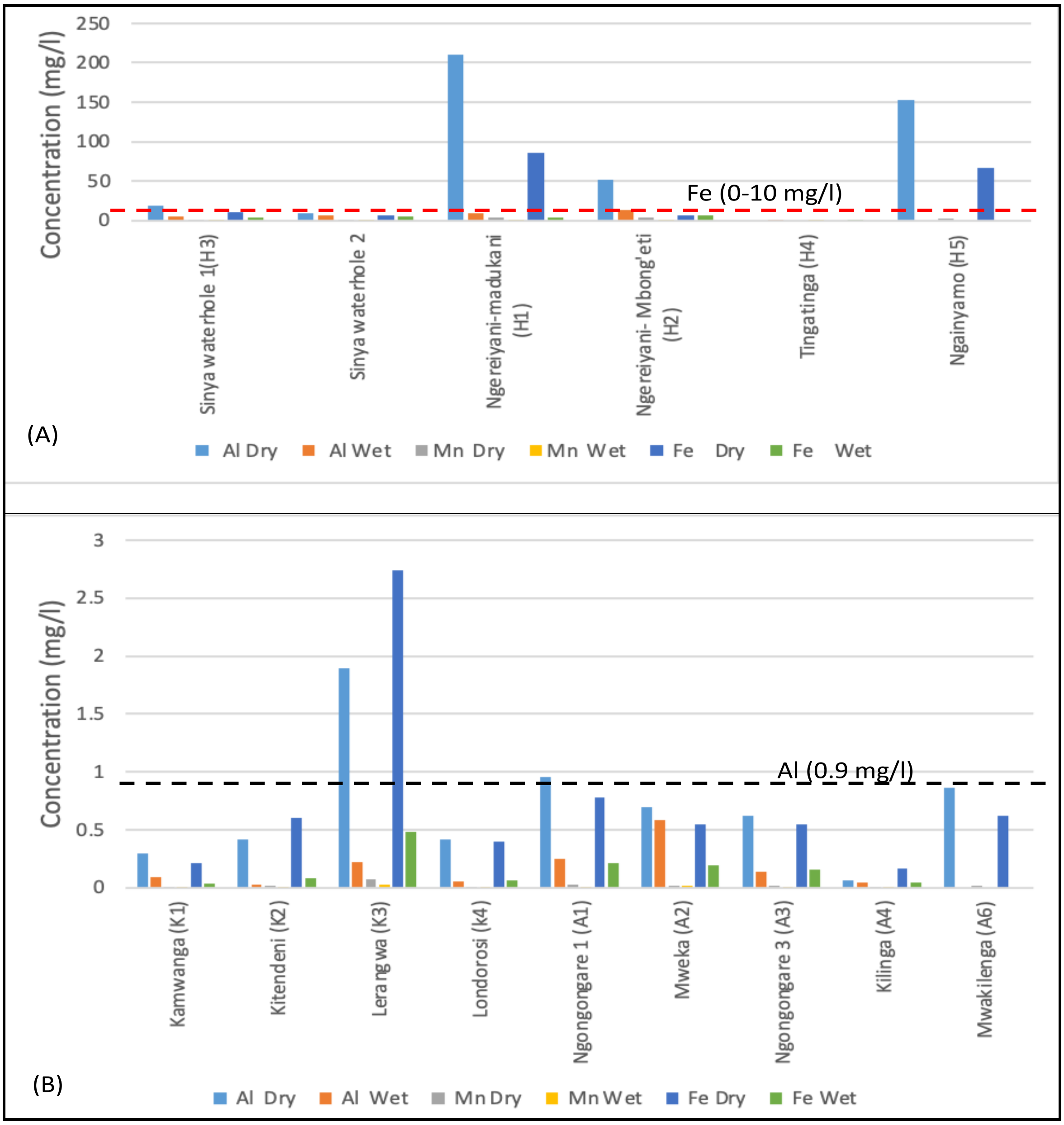

Figure 3. 14: Concentration of heavy metals (A) in water holes, (B) in water extraction sites in ANAPA and KINAPA during dry and wet seasons. Dashed lines represent standard guideline values of water quality for livestock use (red coloured line), and for human use (black coloured line). Source: (DWAF, 1996a, 1996b).

Figure 3.15 shows the spatial distribution of aluminium and lead concentration in the Kilimanjaro landscape. It is clear that areas outside of the National Parks recorded the highest aluminium and lead concentrations, especially along the Ngarenanyuki and the Simba Rivers and associated/neighbouring water holes. Indeed results from generalised linear model (Table 3.8) showed that aluminium increased significantly ($p<0.01$) in the areas outside the National Parks especially in the water holes. However, aluminium concentration declined significantly in wet season and with an increase in pH ($p<0.001$). Similar trends, were also observed with most of the other heavy metals metals, especially iron.

Table 3. 8: Generalised linear model outputs on aluminium concentration in the Kilimanjaro landscape.

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	12.8845	1.7891	7.202	5.94E-13
Area outside the park	3.3932	0.2969	11.427	< 2e-16
Wet season	-1.8821	0.1455	-12.933	< 2e-16
Source: River	0.6407	1.1128	0.576	0.56475
Source:Waterhole	3.134	1.1043	2.838	0.00454
pH	-1.6226	0.1522	-10.663	< 2e-16

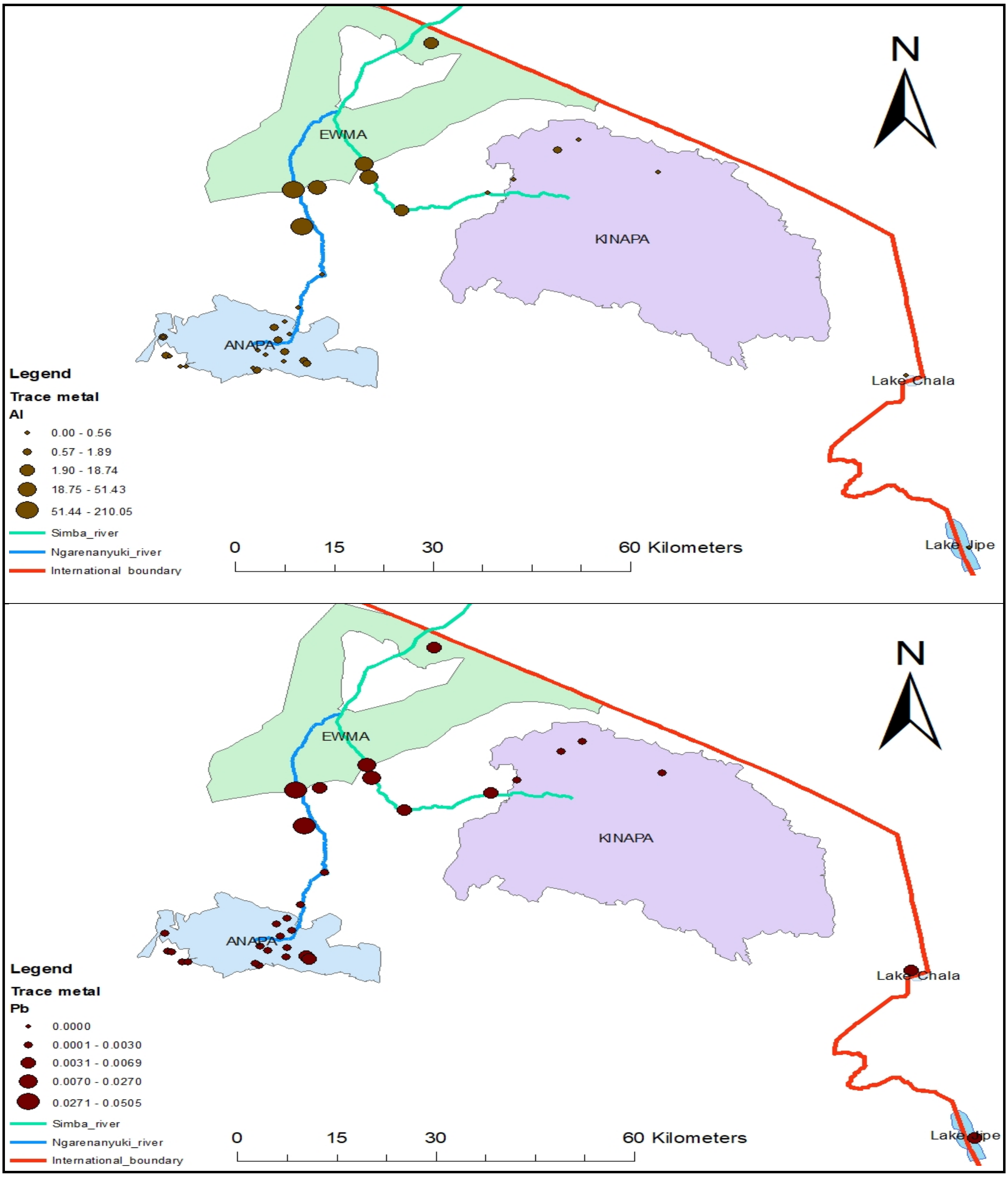

Figure 3. 15: Sketch map showing dry season spatial distribution of aluminium (top), and lead (bottom) concentration in the Kilimanjaro landscape.

Water Hardness

Figure 3.16, shows clearly that areas outside of the National Parks recorded the highest water hardness concentration during the dry season, where the water sources along the Ngarenanyuki and the Simba River sub-basins took the lead in the Kilimanjaro landscape. Calcium concentration was significantly higher ($p<0.01$, $t=3.22$, $n=56$), than magnesium concentration in the landscape.

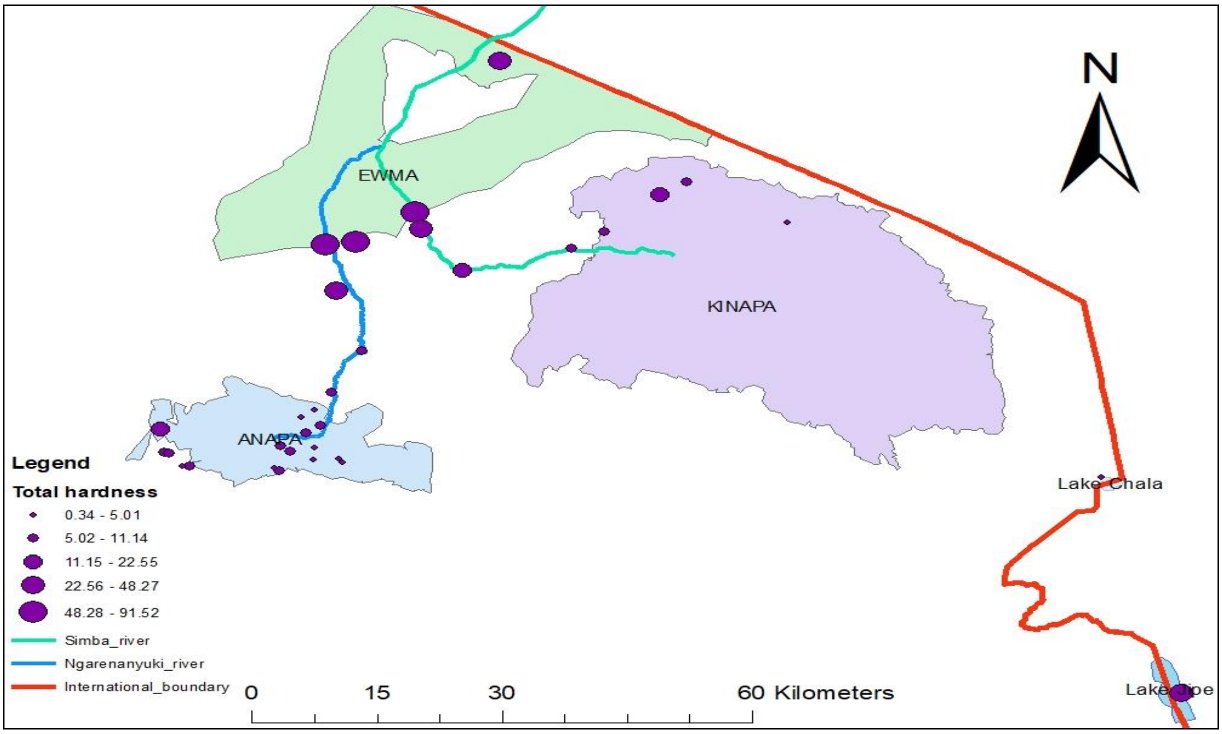

Figure 3. 16: Sketch map showing dry season spatial distribution of total hardness (combined calcium and magnesium) concentration in the Kilimanjaro landscape.

From Figure 3.17A, it is clear that the concentration of both calcium and magnesium markedly increased downstream during both dry and wet seasons in the Simba River. For example, in the dry season, calcium and magnesium concentrations downstream at site S6 were respectively about 11 and 12 fold greater than concentration measured at upstream site S1. Calcium had the highest concentration of 50 ppm as recoded downstream Simba River at the EWMA boundary during the dry season. Water hardness was markedly lower in the Ngarenanyuki River where the highest recorded calcium concentration was almost 6.5 ppm in the upstream site N1 and decreased slightly to 6ppm in the next downstream sites. Magnesium which was generally lower than calcium, also showed a similar spatial pattern, including a low concentration where the maximum value was 2 ppm in the upstream site N1 (Figure 3.17B), which is just about 25% of the total hardness recorded on this site.

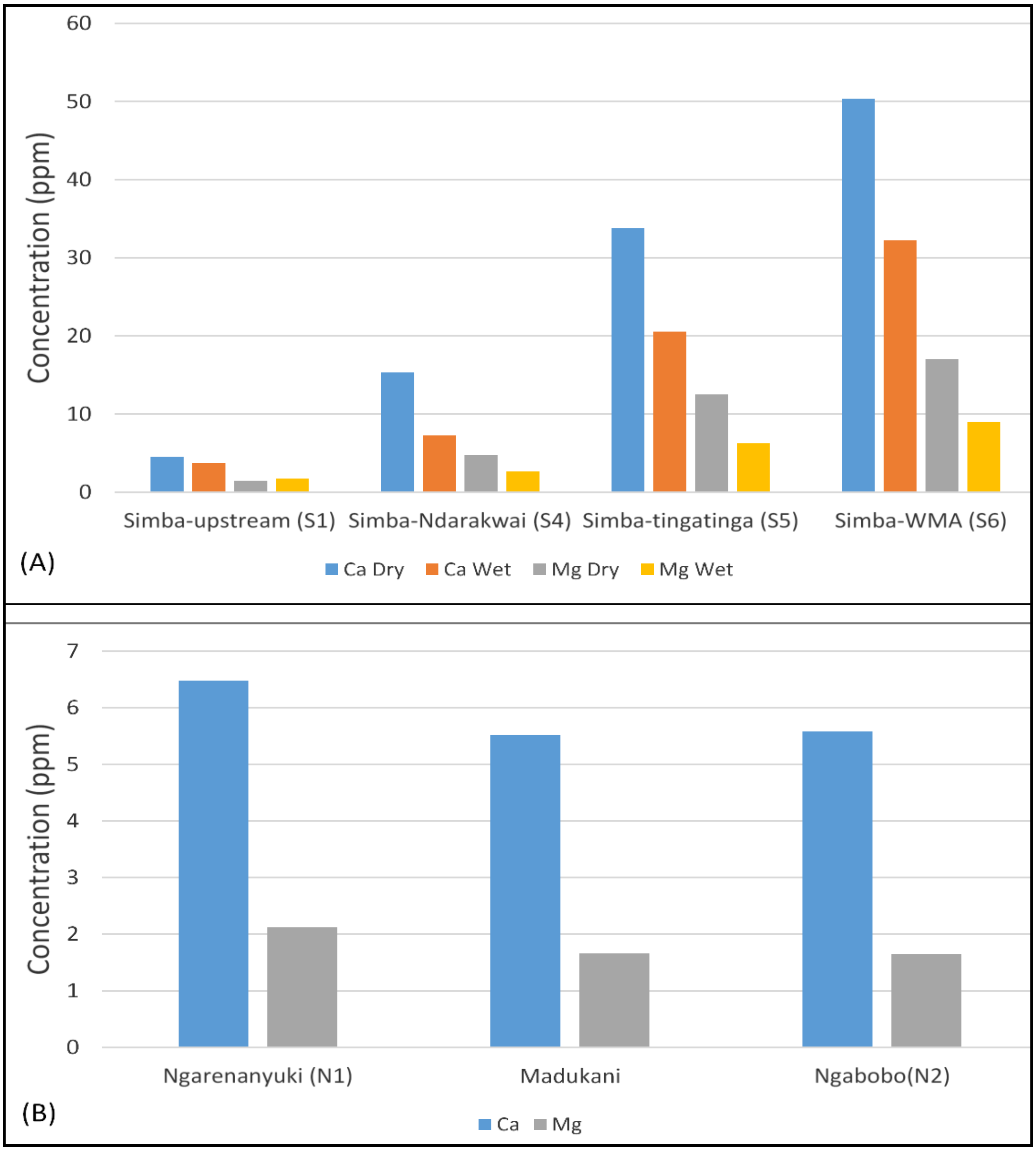

Figure 3. 17: Water hardness (calcium and magnesium) in (A) Simba River during the dry and wet seasons and (B) Ngarenanyuki River during dry season.

As is commonly the case, calcium recorded higher concentration than magnesium in water holes (Figure 3.18A) and both were present at higher concentration in the dry season than the wet season. In the dry season, Ngereiyani-madukani (H1), Ngereiyani-mbong'eti (H2) and Ngainyamo (H5) water holes measured the highest calcium concentration of 70 ppm, almost 60 ppm and 35 ppm respectively. The other water holes had calcium and magnesium concentration of less than 25 ppm in both dry and wet seasons. Figure 3.18B shows that the concentration levels of calcium and magnesium in the water extraction sites within the parks were comparatively lower than in the water sources outside the parks. Lerangwa in KINAPA recorded the highest (~10 ppm) of calcium during the dry season. The next highest calcium concentration in the parks was almost 8 ppm that was recorded during the wet season at the Kitendeni intake in KINAPA. The water sources in KINAPA particularly Lerangwa (K3), Kitendeni (K2) and Londorosi (K4) recorded higher calcium and magnesium concentration (ranging from 2 ppm to almost 10 ppm) than similar sites in ANAPA. On average, total hardness in the water extraction sites of KINAPA was 2.6 times larger than similar sites in ANAPA.

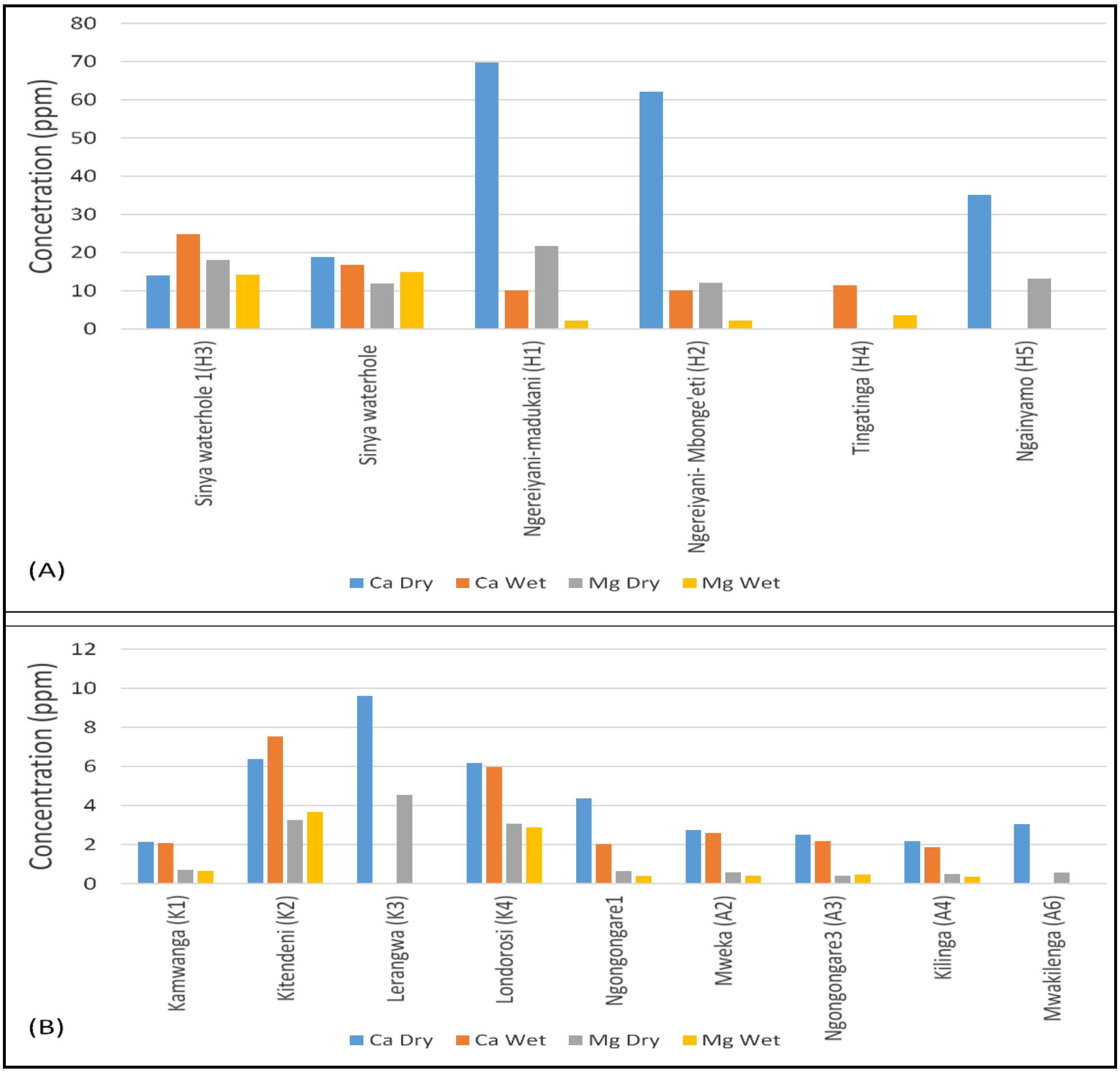

Figure 3. 18: Water hardness concentration (calcium and magnesium) during dry and wet seasons in (A) water holes, and (B) water extraction sites in the National Parks in the Kilimanjaro landscape.

3.4 Discussion

3.4.1 DO, Temperature, Salinity, pH, Fluoride and Nutrients

In the Kilimanjaro landscape, water quality varied across space and time for all sites assessed within the parks and outside the parks. In ANAPA, water quality was largely in agreement with the findings from a similar study by Elisa et al. (2016) as generally the same sites were assessed. Although the discharge was comparatively higher by up to 40% in the wildlife rich water extraction sites during this study due to an increase in rainfall over a period from 2014-2019 compared to 2008-2013 (see Chapter 2), it did not result in any marked differences in water quality. Water quality in KINAPA was not previously assessed but shows broadly similar temporal and spatial changes, the main overall difference being that ANAPA recorded slightly higher water salinity and temperature levels.

The amount of DO in water is affected by several factors including atmospheric pressure, photosynthesis by aquatic plants, respiration by aquatic community, water temperature and whether the water is stagnant or flowing as this affects aeration (UNEP, 2008). While there was a variation in dissolved oxygen (DO) levels across different sites in both ANAPA and KINAPA, there were only slight differences in the amount of oxygen in the same watercourse before and after abstraction. Differences in DO upstream of abstraction sites were probably due to increased aeration and oxygen solubility. Existing water abstraction did not significantly affect DO within the parks and in most cases, DO declined only slightly downstream of extraction sites by an amount that did not exceed 2 mg/l. The change was likely due to elevated temperature, and decreased discharge (Abowei, 2010). Water abstraction reduces downstream water flow, resulting in less dilution, increased residence time and thus increased evaporation. However, there were few sites, especially in ANAPA, where DO increased downstream possibly due to high aeration of the shallow and fast moving water on the steep slopes plus reduced temperature (and hence higher oxygen saturation) under the forest canopy. Local effects were also observed; for instance, the lowest average DO concentration of 3 mg/l (41 % saturation) occurred at Ngongongare 1 (Site A1) in ANAPA. This was likely due to the local accumulation of dead plants retained by an abstraction weir forming a stagnant pool; presumably the decaying plants consumed large amount of DO in the processes of decomposition (Mnaya et al., 2006; Abowei, 2010). The weir also decreased flow sufficiently such that water behind the weir had longer residence time sufficient to deplete DO probably due to high biochemical and sediment oxygen demand. DO concentration < 5 mg/l, may cause stress to some aquatic organisms (Hunt and Christiansen, 2000). However, overall, DO levels increased in ANAPA and KINAPA with increasing river discharge, and a decrease in temperature, and this was likely due to an increase in re-aeration in the shallow erosional watercourses and reduced exposure to solar- heating.

Water abstraction reduced downstream water flow, resulting in less dilution of organic and inorganic materials, and hence high total dissolved solids which reduce oxygen solubility. Increased residence time and thus increased evaporation in the high temperature, low lying dry areas outside both parks will also lead to reduced solubility of oxygen (Abowei, 2010; Oliveira et al., 2019). There were several cases of low DO levels in water sources outside the parks, particularly downstream the Ngarenanyuki and Simba Rivers during dry season, in contrast to the wet season during which DO levels were higher and almost stable. During the dry season, the DO levels in the water holes were as low as 33 % saturation (2.2 mg/l), probably due to a large decline in water volume of up to almost 80 %, and high water temperatures. Such low values may likely cause stress and reduce growth of aquatic organisms (Hunt and Christiansen, 2000). However, different aquatic species have different tolerance levels to low DO and therefore a water body with low DO may still support some species. Tilapia for instance, are relatively more tolerant to low DO levels compared to most other fish species (Mallya, 2007). In practice however, the welfare of herbivores is prioritised over aquatic species in the management of surface water bodies in the dry community areas and low DO is unlikely to be a limiting factor to terrestrial herbivores unless the water is totally deoxygenated. In most cases DO levels in the landscape were well above 5 mg/l and hence from this perspective both the standing and running waters are considered to be of good quality (UNEP, 2008).

The pH of surface water in the landscape was alkaline varying between 7.5 and 10, reflecting the underlying geology of the catchment (UNEP, 2008). Geology of the Kilimanjaro consists of extrusive volcanic rocks such as porphyries, basalts, trachytes, and the soil is derived largely from the volcanic rocks and ashes that formed during the major eruptions of Kilimanjaro and Meru over 250,000 ago (Istituto Oikos, 2011). This geology supports alkaline conditions. The pH in this area might also be influenced by biological effects, i.e. aquatic plants increase the water pH by consuming carbon dioxide during photosynthesis and hence influence the carbonate/bicarbonate equilibrium in the water (Baldisserotto, 2011; Nelson et al., 2011). Only a small change in pH (< 1) was noted between upstream and downstream extraction sites in both ANAPA and KINAPA. pH values in the parks slightly decreased with increasing temperature which increases ability of water to ionise to form more hydrogen ions (Gillespie, 2021). Both parks recorded a range of pH values that were well within the good quality range of 6.5 to 9 (UNEP, 2008) and which supports the growth of most aquatic species (Baldisserotto, 2011). However, outside the parks, there was a different picture as the Ngarenanyuki River and water holes in Enduimet WMA had high pH values of between 9 and 10. These higher pH values were likely caused by the long residence time due to reduced flow and water stagnation leading to an enhanced interaction between water, and rocks and soils rich in carbonate salts such as sodium carbonate (UNEP,2008), and also algal blooms, especially in the water holes. A pH above 9 exceeds the tolerance limit for the majority of aquatic wildlife species, and is generally not suitable for the health of people and terrestrial herbivores as high alkaline pH often increases carbonate complexes (Esbaugh et al., 2013). High pH levels are often harmful to aquatic wildlife such as fish, may damage external surfaces like skin, eyes, and gills, may impair organism's ability to dispose metabolic waste, and in some case may even lead to death. In addition, high water pH level enhances toxicity of some chemicals such ammonia (Lenntech, 2021). Livestock consumption of water with a pH above 9 may lead to various health problems including reduced water/feed intake, diarrhea and poor feed conversion (Bagley et al., 1997), and this healthy problem may also occur to the wild herbivores with ecological/biological requirement similar to the livestock.

Salinity varied between upstream and-downstream, and across sites within the parks. Generally, salinity in the rivers within the parks increased downstream of the extraction sites. However, salinity within the park water bodies ranged from 20 ppm to almost 350 ppm, and is therefore of sufficient quality for a healthy biotic community. The maximum salinity levels of the abstracted water within the parks were <2000 ppm a concentration at which may trigger a search for alternative less saline water sources by elephants and hence affecting distribution and space use of these, and other wild animals such as wildebeest, zebra, and buffalos (Gereta et al., 2004). However, dry season salinity outside the parks in the Ngarenanyuki River and the wild/domestic animal water holes in Ngereiyani and the Enduimet WMA were substantially higher than in the parks and the Simba River, on average ranging from 1,000 ppm to almost 4,000 ppm. Excessive water abstraction in the upland villages reduced volume of water and hence its dilution effect that led to increased salinity concentration in the downstream semi-arid areas. Further, such excessive water abstraction likely caused an increase in the residence time of the remaing little and/or slowly flowing water, and thus an increased water surface area to volume ratio resulting in enhanced evaporation and hence markedly increasing salinity (UNEP, 2008; Bhat et al., 2014). The Simba River seemed particularly affected as it consistently measured high AF values for salinity of about 1.9 and 2.7 respectively in the dry and wet seasons. Higher values in the wet season reflect the wash-up effect of run-off water, most likely due to anthropogenic inputs from farming and settlements (Fakayode, 2005; Bhat et al., 2014). Levels of salinity below 1000 ppm may not alter diversity of some aquatic species. However, diversity of wetland and riparian plants along downstream Ngarenanyuki River and around water holes in the Ngereiyani and Sinya (Sites H1, H2 and H3) is likely to be affected by high salinity levels especially so during the dry season when the concentration exceed 1,000 ppm (Hart et al., 1990). Biotic communities such as macroinvertebrates, macrophyte, micro-algae are particularly sensitive to an increase in salinity (Hart et al., 1990). However, based on the experience from other similar wildlife areas, a salinity level below 2000 ppm is considered to be within normal limits for herbivore consumption (Wolanski et al., 1999; Gereta et al., 2004) and thus, the higher concentration may limit water access. In a case where the herbivores have no reasonably accessible alternative water sources, they may consume water with a salinity in excess of 2000 ppm up to a certain level. Some very high salinity values (7700 ppm) were noted in the Sinya water hole (Site H3) in April 2019 during the peak of the dry season. Herbivores such as livestock may drink this high saline water without being adversely affected if they are not subjected to other environmental stress such high heat stress and water loss. Sheep can tolerate a high saline water of up 10, 000 ppm for a few months (Wolanski et al., 1999). However, long-term consumption of such high saline water may pose a significant risk to lactating, pregnant and young livestock species (Bagley et al., 1997). While animals, i.e. cattle, zebra and wildebeest, were observed drinking this high saline water at Sinya water hole, this was not considered a limiting factor as they could readily switch to the neighbouring less saline water hole (i.e.H3B) that had a salinity of about 1,300 ppm. Usually wild and domestic herbivores respond differently to high salinity levels, and such a response partly seems to depend on whether there is an alternative and reasonably available low saline water. In Tarangire, one of the neighbouring ecosystems in northern Tanzania, a surface water salinity of about 8000 ppm may trigger seasonal migrations of wildebeest, zebra, buffalos and elephants (Gereta et al., 2004). A salinity level ranging from 5000 ppm and above in the southern plains of Serengeti is linked to the onset of wildebeest and zebra annual migration (Gereta and Wolanski, 1998). However, such migrations are also influenced by other factors, especially rainfall that in turn influences water and grass availability, and pasture and grazing opportunities (Boone et al., 2006).

While moderate fluoride level in water is important for the maintenance of bone and teeth in both humans and other animals, consumption of water or foods with excessive amount of fluoride over a long period will lead to fluorosis that affects both skeletal and soft tissues such as brain and kidney (Kilham and Hecky, 1973; Shahab et al., 2017). The level of fluoride in the extraction water sources in both National Parks also showed spatial and temporal variability similar to the other water quality parameters. Fluoride in ANAPA mirrored changes in salinity as it showed a common spatial pattern. This suggests that factors affecting salinity such as rock/soil minerals and water abstraction may also apply to fluoride. Fluoride recorded relatively high values in somes sites of western and eastern parts, which is possibly due the interaction of water with fluoride-rich volcanic rocks, plus volcanic ash that extruded from the past explosion of Mount Meru and Kilimanjaro (Kilham and Hecky, 1973; Istituto Oikos, 2011; Ghiglieri et al., 2012). East African rift valley countries are among the regions with records of high fluoride levels particularly in surface water bodies, rather than aquifer (Malago et al., 2017). Only about 40% of the sites in ANAPA recorded a fluoride level within WHO guideline of ≤ 1.5 mg/l, which is for drinking water, and about 25% of the extraction sites were above this value and even the 4 mg/l limit for Tanzania (Malago et al., 2017). However, all sampled sources in KINAPA met the WHO fluoride limit for potable water. While there are guidelines for human drinking water, the level of fluoride in water for wild animal consumption is not very clear, and hence not clearly known whether fluoride levels both within the parks and in the dry areas should be considered safe for wild herbivore consumption. However, high fluoride levels may be harmful to wild herbivores because of their relatively non selective feeding and drinking habit (Shahab et al., 2017). A study by IPCS (2012) demonstrated that skeletal fluorosis in wild deer can be caused by a fluoride level of or above 35 mg/l in drinking water. A fluoride level > 12 mg/l may cause adverse chronic effects to ruminant livestock including crippling, although short-term exposure might be tolerated based on site-specific conditions such as water requirements and nutritional interactions (DWAF, 1996a). Higher levels of fluoride ranging from 30 to 36 mg/l in the downstream reaches of the Ngarenanyuki River and the Ngereiyani water holes (Site N2 and H1) respectively might therefore be harmful to the wild animals. The impacts of such high fluoride on the wild herbivores and livestock in these dry areas have not been examined and clearly, merits further study.

The level of the nutrients nitrate and phosphate was relatively low within the parks compared to outside areas. Five sites in ANAPA and one in KINAPA measured nitrate levels above 3 mg/l which possibly indicate the presence of pollution (WaterAid-Nepal, 2011). Out of these sites, the highest value of almost 9 mg/l was recorded at Sambasha (site A18) in ANAPA. High nitrate values in the parks, may be mainly due to run-off of animal waste, and also effluent from human activities for Sambasha (A18) as this site was in close proximity to community areas. In contrast to ANAPA, KINAPA measured relatively high nitrate values in the rainy period, suggesting a possible release of nitrate, likely from animal waste and decomposing plant matter from the forest, which enters the river via run-off water (Bhat et al., 2014). However, the level of nitrate in the parks and also most of the outside dry areas remained well within the guideline limits of 50 mg/l for drinking water (WHO, 2004b). When within required limits, these nutrients support physiological processes of living organisms, and therefore are important for ecosystem health (Singh, 2016). However, the water holes recorded high levels of both nitrate and phosphate. In particular, the Ngereiyani water hole (Site H1) consistently recorded extremely high dry season values, likely due to cattle defecating in the water hole, and at one occasion the level of nitrate reached almost 470 mg/l. Excessive water abstraction especially in the Ngarenanyuki River reduced the amount and frequency of water release into the water holes, which then led to less dilution and flushing effect, and hence caused an increase in nutrients concentration in the water holes during the dry season. This high level of nitrate may cause eutrophication (Singh, 2016). Also, eutrophication may elevate pH due to dissolved carbon in photosynthesis (Chislock et al., 2013). In addition to nutrient availability, high amount of sunlight and carbon dioxide will also contribute to eutrophication (Chislock et al., 2013). During the dry season, algal growth could be noticeable in patches in those water holes exposed to full sunlight. However, algal blooms in the water holes were not widespread possibly due to frequent disturbance by herds of livestock and wild animals. Eutrophication not only adversely affects the water quality by changing the DO and pH, but may also produce harmful chemicals such as toxic ammonia (Rahman and Jewel, 2008). In addition, cyanobacteria release toxins i.e. neurotoxins and hepatotoxins that can be harmful to domestic animals and wildlife (DWAF, 1996a; Stewart et al., 2008).

3.4.2 Heavy metals

Heavy metal concentrations varied both temporally and spatially across the landscape with high values occurring outside the parks in the lowland semi-arid areas. Aluminium, iron and manganese recorded the highest values in the landscape which is to be expected as these elements are the most abundant components of minerals in the earth's crust (DWAF, 1996b). Relatively low values of all metals were measured in the National Parks and generally, the level of heavy metals in most of the water extraction sites within the National Parks were within recommended guideline values for animals as well as humans. The highest metal concentrations measured at the existing water extraction sites in the parks during the dry season were: copper (0.014 mg/l); zinc (0.04 mg/l); arsenic (0.002 mg/l); cadmium (0.0003 mg/l); lead (0.005 mg/l). None of these metal concentrations exceeds the recommended guideline values (WHO, 2004a; 2004b; 2003b; 2003a), which are: copper (2 mg/l); arsenic (0.01 mg/l); cadmium (0.003 mg/l); lead (0.01 mg/l; and zinc (3mg/l). Being dependent on the environmental pH (Esbaugh et al., 2013), toxicity of these metals is also unlikely as the pH level in the landscape was essentially alkaline.

However, iron and aluminium measured relatively higher dry season concentrations and the highest values were almost 3 mg/l and 2 mg/l respectively in KINAPA at Lerangwa intake (Figure 3.14B). These metals are always present in higher concentration than other heavy metals (DWAF, 1996b). This iron concentration exceeds the South African water quality guidelines of 1-3 mg/l (DWAF, 1996b) as well as the WHO recommended guideline of 0.3 mg/l which is mainly based on taste and appearance rather than harmful health effects (WaterAid-Nepal, 2011). Aluminium concentration of 3 mg/l also exceeds the human drinking water value recommended by WHO (2010) and DWAF (1996b) of 0.9 mg/l and 0.15 mg/l respectively. However, as this was only a case in a single site whose pH was alkaline, and for the only two metals, usually categorised as less toxic (Raikwar et al., 2008), this is not deemed as a significant water quality issue. Excessive metal concentration in the surface water within the parks, was however not expected, as the areas are not exposed to human activities such as industrial, mining, or agricultural activities, which in many cases are behind heavy metal pollution. Therefore, the relatively high aluminium level encountered in in KINAPA has likely originated from physical and chemical weathering of rocks e.g. igneous rock. Availability of aluminium rich soils in Mt. Kilimanjaro is clearly demonstrated by Mwende (2009). As the concentration of the heavy metals did not generally exceed the limits for human drinking water, then the available surface water is also considered suitable for biodiversity.

In most cases, metal concentrations increased upstream in the Ngarenanyuki River where higher values were recorded at site N1 in ANAPA, and downstream for the case of Simba River, where higher values were recorded in the section of the river that traverses the community lands at sites S5 and S6. On average, metals concentration in the downstream Simba River was almost 20 times larger than concentration recorded in the upstream. Again, aluminium (Al) and iron (Fe) had relatively higher values than other metals. The concentrations of Al and Fe increased downstream the river, with high values ranging from almost 10 mg/l to almost 40 mg/l and 37 mg/l respectively, which implies almost 70 and 66 times greater downstream concentration than the upstream concentration. Thus, the concentration of Al and Fe not only increased downstream the Simba River, but also exceeded guideline range for livestock use of 0-5 mg/l and 0-10 mg/l (DWAF, 1996a), and guideline values for human use of 0.9 and 1-3 mg/l respectively (WHO, 2010). Lead (Pb) also had high concentration downstream the Simba River measuring almost 0.02 and 0.03 mg/l at sites S5 and S6 respectively which exceeded the guideline limit (0.01 mg/l) of water quality for human use (WHO, 2004). As reflected in accumulation factor (AF), metal concentrations increased considerably downstream the Simba River. This excessive metal concentration suggests that water in the downstream Simba River is unfit for human, domestic and wild animal consumption. A downstream metal increase is most likely an indication of anthropogenic factors including excessive water abstraction that reduced dilution of contaminants, resulting in increasing downstream concentrations of heavy metals in the Simba River during the dry season. Further, the increase in downstream metal concentration was also likely contributed by other anthropogenic sources such as fertilisers, pesticides, domestic waste and other effluents that drain from the adjacent farms into the downstream river during both dry and wet seasons (Bourg and Loch, 1995; Yahaya et al., 2009; Bhat et al., 2014). Leeching of domestic waste and agricultural inputs into the surface waters due to among other, poor land management, is likely to be the main source of metal pollution, and this is also a problem in the rest of Tanzania as reported by Mohammed (2017).

Compared to the Ngarenanyuki River, the Simba River measured slightly lower pH (pH of about 8 compared to 9.5 in the former river) and relatively low upstream temperature (about 13°C) but high downstream temperature (around 25°C) which might have facilitated release of metals and hence higher downstream metal concentration (Li et al., 2013; DWAF, 1996b). On the other hand, a decline in downstream metal concentration in the Ngarenanyuki River is possibly contributed by its high water pH (almost 10) which has the potential to reduce trace metal solubility (Li et al., 2013). Difference in the types of crops grown and agricultural inputs applied in the Simba and Ngarenanyuki Rivers sub-basins might also contribute to the observed difference between the rivers metal concentration (Bonten et al., 2008). However, surface water interactions with different soil and mineral composition might more likely explain the spatial variations in the concentration of heavy metals in the landscape (Alloway, 2012). The Ngarenanyuki River water interacts with high sulphur bearing minerals, as it flows from the ash cone in mountain crater to the downstream areas (Surdy et al., 1932). Excess sulphur and the associated reduced conditions are known to lower heavy metal solubility (Bourg and Loch, 1995), and this might be another possible reason for lower metal concentration in the Ngarenanyuki River compared to the Simba River. The observed higher heavy metal concentrations in the upper than lower reaches of the Ngarenanyuki River might be due to weathering of the metal rich volcanic rock in the Mount Meru from which the river originates. As there is no anthropogenic source of heavy metals in the park, this seems a plausible explanation because rocks and soils close to the volcanic mountains including those in Tanzania have been shown to have higher concentration of heavy metals (Amour and Mohammed, 2015).

Water sources in the lowland semi-arid areas, i.e. water holes and the downstream part of the Simba River, recorded relatively high metal concentrations compared to the values in the National Parks. All these surface water sources are located in the downstream and thus influenced by among other factors, human and natural, that take place around them and in the upstream areas. Thus, some water sources measured metal concentration that exceed the recommended values for human and livestock drinking water. Metal concentrations increased during the dry season. For instance, the percentage change in metal concentration between wet and dry season ranged on average from 40 to 70% in the water holes. High dry season concentration is likely due to less dilution and lack of flushing due to low or no rain and/or river water, and an increase in water evaporation (Abowei, 2010). A similar study by Shanbehzadeh et al.(2014) examined heavy metal pollution in the Tembi River in Iran and found a higher downstream metal concentration due to municipal wastes and higher dry season concentration due to the water evaporation and little/no rain. The highest metal concentrations in the Kilimanjaro landscape were found in water holes for livestock and wild animals in the semi-arid area. In particular, aluminium and iron measured highest concentrations, which were at least thrice the maximum values measured in the Simba River. For instance, Ngereiyani-madukani (site H1) and Ngainyamo (site H5) water holes had the highest aluminium concentration recording almost 200 mg/l and 150 mg/l respectively during the dry season. Iron concentration marked the second highest value, ranging from almost 50 mg/l in Ngainyamo water hole to almost 100 mg/l at Ngereiyani-madukani water hole. Higher metal concentration in these water holes that are located in the dry lowland downstream area, are also contributed by a number of other factors especially water insteractions with different soils and minerals, and also leaching from the agricultural dominated catchment.

While, aluminium is relatively less toxic to animals than other heavy metals such as copper and cadmium, its consumption at high levels may have adverse health effects (DWAF, 1996b). However, actual effects of aluminium on terrestrial wildlife are poorly known (Rosseland et al., 1990). According to DWAF (1996a), a concentration between 0 and 5 mg/l has no adverse effects on the livestock but there could be adverse impacts on animal health (e.g.neurotoxicity) when this limit is exceeded. Toxicity of aluminium is highly pH dependent and mainly increases in strong acidic ($pH<6$) conditions but also in strong alkaline($pH>8$) conditions (Scanca and Milacic, 2006). For instance, a concentration above 10 mg/l, may cause neurotoxicity to livestock species (DWAF, 1996a). This might be the case for some of the dry season water sources, i.e. downstream section of the Simba River and the Ngereiyani and Ngainyamo water holes, which had higher dry season concentration. Such metal concentration does not only affect the water quality but also may cause health problems to animals and impair species diversity and abundance over long-term (Adeogun et al., 2012). High aluminium concentration is also toxic to aquatic life especially for gill-breathing biota including invertebrates and fish when inhabiting acidic waters. However, aluminium may also excessively accumulate in some aquatic biota such as crustaceans living in neutral pH conditions (Elangovan et al., 1999). Study by Brautigan et al.(2012) also demonstrated that aluminium is phytotoxic at high $pH>9.2$ where it substantially reduces development of plant roots and stem. In addition to the concentration in water, aluminium accumulates in plants and invertebrates and easily it enters terrestrial food chain where it can interfere with the metabolic processes including breeding in mammals and birds (Rosseland et al., 1990).

Iron was also common in the surface water of the study area and was more abundant outside the National Parks. However, unlike aluminium, iron is an essential nutrient to both plants and animals supporting biochemical and physiological processes. Iron is important part of haemoglobin, which is responsible for transportation of oxygen in the red blood cells (Alloway, 2012). However, when consumed in excess quantity it may lead to health problems (DWAF, 1996). Like aluminium, iron toxicity is also pH dependent, and the metal is commonly found at high concentration in acidic conditions (de Souza et al., 2021) and usually dissolved iron concentration is very small under neutral and alkaline pH and oxidising conditions (DWAF, 1996a). Most water in the landscape have iron concentration within a range of 0< to >10 mg/l which is acceptable for animal use based on the South African water quality guideline for livestock use (DWAF, 1996a). However, concentration beyond this range may be toxic to animal health. In particular, concentration beyond 50 mg/l, such as that recorded in Ngereiyani (H1) and Ngainyamo (H5) water holes during the dry season may cause serious health problems such as diarrhoea, and damage to internal organs (DWAF, 1996a). The metal toxins may enter biota directly through ingestion from water or indirectly through food chain (Verma and Dwivedi, 2013). High concentration of iron may also impair animal physiology and may reduce diversity and abundance of invertebrates and fishes, mainly affecting metabolic processes and osmoregulation and also by altering the structure of benthic habitat (Coup and Campbell, 1964; Vuori, 1995; Adeogun et al., 2012).

Manganese which is one of the most abundant metals in earth's crust (Kamble and Thakare, 2014) was also measured in the Kilimanjaro landscape. This is a less toxic and an essential nutrient which activates large number of enzymes in plants and animals and its deficiency in animals leads to among other impaired reproduction and skeletal/bon deformities (Alloway, 2012). The concentration of this element whose availability in the water environment also depends on pH (Kamble and Thakare, 2014), is important for bone formation, reproduction, growth and fertility. Its concentration in the Kilimanjaro landscape, did not exceed 10 mg/l, and therefore falls well within a no adverse effect range according to South African water quality guidelines for livestock use (DWAF, 1996a). As such, surface water in the landscape is also deemed to be within required limits for wild herbivores use.

The maximum concentration for the remaining heavy metals in the water holes during the dry season were: Cu (0.1 mg/l); Zn (0.24 mg/l); Pb (0.05); As (0.0124 mg/l); and Cd (0.00076 mg/l). According to the South African water quality guideline for livestock use, such levels of metal concentration in water holes are considered to be within a safe range for terrestrial herbivore consumption. Although many metals are essential to biota, it is widely accepted that they may be toxic at a higher dose (Lazarus et al., 2005; Alloway, 2012; Verma and Dwivedi, 2013). This is also a case for some of freshwater organisms which are relatively sensitive to heavy metal pollution (Javanshir et al., 2011).

3.4.3 Water hardness

Water hardness was also assessed as it has the potential to negatively or positively affect the health of biota (DWAF, 1996b; Sahinduran et al., 2007). Water hardness is usually defined as the amount of calcium and magnesium dissolved in water (USGS, 2021). Surface water in the Kilimanjaro landscape may generally be characterised as soft water (due to their low levels of calcium and magnesium concentrations which on equivalency is less than 60 mg/l of calcium carbonate (WHO, 2011). Most of the surface water in the landscape is considered good with no significant adverse impacts as calcium and magnesium concentrations was less than 32 mg/l, beyond which there could be health effects, as well as scaling and impairing of soap lathering (DWAF, 1996a).

In the Kilimanjaro landscape, calcium concentration was significantly higher than magnesium concentration. This is commonly the case as calcium is usually more abundant in natural water and the wider environment compared to magnesium. Calcium concentration may increase to 100 mg/l or more in the natural environment, especially groundwater, whereas magnesium concentration in such environments is usually around 50 mg/l (WHO, 2011). Calcium and magnesium concentration generally increased in the dry season and downstream for the case of the Simba River, but upstream for the case of the Ngarenanyuki River. Water holes in lowland semi-arid areas measured the highest total water hardness, and like the heavy metals, showed a similar spatial and temporal pattern. This situation suggests that they might both be influenced by the same factors. Dry season increase in water hardness is likely due to high evaporation and lack/less flushing that leads to concentrating of existing metals.

On the other hand, accumulation factor (which often estimates the degree of contamination due to anthropogenic inputs) indicated a significant downstream increase in water hardness in the Simba River, which was more than 10 times higher than the upstream concentration. Such higher accumulation factor likely suggests a presence of anthropogenic contributions. A similar situation was reported in the Sukhnag River in the Kashmir Himalayas (Bhat et al., 2014). However, change in soil and mineral compostion along the gradient from Mt. Meru and Kilimanjaro downsream to the Amboseli basin, might also contribute to the observed variations in water hardness.

In several water sources, calcium concentration was positively correlated with magnesium. For instance, the wet season concentration revealed a strong and positive correlation (R=0.86) between calcium and magnesium in both Sinya water holes and both Ngereiyani water holes. Such observation is in line with a study by Venkatasubramani and Meenambal (2007) which showed that calcium and magnesium are usually associated in the aquatic environment. While an experiment to differentiate the impacts of soil from that of humans would provide more conclusive evidence, this correlation suggests that aqueous calcium and magnesium concentrations are closely related due to-soil-surface water interaction during the wet season and subsurface leaching during both the wet and dry seasons .

Calcium and magnesium are essential elements for human and other animals and their deficiency has been linked with several diseases such as osteoporosis (bone fragility), and nephrolithiasis (kidney stones) and hypertension (WHO, 2011). Extreme water hardness is also linked with urolithiasis in livestock (Sahinduran et al., 2007). In addition, water hardness may also influence toxicity of heavy metals. For instance a study by Kiyani et al.(2013), demonstrated that toxicity of some heavy metals e.g. copper and zinc, increase with a decrease in water hardness. In the study area, none of the surface water sources was classified as extremely hard but a few surface water sources such as Ngereiyani and Ngainyamo water holes could be characterised as having moderately hard water especially during the dry season as calcium measured more than 60 mg/l of calcium carbonate (WHO, 2011). Yet, while relatively high, calcium and magnesium concentration in these water holes did not exceed 100 mg/l and these values fall well within a range of water quality that is permissible for domestic, industrial and irrigation use as reported in study by Venkatasubramani and Meenambal (2007) in Tamilnadu. Du Toit and Ebedes (1996), reported a calcium concentration of 1000 mg/l as the maximum allowable limit in water for the wildlife use. Water calcium levels between 0 <and <1000 mg/l and magnesium levels between 0< and <500 mg/l are generally without adverse effect for livestock species (DWAF, 1996a), and hence with respect to water hardness, the water holes are considered safe for use by the wild herbivore species.

3.5 Conclusion

This study aimed to examine temporal and spatial changes in surface water quality due to natural and anthropogenic factors focusing mainly on abstraction, and their effect on biodiversity health at the ecosystem scale in the Kilimanjaro landscape with particular reference to wildlife. The study focused on the current status of salinity, water hardness (calcium and magnesium), fluoride, nutrients, dissolved oxygen, pH, temperature, and heavy metals in the Kilimanjaro landscape. While the values of these water quality parameters varied across space and time, in general, the concentration of many parameters increased with the dry season in both the standing and running waters and this was largely due to evaporation. Overall, there were higher concentrations of all water quality parameters except DO outside (hence downstream) than inside the National Parks. The quality of water in the National Parks did not seem to be significantly impacted by anthropogenic activities such as water abstraction. However, water quality outside the National Parks, particularly in the lowland and semi-arid community wildlife areas was adversely affected by human-induced pollution and excessive water abstraction which reduced dilution of contaminants, resulting in low DO and increasing downstream concentrations of heavy metals salinity and nutrients. Human impacts were particularly apparent in the Simba River, and in the water holes in the semi-arid areas. However, water insteractions with soils and minerals likely also contributed to variations in concentration of physicochemical parameters in the Kilimanjaro landscape. Overall, water quality in the Kilimanjaro landscape is largely of good quality for wildlife use. However, the dry season level of Al, Fe, salinity, fluoride, nitrate and phosphate in some of the surface waters of the semi-arid areas were likely too high to support a healthy biodiverse community, particularly over a prolonged exposure (Alloway, 2012). The potentially affected water bodies are the downstream sections of the Simba and Ngarenanyuki Rivers, and Sinya, Ngereiyani and Ngainyamo water holes.

This study has provided key information on the status of surface water quality which provides crucial benchmark information for water quality monitoring and further study of water quality in the Kilimanjaro landscape. The study has identified the factors contributing to the degradation of water quality, among which anthropogenic factors play a substantial role. The study therefore paves the way for establishing and implementing appropriate policy and management actions to mitigate negative anthropogenic impacts on surface water in this and similar ecosystems. In particular, urgent policy and management actions geared at controlling excessive water abstraction and un-sustainable farming practices is required to address the existing threat especially of dry season poor quality water in the semi-arid wildlife rich areas in the Kilimanjaro landscape.

3.6 References

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Chapter 4: Surface water determines the abundance and space use of herbivores in the Kilimanjaro landscape, Tanzania

Abstract

This study investigated how changes in surface water and water extraction influenced herbivore abundance and space use patterns in the Kilimanjaro landscape in northern Tanzania**.** This study covered a semi-arid low-lying area of West Kilimanjaro and two mountainous Arusha and Kilimanjaro National Parks. A transect-based ground count was carried out to quantify herbivore species in relation to available surface water sources. The survey was conducted monthly and extended over dry and wet seasons for 6 months in the Parks and 15 months in the semi-arid region outside the parks. The number of transects in the parks was 10, and 13 outside the parks. In the parks, herbivores were counted along extracted watercourses extending upstream and downstream of water extraction sites in Arusha National Park and only downstream of extraction sites in Kilimanjaro National Park, while outside the parks the transects were established perpendicular to surface water sources. Water quantity (i.e. volume and discharge) and quality at these study sites were also measured. The survey team walked quietly along the transects recording the number, species of animals on each side of the transects, and the dominant habitat type. Field data were analysed by using ArcGIS, and generalised linear mixed effects model in software R version 3.6.1. Herbivores abundance and distribution varied across space, seasonality and species. Overall, herbivore abundance increased during the dry season and decreased during the wet season. Wild herbivores and livestock were mainly influenced by water availability as opposed to water quality especially during the dry season when water was scarce. Water quality especially in the semi- arid areas varied markedly with seasons, where some sources attained high pH values (pH>9) and high salinity levels above 7,000 ppm. During this period, herbivores (both domestic and wild) relied heavily on the few remaining water sources. Several species such as cattle, zebra, and wildebeest were drinking from the scarcely available water sources including the highly alkaline and saline waters of Sinya, Ngereiyani and Ngainyamo water holes in the semi-arid areas. While forage was also an

important factor, it was obvious that water played a central ecological role as manifested in the herbivore abundance and distribution that concentrated around available water sources during the dry season. The landscape is faced by a growing water crisis largely from excessive water abstraction within and outside the parks, mainly to meet growing needs for domestic and irrigation farming that in turn leads to deprivation of water to the people, livestock, and wild animals in the downstream areas. Such crisis is affecting both the people and wildlife through emergence of conflicts and changes in abundance and distribution of animals particularly during the dry season. This water crisis might also be causing a physiological stress to herbivores including that related to drinking poor quality water. It is therefore a matter of urgency to plan and implement ecologically sustainable water and land resources management to promote sustainable biodiversity conservation and development in the whole landscape.

4.1 Introduction

Excessive freshwater extraction often associated with a rapidly growing human population is massively affecting hydrology and surface water availability (Drijver and Marchand, 1985; Duda and El-Ashry, 2000; Smit et al., 2013; Allam et al., 2018). Such extraction is conducted to cater for increasing human demands especially for irrigation farming, hydroelectric production, livestock watering and domestic supply. Unfortunately, this extraction is usually not environmentally friendly and it threatens biodiversity in protected areas (PAs) both upstream (those located in the upper catchment) and downstream (those located in the lower catchment) and also in community wildlife areas in the sub-Saharan Africa (De Leeuw et al., 2001; Kiwango and Wolanski, 2008; Elisa et al., 2016). Although there are few cases documented on the impacts of excessive water extraction within PAs, it is increasingly evident that impacts of water mismanagement are widely felt both in the protected areas' freshwater and terrestrial systems (Stommel et al., 2016; WWF, 1999). Unsustainable irrigation farming often characterised with excessive water extraction is the common cause of the pervasive water and biodiversity crisis especially in arid and semi-arid areas (Lemly et al., 2000; Vörösmarty et al., 2010).

Although the existing water crisis is predominantly human-induced, it may be exacerbated by climate change (Vörösmarty et al., 2000). Some climate change predictions point to an increase in rainfall (Christensen et al., 2007), rising temperatures and rainfall variability in East Africa including the Kilimanjaro landscape (Agrawala et al., 2003; Otte et al., 2017; Said et al., 2019). These will interact with the human-driven water and the associated biodiversity resources crisis. Biodiversity will be increasingly affected in various ways including loss of species, degradation and or loss of habitat, blockage of migration routes, changes in species behaviour, abundance and distribution, and infestation by invasive plant species. For instance, establishment of a dam for irrigation farming in the upstream Logone River in Cameroon adversely affected the downstream Logone floodplains and Waza National Park, through reduction of the grazing land for ungulates and elephants and of water bird habitats. This led to an overcrowding of water birds in the small remaining suitable habitats (Lemly et al., 2000; Loth, 2004). Aggregations of animals around scarce water sources in the semi-arid savannah may increase the risks of contracting infectious diseases including intestinal parasites and contact transmitted diseases such as the foot and mouth disease (Ogutu et al., 2010; Strauch, 2013; Jori and Etter, 2016). In Tanzania, Stommel et al. (2016) found that extraction of the Great Ruaha River water for irrigation in the upstream areas during the dry season caused a marked reduction in river flow, leading to poor water quality through increased salinity and bacteria load. In such cases, wild animals fail to consume the poor quality surface water. The African elephant (*Loxodonta Africana)*, plain zebra (*Equus quagga)* and warthog (*Phacochoerus africanus)* can find other water sources underground by digging waterholes to find good quality drinking water that is then also consumed by other wild animals. However, excessive river water extraction upstream of PAs often forces some wild animals to move outside the PAs in search for water, and during which the animals suffer from poaching and human-wildlife conflicts. Gichuki (2002) reported a case of excessive water extraction in the upper Ewaso Ng'iro River in Kenya, which resulted in a marked water scarcity, affecting both people and wildlife in the lowlands, and this resulted in human-wildlife conflicts that resulted in the killing of wild animals. The occurrence of water related conflicts are also reported in several PAs in Tanzania such as in the Katavi and Ruaha National Parks (Mtahiko et al., 2006; Elisa et al., 2010).

While, water plays a central role for biodiversity inside and outside PAs as it helps control the abundance and distribution of wild herbivores in the savannah of eastern and southern Africa (Chamaillé-Jammes et al., 2007; de Beer & van Aarde, 2008; Douglas-Hamilton et al., 2005; Kikoti, 2009), other factors may also be important, such as vegetation quality and quantity, and human disturbance i.e. livestock grazing (De Leeuw et al., 2001; Ndaimani et al., 2017; Redfern et al., 2003). In addition, the species vary in their water dependence in which case some need to drink every day while some are more drought tolerant (Redfern et al., 2003). Therefore, species are expected to change their space use patterns to be more dependent on perennial water sources in the dry season, and there will be a differential effect on species with the biggest impacts on highly water dependent species.

The trans-boundary (as it extends in both Kenya and Tanzania) Kilimanjaro landscape is famous for its abundant wildlife. This landscape comprises strictly PAs, community conservation areas, village grazing and farming lands (Figure 4.1). Part of the landscape includes the mountainous Arusha and Kilimanjaro National Parks, which serve not only as biodiversity rich areas but as also important water catchments, relatively rich in freshwater that drains to the neighbouring low-lying areas. Part of the low-lying areas include the northern semi-arid leeward area (north of Mt. Kilimanjaro and Mt. Meru) also known as West Kilimanjaro. Being on the leeward side, this area is semi-arid and it is further deprived of water by excessive river extraction taking place in the upper villages and also inside the National Parks to cater for irrigation, livestock and domestic uses. The same freshwater is also needed by the wild animals. The landscape is also harbouring a high and rapidly growing human population (NBS, 2012) that exerts a high pressure on water and land resources (Agrawala et al., 2003). Fresh water is therefore a pivotal resource supporting socio-economic and ecological functions in the entire landscape. Unfortunately, only little is known about the status of surface water availability and its impacts on the biodiversity and ecological integrity in this area. With the exception of the small study conducted in Arusha National Park (Elisa et al.,2016), I have found no other studies explicitly focusing on how changes in surface water availability are impacting on the biodiversity in the landscape. However, a few studies in the landscape suggest that water might be playing an important role on the distribution of wild herbivores (Western and Lindsay, 1984; Muruthi and Frohardt, 2006; Kikoti, 2009, 2010; Elisa et al., 2016). However, there is no study quantifying how the anthropogenic changes in surface water affect the abundance and distribution of herbivores in the landscape.

This study examines how surface water availability affects herbivore abundance and space use. I hypothesized that species' abundance and space use patterns are likely to be more dependent on perennial water sources in the dry season and there may be a differential effect on species with the biggest impacts on highly water dependent species. It was predicted that herbivore species abundance would increase close to surface water sources during the dry season when water is scarce, and increase away from surface water sources in the wet season when water is abundant in the landscape.

4.2 Methods

4.2.1 Study area

The Kilimanjaro landscape (Figure 4.1) is located in northern Tanzania and southern Kenya and consists of Arusha National Park (ANAPA) and Kilimanjaro National Park (KINAPA), and the lowland semi arid area that includes West Kilimanjaro region, which is largely situated between, and north of these parks. ANAPA (552 km^2) and KINAPA (1,665 km^2) are mountainous parks and key water catchments on the slopes of Mt. Meru and Mt. Kilimanjaro respectively. Kilimanjaro is the highest mountain in Africa and the highest free standing mountain in the world, with its highest point at 5895 m above sea level (Kaseva & Moirana, 2010). Both parks have high annual rainfall, which is up to 2200 mm in the southern slope of Kilimanjaro (Røhr & Killingtveit, 2003), and almost 1480 mm in ANAPA (ANAPA, 2020). The parks are thus important sources of water for the neighbouring human community and the wildlife within the entire landscape. There are several water extraction infrastructures within the parks, some of which excessively extract the available water leading to water deprivation in the downstream environments during the dry season. In KINAPA, there were several open chambers and pools/ponds due to leakages along the water extraction pipelines from which wild animals drink water. The available open chambers were established mainly to protect the water infrastructures from damages caused by wild animals seeking for water when the downstream areas are deprived of water due to excessive dry season abstraction of rivers in the upstream areas within the park.

In contrast to the high rainfall in the forested parks, the West Kilimanjaro region receives a much smaller annual rainfall of about 350 mm (Altmann & Alberts, 2020). The area consists of several protected and non-protected areas including the livestock ranch NARCO (303 km^2), and Ndarakwai wildlife ranch (44 km^2), Enduimet Wildlife Management Area (EWMA) (1100 km^2), and two wildlife corridors. These are Kisimiri (currently encroached by expanding human settlements and farming) and Kitendeni, that respectively link ANAPA to the West Kilimanjaro area, and KINAPA and the Amboseli National Park in Kenya (Kikoti, 2009; Istituto Oikos, 2011). The main sources of water to these semi-arid areas are the perennial Ngarenanyuki and Simba Rivers that respectively drain Mt. Meru (ANAPA) and Mt. Kilimanjaro (KINAPA). However, water from these rivers is currently excessively extracted mainly for legume, vegetable and cereal crops irrigation farming in the upstream villages, and it now rarely reaches the downstream wildlife-rich areas during the dry season. In addition, there are a number of man-made water holes scattered in the communal and private lands, these supply water for the livestock and wildlife, however most of them are seasonal and contain water only during the wet season. The study area contains a number of charismatic wildlife species including African elephant (*Loxodonta Africana)*, plains zebra (*Equus quagga)*, African buffalo (*Syncerus caffer)*, wildebeest (*Connochaetes taurinus)*, Grant's gazelle (*Nanger granti)*, Thomson's gazelle (*Eudorcas thomsonii)*, giraffe (*Giraffa camelopardalis ssp. Tippelskirchi)*, lesser kudu (*Tragelaphus imberbis)*, striped hyena (*Hyaena hyaena),* leopard (*Panthera pardus)*, common duiker (*Sylvicapra grimmia)*, bush buck (*Tragelaphus sylvaticus)*, warthog (*Phacochoerus africanus)*, waterbuck (*Kobus ellipsiprymnus),* eland (*Taurotragus oryx),* black and white colobous monkey (*Colobus guereza)*, blue monkey (*Cercopithecus mitis)*, and olive baboon (*Papio anubis)* (Kikoti, 2009; Istituto Oikos, 2011; Elisa et al., 2016).

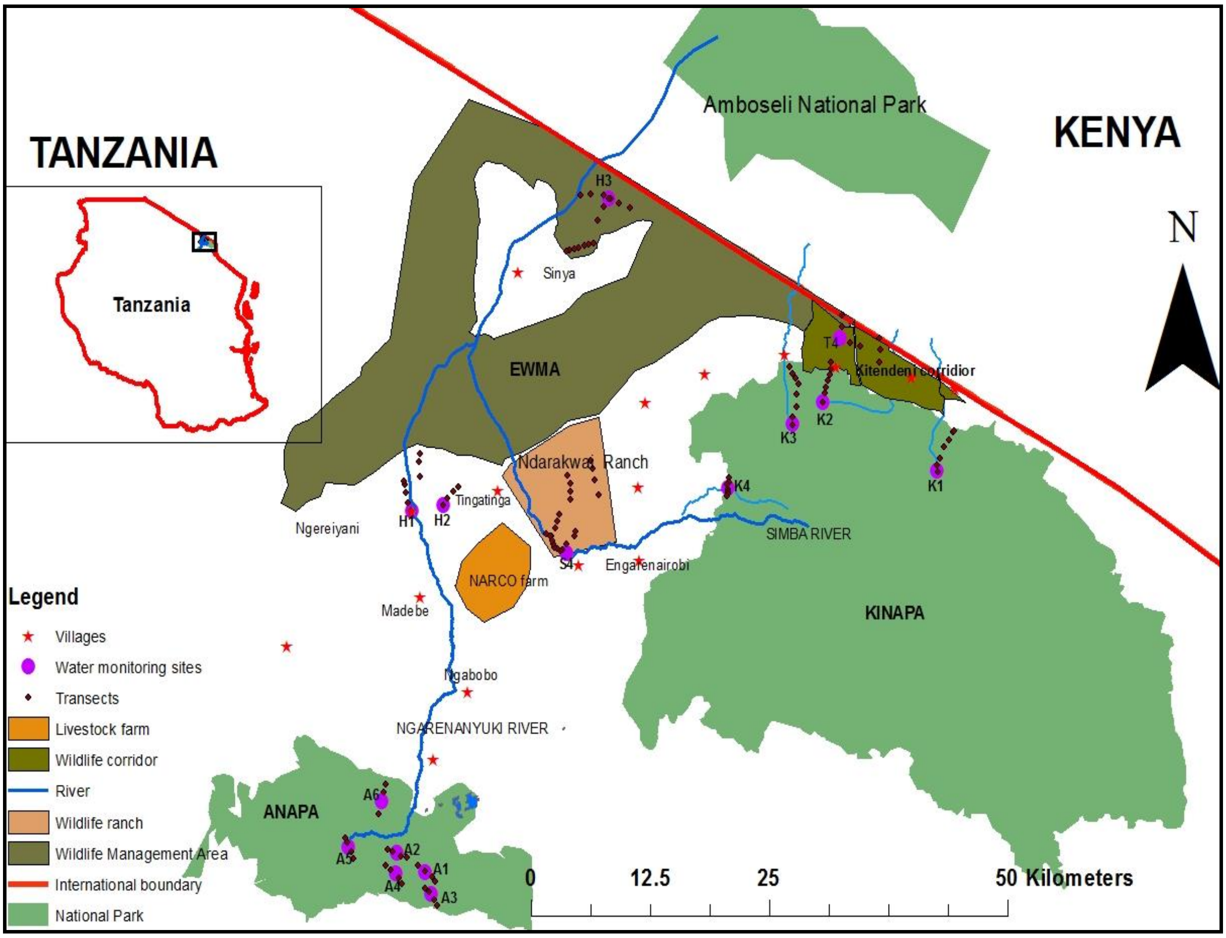

Figure 4. 1: Map of the study area and sites.

4.2.2 Data collection

Herbivore surveys in relation to surface water availability were conducted in two main areas; (i) upstream mountainous National Parks (KINAPA and ANAPA) and (ii) lowland semi-arid areas (EWMA, Ndarakwai wildlife ranch, Ngereiyani village, and Kitendeni corridor) (Figure 4.1).

4.2.2.1 Impact of water extraction on herbivores in National Parks

Transect location and characteristics in National Parks

Herbivore surveys were carried out during dry and wet seasons. September to October 2018, January 2019, and July to Sept 2019 recorded total monthly rainfall less than 50 mm, and therefore were categorised as dry season in ANAPA. The other months recorded monthly rainfall larger than 50 mm and they were thus categorised as wet season in ANAPA.

In KINAPA, only three months (October 2018, June 2019 and October 2019) recorded a total monthly rainfall larger than 50 mm and were therefore categorised as wet season. The other months were categorised as dry season.

In ANAPA, herbivore survey was conducted in 6 transects all of which were mainly dominated by forest habitat. Transects surveyed were: Mweka (A2), Kilinga (A4), Ngongongare 1(A1), Ngongongare 3 (A3), Mwakilenga (A6) and Malemeo (A5) (Figure 4.1). These transects ran along the rivers/streams at the extraction sites and the length of each transect was 1 km to ensure counting of only those wild animals that were obviously associated with water sources. In KINAPA, herbivore survey was conducted in 4 transects also characterised by forest habitat. The transects were: Londorosi (K4) 1km, Lerangwa (K3) 5.23 km, and Kitendeni (K2) 3.67 km, and Kamwanga (K1) 4.17 km. The herbivores were surveyed downstream of the extraction points, as upstream of the extraction points the areas had high riverbanks, thick forest, and steep gradients, making it difficult for people and wild animals to walk along the transect. Water sources in KINAPA were relatively large rivers where the existing extraction took about 70% of the available water, and the remaining water flowed only over a short downstream distance (<0.5km) from the extraction points during the dry season. Therefore, in KINAPA, the herbivore survey was confined along the water extraction pipelines where wild animals accessed water from the available open chambers and pools during the dry season.

Counting of wild herbivores in the National Parks

Ground counts for herbivores in KINAPA and ANAPA were conducted along extracted watercourses during the dry and wet seasons on a monthly basis over a period of six months from 2018 to 2020. The count was carried out mostly in the morning between 07:00 and 11:00 and few times in the afternoon between 15:00 and 17:30. Animals were active at both times. As logistically it was not possible to cover all transects work in one day, transects survey was carried out on consecutive days. The survey team walked the transects along the watercourses (as shown above) within the parks and counted and recorded number, species of animal and type of habitat. Due to poor visibility in the forested habitats, the maximum distance from the transect line within which animals were counted was 100 m. GPS coordinates of the sighted individual or group were also recorded (Brennun et al., 2002). Distance from the transect line to each animal or group of animal was measured with a laser range finder. Animals observed 100 m around the water extraction site was considered to be on the upstream side in ANAPA. To evaluate the impact of surface water on herbivore abundance and distribution, water quantity (i.e. discharge) and quality were measured at two points located up-and downstream of the extraction sites as described in details in the previous Chapters 2 and 3. In addition, surface water points (pools and open water chambers) along the extraction pipeline were identified and georeferenced to aid in calculating the nearest distance between animal observation and water points in KINAPA. The animals encountered during the survey in the parks were: African elephant, African buffalo, giraffe, red duiker; *Cephalophus natalensis*, bush buck, warthog, kirk's dik dik; *Madoqua kirkii,* waterbuck, black and white colobous monkey, blue monkey, and olive baboon.

4.2.2.2 Impact of surface water on herbivores in the semi-arid areas of West Kilimanjaro

Transect location and characteristics in the semi-arid areas

To examine the impacts of changes in surface water in the dry west Kilimanjaro areas, 13 ground transects were established and counts conducted monthly for a total of 15 months period covering both dry and wet seasons from 2018 to 2020. The months were categorised as dry or wet based on the monthly rainfall recorded at the Amboseli baboon research centre (Altmann & Alberts, 2020) which was the nearest (hence representative) rainfall station with consistent long-term data. September 2018 to April 2019, and July 2019 to September 2019 were categorised as dry season, and May and June 2019, and October 2019 to April 2020 were categorised as wet season. A month was categorised as wet season if its total rainfall was at least 1.5 of the mean monthly rainfall from 1997 to 2018, otherwise it was categorised as dry season. The lowland semi-arid area receives far less rainfall compared to the two mountaineous National Parks, and hence it was deemed necessary to use two different ways in categorising between wet and dry months in these areas. The mean monthly rainfall over this 42-year period in the semi-arid area, ranged from almost 0 mm to almost 70 mm. Transects were conducted in four wildlife areas with different management which were; Enduimet Wildlife Management Area (EWMA), Ndarakwai wildlife ranch, Kitendeni wildlife corridor and Ngereiyani village grazing land. The selection of these study sites in the semi-arid wildlife areas was mainly based on the locations and the status of surface water availability and distribution of the wild herbivores. In the semi-arid land, there were only four areas that had relatively reliable water availability and wildlife throughout the year. As livestock might influence herbivores access to water (De Leeuw et al.,2001), they were also taken into consideration when selecting the study sites. The four wildlife areas, which were at least 20 km from one another, were each assigned 3 transects (spaced 1.5 km apart) except Ndarakwai wildlife ranch that had 4 transects with one running along the Simba River. Based on the height, nature of the canopy, and percentage cover of woody species as described by Mengist (2019), transect habitat types were categorised as wooded grassland, grassland, bush land and thicket. In the semi-arid areas, transects were 2 km in length and running directly from the water sources, except the one transect in Ndarakwai wildlife ranch which ran along the Simba River that retained water during the dry season and thus provided a supplementary opportunity for the comparison of impacts of the change in surface water on herbivore abundance between wet and dry season periods. In each of the four areas, there was one control transect located 5 km away from the available water source. Water quantity (i.e. volume and discharge) and quality in these study sites were measured as described in details in the previous Chapters 2 and 3 to aid in the assessment of the impacts of water on the wild animals. Salinity was taken as an important water quality parameter due to its known impacts on the herbivores space use in similar wildlife ecosystems of Tanzania (Gereta and Wolanski, 1998; Gereta et al., 2004). Water salinity is also a limiting factor that may affect drinking and eating behaviour of herbivores. For instance, the Government of Western Australia (GWA) acknowledges salinity as the most important water quality limitation factor for livestock production (GWA, 2021). Surface water with salinity level above 500 ppm was categorised as saline water while water with lesser salinity was categorised as freshwater (Groundwater Foundation, 2018). Other water quality parameters such as nutrients, heavy metals and water hardness, might also influence herbivores abundance and distribution, but they were not accommodated in the herbivores modelling due to insufficient data as they were only measured once per each season.

Counting of herbivores in the semi-arid areas

The ground count of animals was carried out along the transects to quantify mammals species in relation to available water sources that included rivers and man-made (water-holes and watering troughs) water sources. Like in the National Parks, the count of herbivores in the semi-arid areas was also carried out between 07:00 and 11:00 in the morning and between 15:00 and 17:30 in the afternoon when many animals are active. The survey team walked quietly along the transects recording the number, species of animals on each side of the transect, and the dominant habitat type. The maximum distance from the transect line within which animals were counted was 500 m (Caro, 1999). GPS coordinates for each sighted individual animal or group of animals were recorded respectively using a hand-held GPS unit. All water sources were geo-referenced to facilitate mapping and the computation of animal distances to the water sources. The counting of livestock (cattle and sheep) along the transects was also simultaneously conducted, this was important as the livestock may affect the distribution and the space use of the wild animals (De Leeuw et al., 2001).

Wild herbivore species commonly observed in this semi-arid area were: African elephant, plain zebra, African buffalo, wildebeest, Grant's gazelle, Thomson's gazelle, giraffe, lesser kudu, eland, waterbuck, and impala. On the other hand, livestock species counted were cattle, and sheep and goat (combined in this study as sheep). Domestic donkeys were also observed but due to their low numbers they are not included in this study.

In addition, supplementary data on the past aerial wildlife surveys in the study ecosystem were obtained from Tanzania Wildlife Research Institute (TAWIRI). These geo-referenced survey data covered both dry and wet seasons for 2010 and 2013 respectively.

4.2.3 Data analysis

To provide a quantified over-view of the dry season herbivores population in the different surveyed areas in the Kilimanjaro landscape, animal density was derived by deviding the number of individuals observed in the dry season for each species by transect area (km^2) of each transect. Area of the transect was computed by multiplying effective strip width of each transect by the length of the transect. Then mean density was calculated for each species across all transects falling under each area category, i.e. ANAPA, KINAPA and West Kilimanjaro. The mean densities for each species in each area category were then presented in a bar chart (Figure 4.2). To quantify the impacts of the changes in surface water on wild herbivores, count data were analysed to compare spatial and temporal variation in herbivore abundance and distribution in relation to the existing water sources. An emphasis was given to the dry season as it represents the period when water is relatively scarce and hence a limiting factor to herbivores. ArcGIS (version 10.4.1) was used to map the study area, illustrate the study sites, and compute the distance between each herbivore sighting and nearest available water source (distance to water), using the Euclidean distance function. Past aerial wildlife survey data were processed in the ArcGIS by mapping and comparing species distribution in relation to water sources during both dry and wet seasons. The Sinya water hole (H3) in the EWMA was selected as a representative water source due to its year-round availability and its location within a wildlife-rich community area. Using the function 'Buffer' in the ArcGIS the water hole was buffered at a radius of 10 km and then using the function 'Clip' this buffer layer was used to clip out the aerial survey data separately for April 2013 and October 2010 which respectively represented wet and dry season wildlife surveys. Finally, animals sighting distances to the nearest water source were computed using the function 'Near'. Then the output layers for each survey were separately mapped and graphed to visually display distance to water distribution frequency of the selected herbivores in both dry and wet seasons. For the ground transect data, initial processing/formatting was mainly done using Microsoft excel spreadsheet. This involved data wrangling to enrich the data, where errors were eliminated and data organised under specific groups/categories to accommodate all relevant variables. This ensured an appropriate data format suited to the analysis R statistical programming environment, version 3.6.1 (R Core Team, 2019). Species count was taken as a response variable across both spatial and temporal scales. Predictor variables (main effects) were: animal's distance to water (m), water quality (defined as fresh or saline based on salinity), species, season (categorised as wet and dry), wild-domestic (by which animals species were categorised as livestock or wild species), water availability, and habitat type (categorised as forest, wooded grassland, grassland, bush land and thicket). As transects might have some stochastic effects on the wild animals, transect identity was included as a random effect in all mixed effect model analyses. Survey data from the National Parks were also coded, and analysed in the similar way where the response variable was the species count and the predictor variables were: distance to water, species, transects, treatment (upstream and downstream of the extraction points), seasons (wet and dry), and habitats.

To examine the impact of predictor variables on response variable, survey data were subjected to generalised linear mixed model with a Poisson distribution using function glmer or glmmTMB in R. This is the appropriate linear regression model for count data due to its capability to deal with non-normal errors and autocorrelaton issues in the dependent variables with repeated measures (Brooks et al., 2017; Crawley, 2005). As some of the count data was zero-inflated (resulting from a number of counts that didn't observe presence of an animal) and thus characterised with dispersion, the specifically zero-inflated linear mixed effect model with Poisson distribution was employed using the package 'glmmTMB' (Brooks et al., 2017). This is a model suited to handle zero-inflated data from counts conducted over time in a given location (Denes et al., 2015; Sebatjane et al., 2019). The regression analysis was also used to examine how wild herbivore and livestock abundance varied with; distance to water availability, change in water quality, habitat, seasons, and livestock presence in the West Kilimanjaro area. Further, the transect data for the semi-arid West Kilimanjaro were aggregated using the R-function 'aggregate' and then subjected to a generalised linear mixed effect model which was fitted using the function 'glmer'. This enabled a comparison of the abundance of herbivores and livestock between areas adjacent to and far from the water sources.

Model checking and diagnostic involved plotting of residuals against predicted values and standard normal deviates to respectively inspect for non-constancy of variance and non-normality of errors (Crawley, 2005). In addition, likelihood ratio test was used to compare between restrictive and non-restrictive Poisson models for selection of a model with a relatively high goodness of fit and containing only necessary effects to comply with parsimony principle (Crawley, 2005; UCLA, 2020). Further, inferences for the superior model were generated by running type III anova from package 'car' in R (UCLA, 2020). A comparison of the Akaike Information Criterion (AIC) was also used to examine the effects of various model factors and hence to select the best models (Dziak et al., 2020), in which case the models with the smallest AIC values were chosen (Stommel, 2016).

4.3 Results

Herbivores density across species and area

Figure 4.2 gives an overview of the dry season herbivore density (individual/km^2) of commonly sighted herbivores in all survey areas in the Kilimanjaro landscape. Animal density varied across areas and species. Zebra, wildebeest, gazelle and impala species were only found in the semi-arid areas in West Kilimanjaro, whereas duiker and bushbuck were only observed in the parks. KINAPA recorded the highest density of elephants and buffaloes in the landscape. Livestock (cattle and sheep) recorded one of the highest herbivores density and were only observed in the West Kilimanjaro region. There were no livestock in KINAPA and ANAPA as these areas are strictly protected and livestock grazing is prohibited by law.

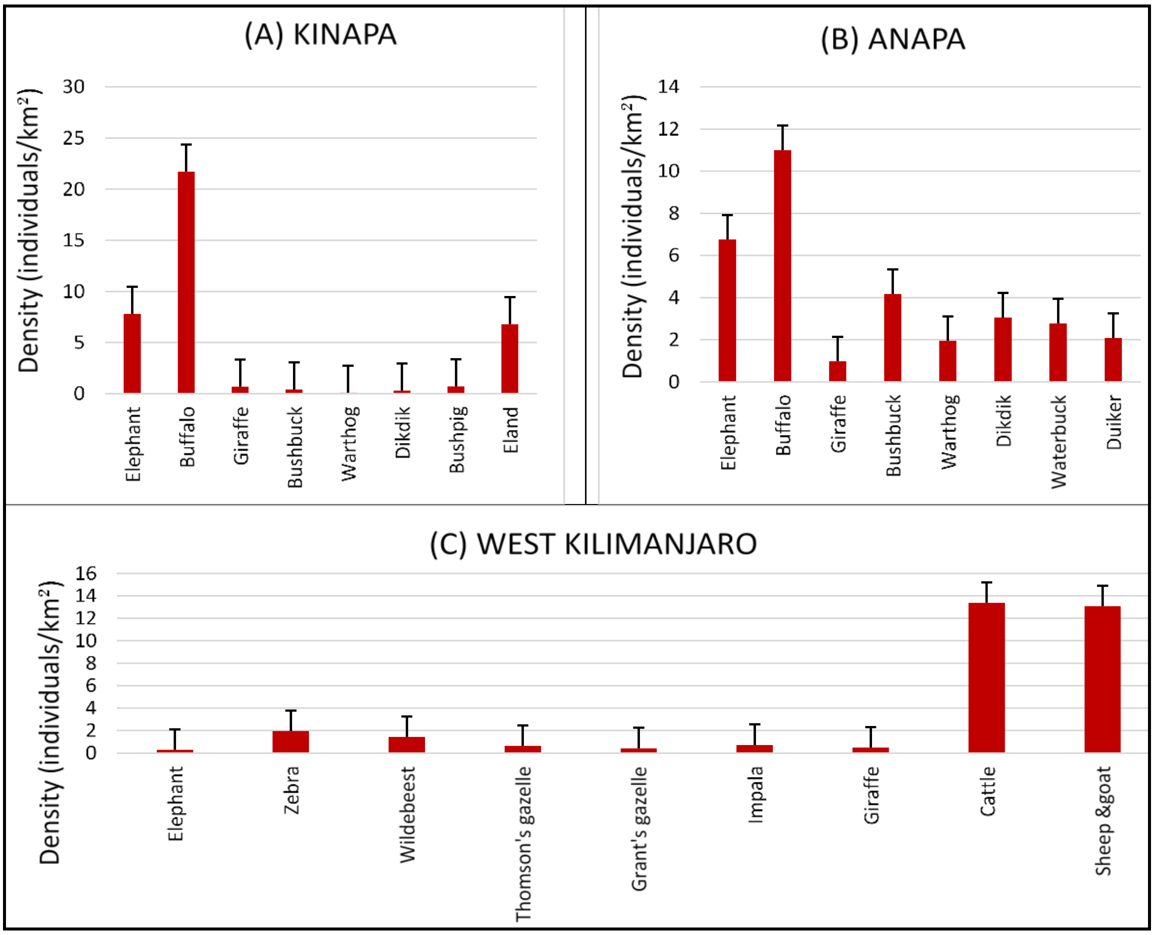

Figure 4. 2: Mean herbivore density for (A) KINAPA (±SE, n=16), (B) ANAPA (±SE, n=12) and (C) West Kilimanajro (±SE, n=108) during the dry season.

4.3.1 ANAPA and KINAPA

Outputs from a zero-inflated generalised linear mixed effects model with interaction effects between species and seasons, and treatment and seasons, and transect ID as a random effect, revealed variations in herbivore abundance across space and time in ANAPA (Table 4.1 and Figure 4.3). Further, a comparison between the full model and the null model, revealed that interaction effect (Species*season) significantly affected species abundance ($X^2(7)$=36.78, p= 5.162e^{-06}. This model was also robust, as the visual inspection of the residual plots did not reveal any heteroscedasticity or deviations from normality.

Table 4. 1: Analysis of Deviance (Type III Wald chisquare tests) based on herbivore abundance as a response factor in ANAPA.

	Estimate	Std.error	Chisq	Df	Pr(>Chisq)
(Intercept)	1.569	0.157	99.731	1	0.000
Species	-1.177	0.325	60.232	7	0.000
Season	-0.247	0.154	2.570	1	0.109
Treatment(up-downstream of extraction)	-0.244	0.098	6.149	1	0.013
Species:Season	-0.215	0.503	35.840	7	0.000

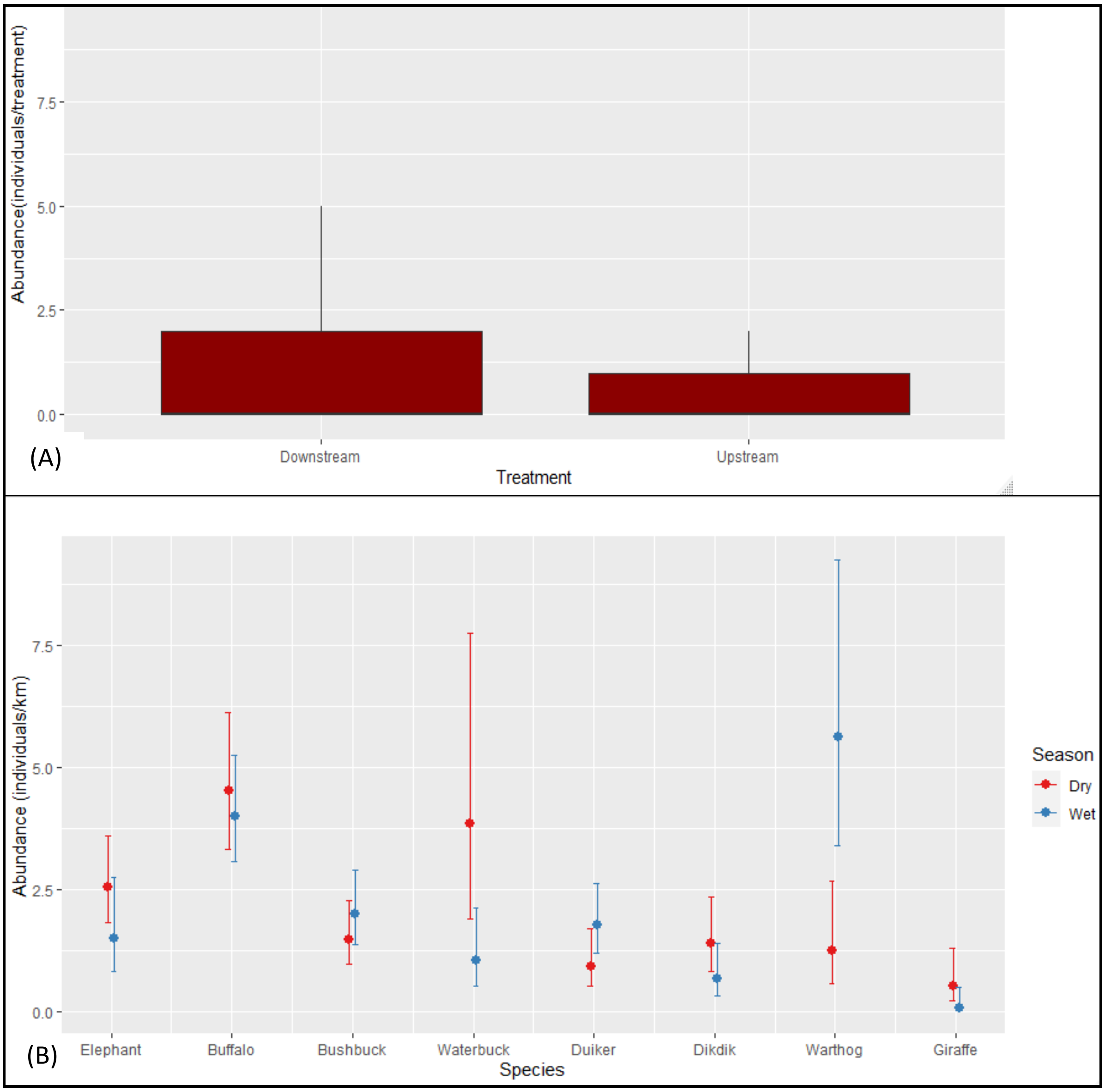

Figure 4. 3: Variation of herbivore abundance by (A) treatment and (B) by species and season effects along the extracted water sources in ANAPA.

During the dry season, species maintained relatively high and stable abundance. In contrast, species abundance was more variable during the wet season (SE±0.5, n=8) and the extent of variations differed between species (Figure 4.3 B). Further, results from the same model which considered buffalo as a reference species, revealed that species abundance in ANAPA declined in the upstream (Figure 4.3A and 4.4). Overall, species abundance showed a weak decline during the wet season. However, the abundance of some species in particular giraffe and waterbuck strongly declined during the wet season. By contrast and supringly, warthog increased during the wet season (Figure 4.4).

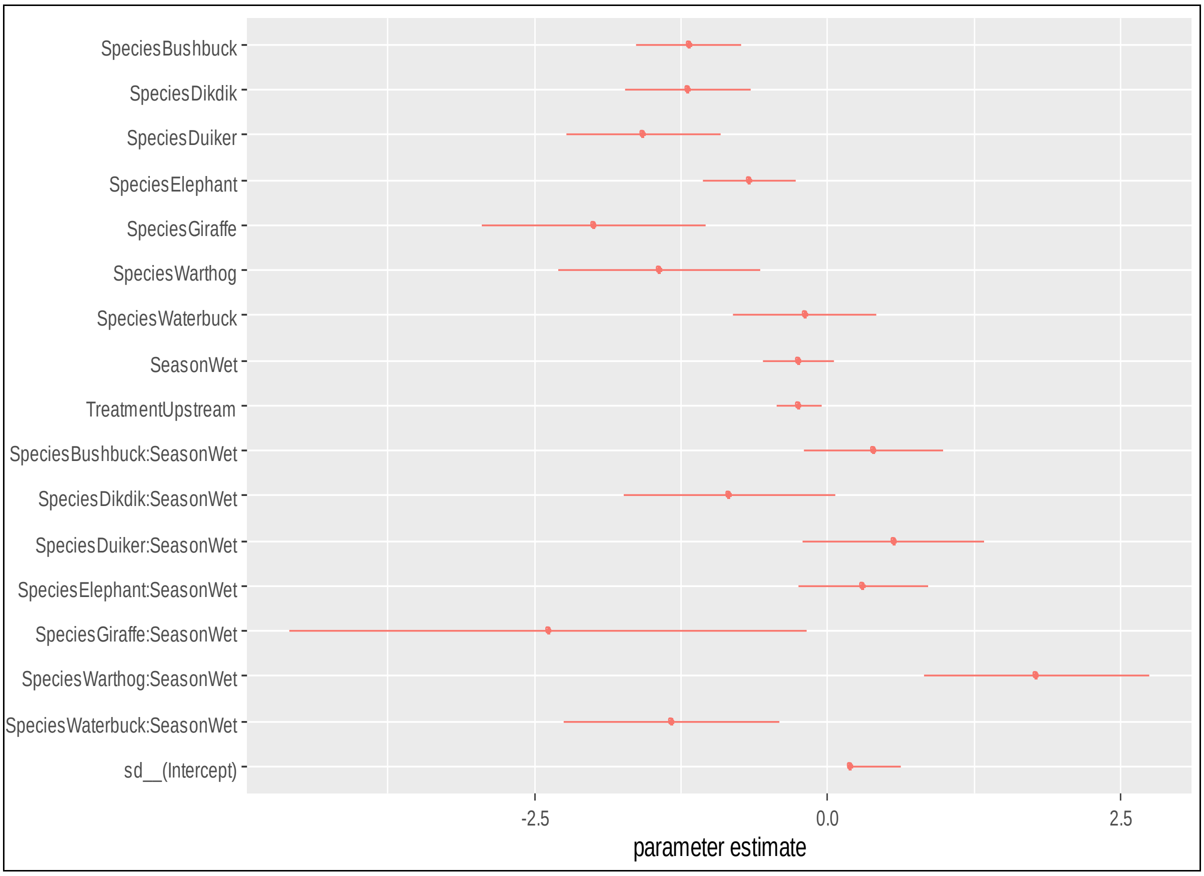

Figure 4. 4: A plot of main effects estimate on herbivore abundance as response in ANAPA. Parameter estimates represent the marginal difference for each species in each season.

The surveys in KINAPA revealed that herbivore abundance and distribution varied across species, season, surface water availability and altitude (Figure 4.5 and Table 4.2). While buffaloes (a reference species in the model) and elephants were relatively closely associated with the available water sources, the abundance of eland and dik-dik strongly increased with an increase in distance from the water sources (Figure 4.5B).

Overall, herbivores were more abundant close to water sources during the dry season, and their abundance declined more sharply with an increase in distance from the surface water (open chambers, leakages, and ponds) sources compared to the wet season (Figure 4.5A). While overall herbivore abundance declined during the wet season, it increased with an increase in distance from the water sources (Table 4.2).

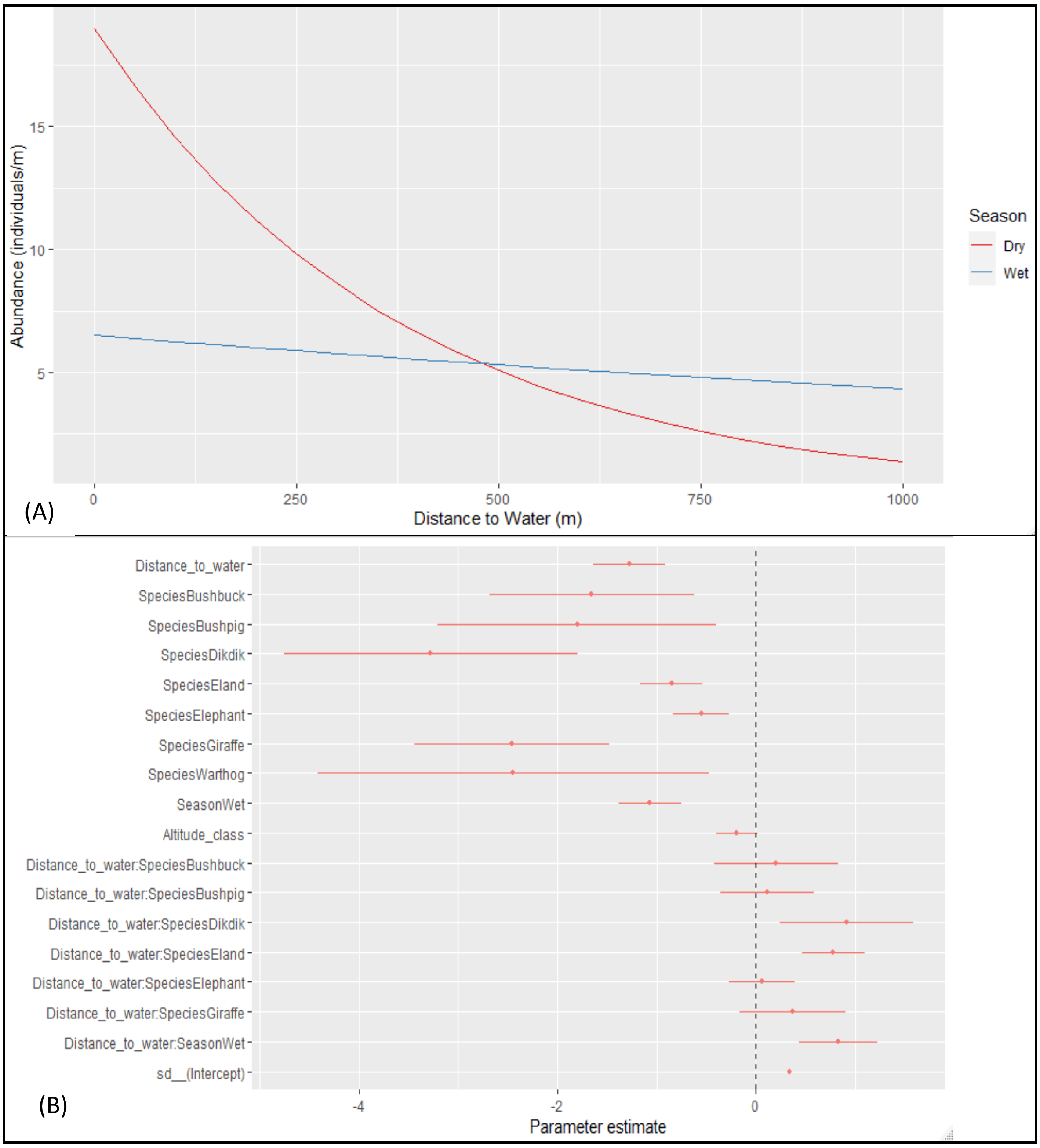

Figure 4. 5: (A) Herbivore abundance in response to the effect of distance to surface water under the conditional effect of seasons, and (B) main effect estimates on herbivore abundance along the water extraction pipeline in KINAPA. Parameter estimates represent the marginal difference for each species in each season.

Table 4. 2: Analysis of Deviance (Type III Wald chi-square tests) based on species abundance as a response factor in KINAPA.

	Estimate	Std.Error	Chisq	Df	P value
Distance to water	-0.003	0.0004	47.2622	1	<0.001
Species	-1.864	0.544	139.39	7	<0.001
Wet season	-1.068	0.1600	14.87	1	<0.001
Altitude	-0.0003	0.0002	3.40	1	>0.05
Distance to water:Species	0.001	0.001	29.869	6	<0.001
Distance to water:Wet season	0.002	0.001	16.74	1	<0.001

4.3.2 West Kilimanjaro

Herbivore distribution and abundance largely varied with respect to surface water availability in the semi-arid West Kilimanjaro region. Overall, herbivores abundance was higher near permanent surface water during the dry season than in the wet season. Table 4.3 below presents results from a zero-inflated generalised linear mixed-effects model (glmmTMB) in the semi-arid West Kilimanjaro region. The model, which inspected factors affecting herbivores species abundance, revealed that, variation in herbivore abundance depended on: distance to water, species, season, distance to water by species, by season and by water quality (salinity). Using the likelihood ratio test, a comparison between the full model (with interactions) and the null model revealed that the full model was superior, and that the interaction effect significantly affected species abundance $\chi^2(10) = 605.52$, p= <0.001).

Table 4. 3: Analysis of Deviance (Type III Wald Chi square tests) based on herbivore abundance as a response factor during dry and wet seasons in West Kilimanjaro.

	Estimate	Std.Error	Chisq	Df	P value
(Intercept)	1.80E	0.46	15.34	1	<0.001
Distance to water (km)	-3.37E-04	1.40E-04	5.81	1	<0.05
Species	0.34	0.14	481.90	6	<0.001
Wet season	-1.09	0.005	447.25	1	<0.001
Habitat	0.34	0.58	1.58	2	>0.05
Water Quality	-0.046	0.06	0.56	1	>0.05
Distance to water:Species	3.88E-05	1.26E-04	162.62	6	<0.001
Distance to water: Wet season	9.67E-04	4.74E-05	415.65	1	<0.001
Distance to water:Habitat	-7.42E-05	1.11E-04	5.79	2	>0.05
Distance to water:quality (salinity)	3.24E-04	5.20E-05	38.80	1	<0.001

As shown in Tables 4.3 and 4.4, and Figure 4.6, surface water availability was the most important factor that affected herbivores abundance and distribution in the semi-arid areas. The model results of the herbivore counts along the Simba River in Ndarakwai Wildlife Ranch clearly indicated that herbivore abundance mainly depended on distance to water and season (Table 4.4). Here, herbivore abundance decreased with an increase in distance from the river, and also decreased in the wet season.

Table 4. 4: Analysis of Deviance (Type II Wald Chi square tests) based on herbivore abundance as a response factor during dry and wet seasons along the Simba River at Ndarakwai wildlife ranch.

	Estimate	Std. Error	Chisq	Pr(>|z|)
(Intercept)	2.547303	0.06091	41.821	< 0.001
Distance to water	-0.00216	0.000438	24.641	<0.001
Wet season	-0.42984	0.064218	48.557	<0.001

Overall, herbivore abundance in the West Kilimanjaro region decreased with an increase in distance to surface water, and also decreased with wet season. However, in the wet season herbivore abundance increased with an increase in distance from surface water sources. According to Figure 4.6 (elephant as a reference species), herbivore abundance varied across species, where some species increased towards the water sources while others declined. For instance, Grant’s gazelle strongly increased with an increase in distance from surface water sources. Whereas wildebeest and zebra strongly decreased with an increase in distance to water sources.

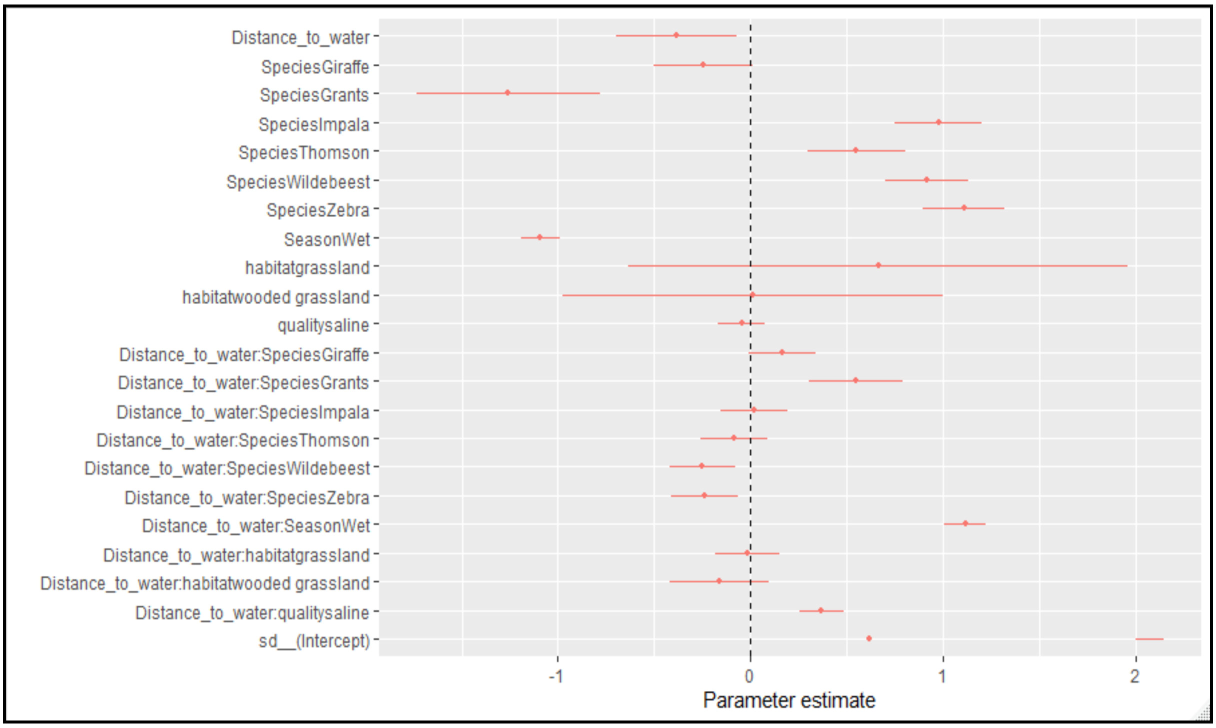

Figure 4. 6: A plot of main effect estimates as generated by zero inflated poisson generalised mixed-effects model of predictors affecting wild herbivore abundance during dry and wet seasons in the west Kilimanjaro ecosystem. Parameter estimates represent the marginal difference for each species in each season.

The results in Figure 4.7 show the conditional effects of water quality (salinity) on the distance to water effect on herbivore abundance. The result indicates that herbivores abundance increased with an increase in distance to saline waters in wet season.

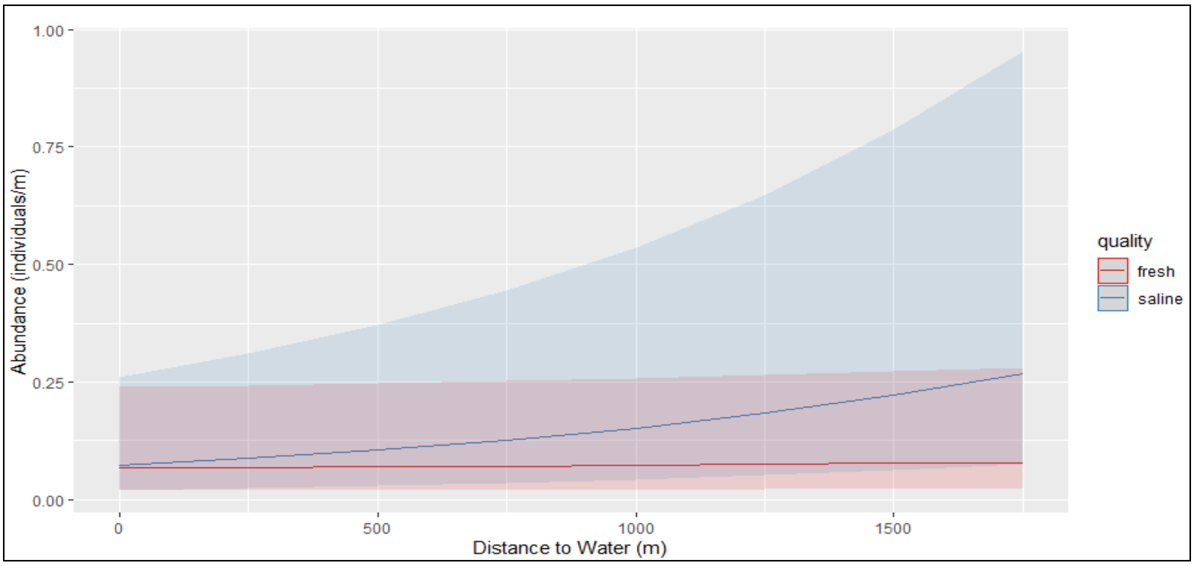

Figure 4. 7: Variation in wild herbivore abundance in response to distance to water by the effect of water-salinity in the West Kilimanjaro in wet season.

Figures 4.8 to 4.11 below present water- related spatial patterns in the abundance of domestic and wild herbivores (both water-dependent and water-independent) in the West Kilimanjaro region in the dry and wet seasons.

In the dry season, wildebeest (a water-dependent species) recorded higher abundance close to surface water, which gradually declined with an increase in distance from the water source (Figure 4.8A). On the other hand, wildebeest abundance increased with an increase in distance to water during the wet season (Figure 4.8 B). However, Grant's gazelle (a water-independent species) abundance increased gradually with an increasie in distance from water sources during both dry and wet seasons (Figure 4.8 A and B).

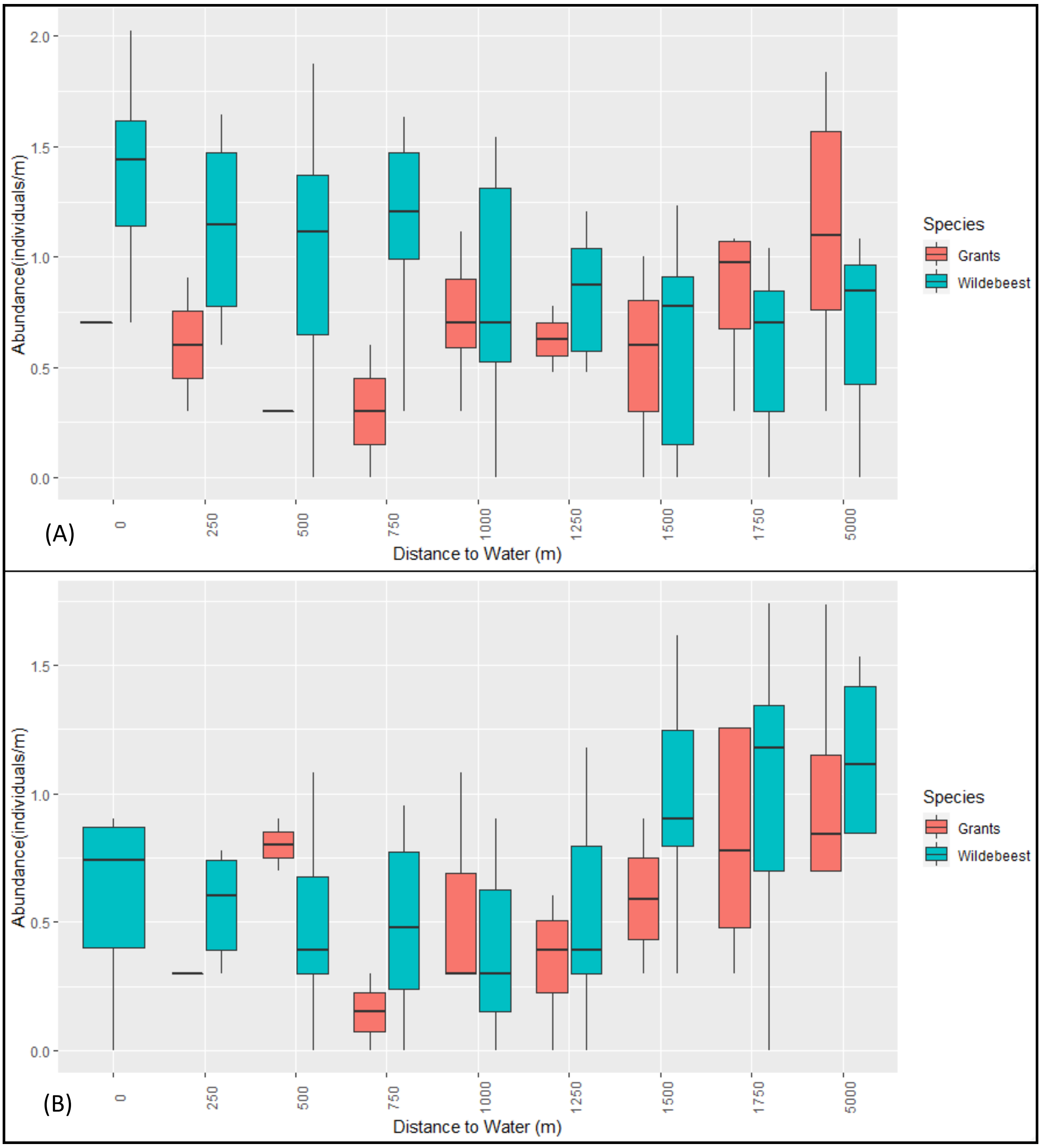

Figure 4. 8: Average abundance and distribution of wildebeest and Grant's gazelle with respect to surface water in West Kilimanjaro during (A) the dry season and (B) the wet season.

Figure 4.9 shows that the abundance of zebra (a water-dependent species) increased with decreasing distance to the surface water sources in the dry season but increased with increasing distance from surface water source in the wet season, where most of the animals were sighted at distances beyond 1 kilometre from the water sources.

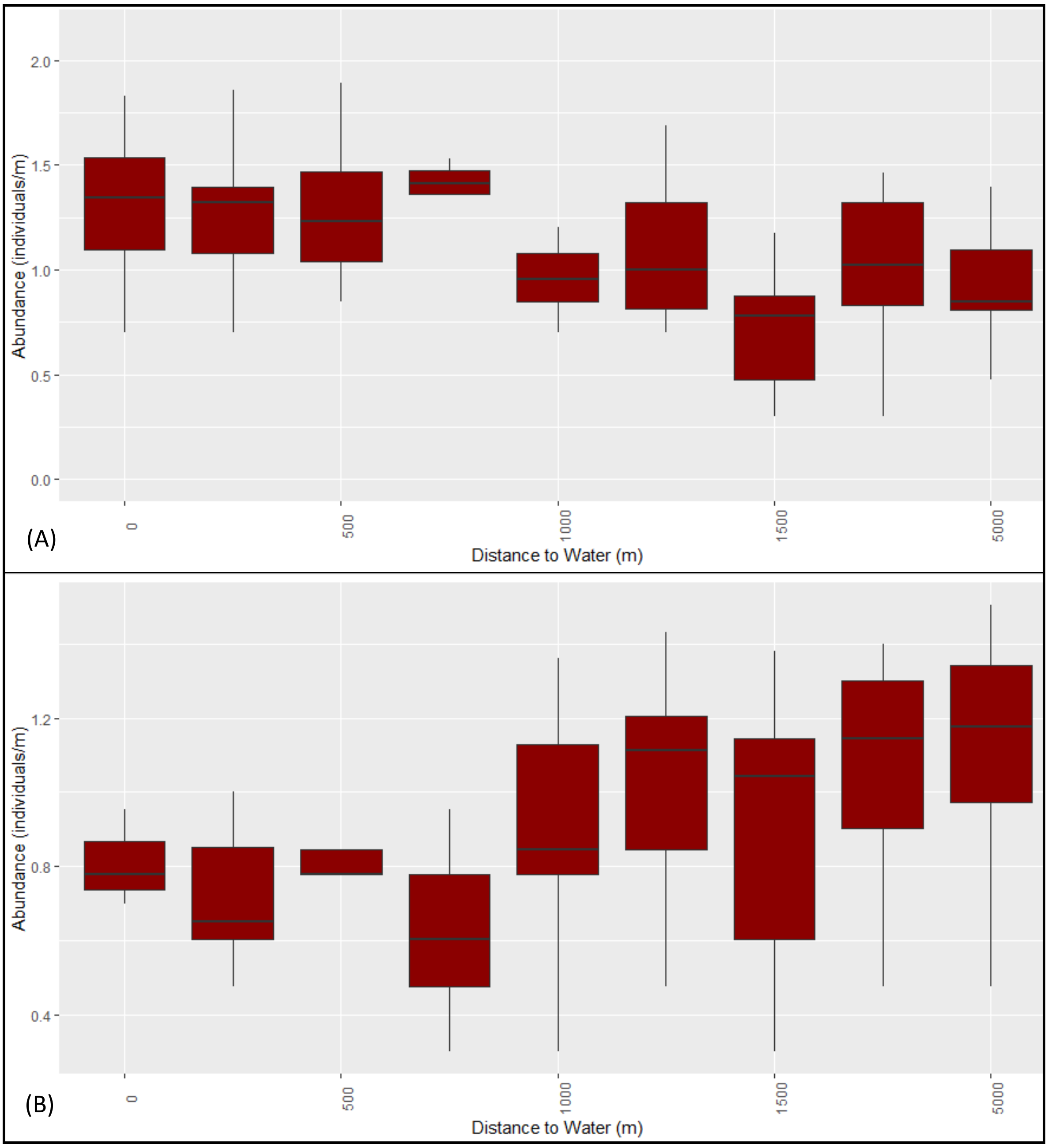

Figure 4. 9: Average abundance and distribution of plains zebra with respect to surface water during (A) dry season and (B) wet season in West Kilimanjaro.

Results in Figure 4.10 show that the abundance of elephants varied with distance from water sources during both the dry and wet seasons. There were more elephants close to water sources in the dry than in wet season in the semi-arid area of West Kilimanjaro. In the dry season, most of the elephants were sighted within 0 to 1 kilometre from the water sources. While in general, the elephant abundance increased with increasing proximity to water during the dry season, its spatial pattern appeared relatively irregular compared to other herbivore species. During the onset of wet season, the overall elephant abundance was relatively low and were then concentrated within the first 2 kilometres from the permanent water sources

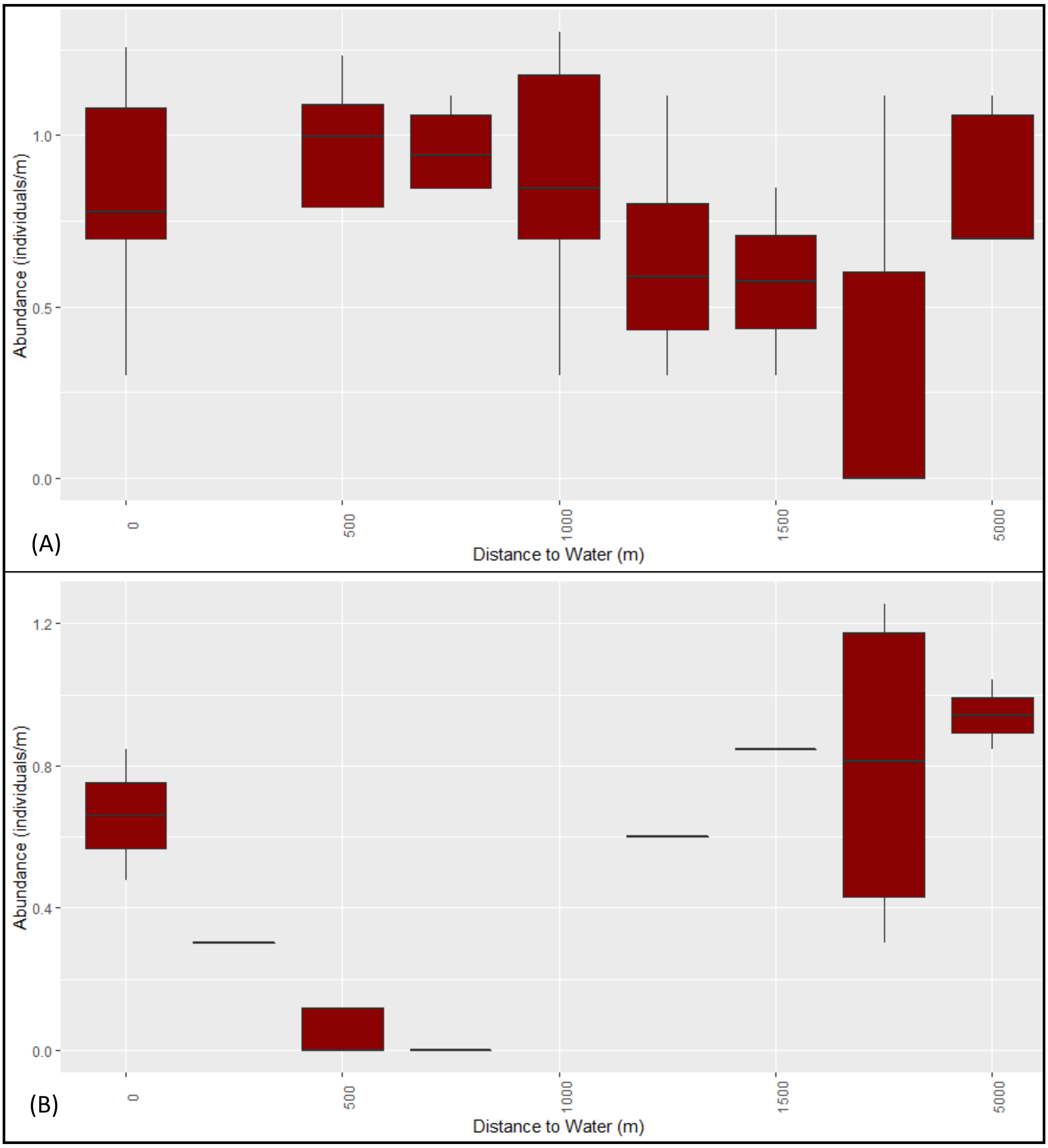

Figure 4. 10: Average abundance and distribution of elephants in relation to surface water availability in West Kilimanjaro during (A) dry season (B) wet season.

Figure 4.11 shows how the abundance of wild herbivores and livestock (cattle, sheep and goat) are inversely related with one another, and it also varies with distance from surface water and changes in water quality (salinity) with seasons. Overall, wild herbivores and livestock abundance was higher in the dry than in the wet season and increased with increasing proximity to the scarcely available water, even if the water was saline and mineral concentrated, suggesting that the animals may have no other water source. In Figure 4.11A, average livestock abundance in the West Kilimanjaro areas was higher (reaching an average of about 100 counts) than wild herbivores abundance whose highest average value was below 20 counts. However, according to a generalised mixed-effects model, abundance of both wild herbivores and livestock showed a similar pattern around water sources in the semi-arid areas in West Kilimanjaro. From Figure 4.11B, in the wet season, both wild herbivores and livestock declined around permanent freshwater and saline water. In contrast, during the dry season both wild and domestic herbivores consumed saline water as manifested by their substantial increase and high abundance around surface saline water sources. On the other hand, areas located far from water sources recorded highest animal abundance in the wet season but lowest animal abundance during the dry season.

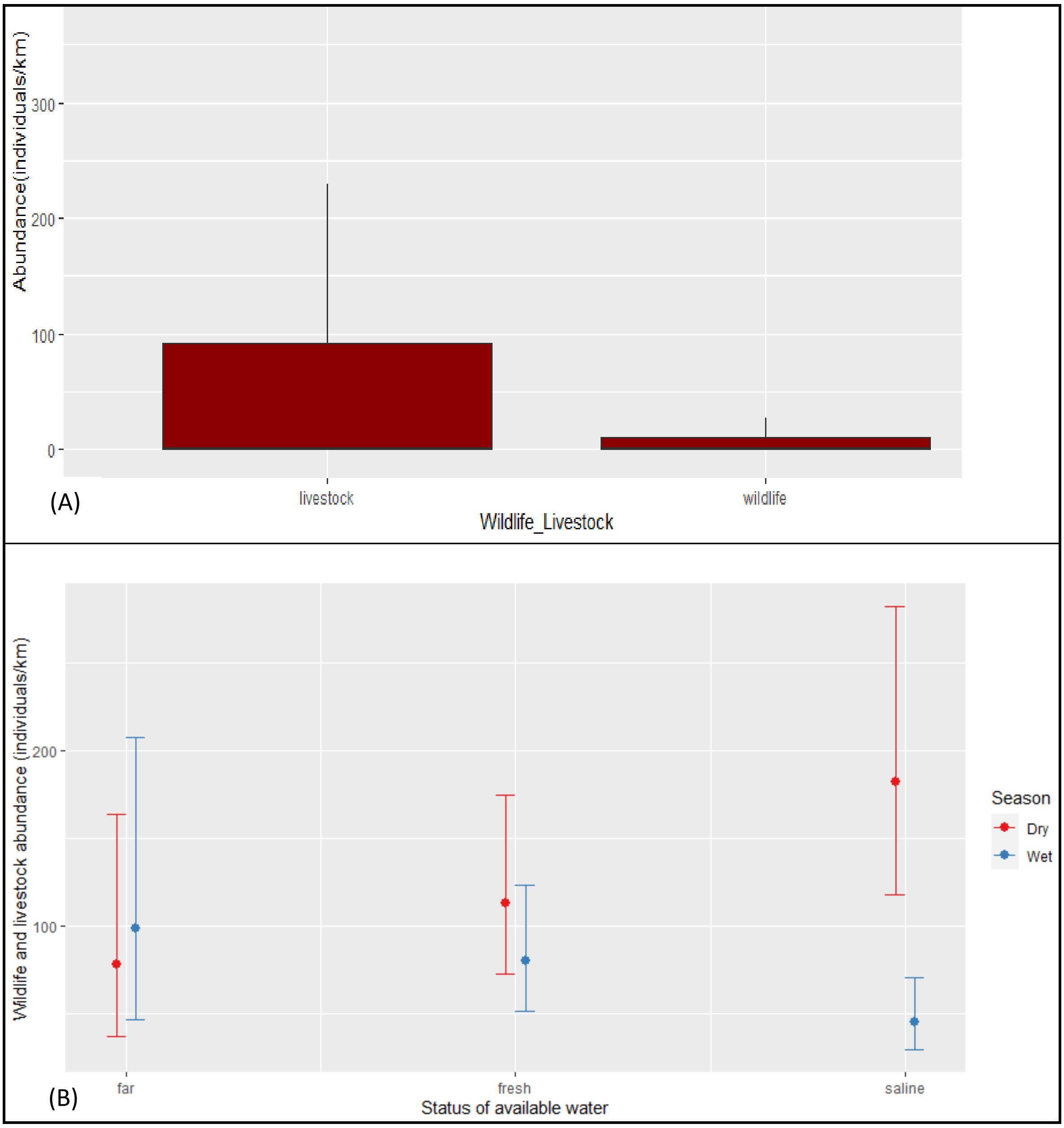

Figure 4. 11: Spatial variation in wild animal and livestock abundance in West Kilimanjaro as function of (A) wildlife-livestock effect and (B) status of available water by season effect. Note: status of available water is categorised as fresh, saline, and location (far) where animal has to move at least 5 km to reach the water source. The seasons are classified as either dry or wet.

Results in the Figure 4.12 show distribution patterns of wildebeests and zebras with respect to surface water sources as observed during the aerial surveys in dry and wet seasons in 2010 and 2013 in the West Kilimanjaro. It is evident that wildebeests and zebras were predominant within 0 to 4 km of the surface water sources during the dry season (Figure 4.12A) as compared to the wet season. During the wet season, these herbivore species were more common and more widely distributed away (around 10 km) from the permanent water sources (Figure 4.12B). This pattern is similar and compares well with the observations of this study.

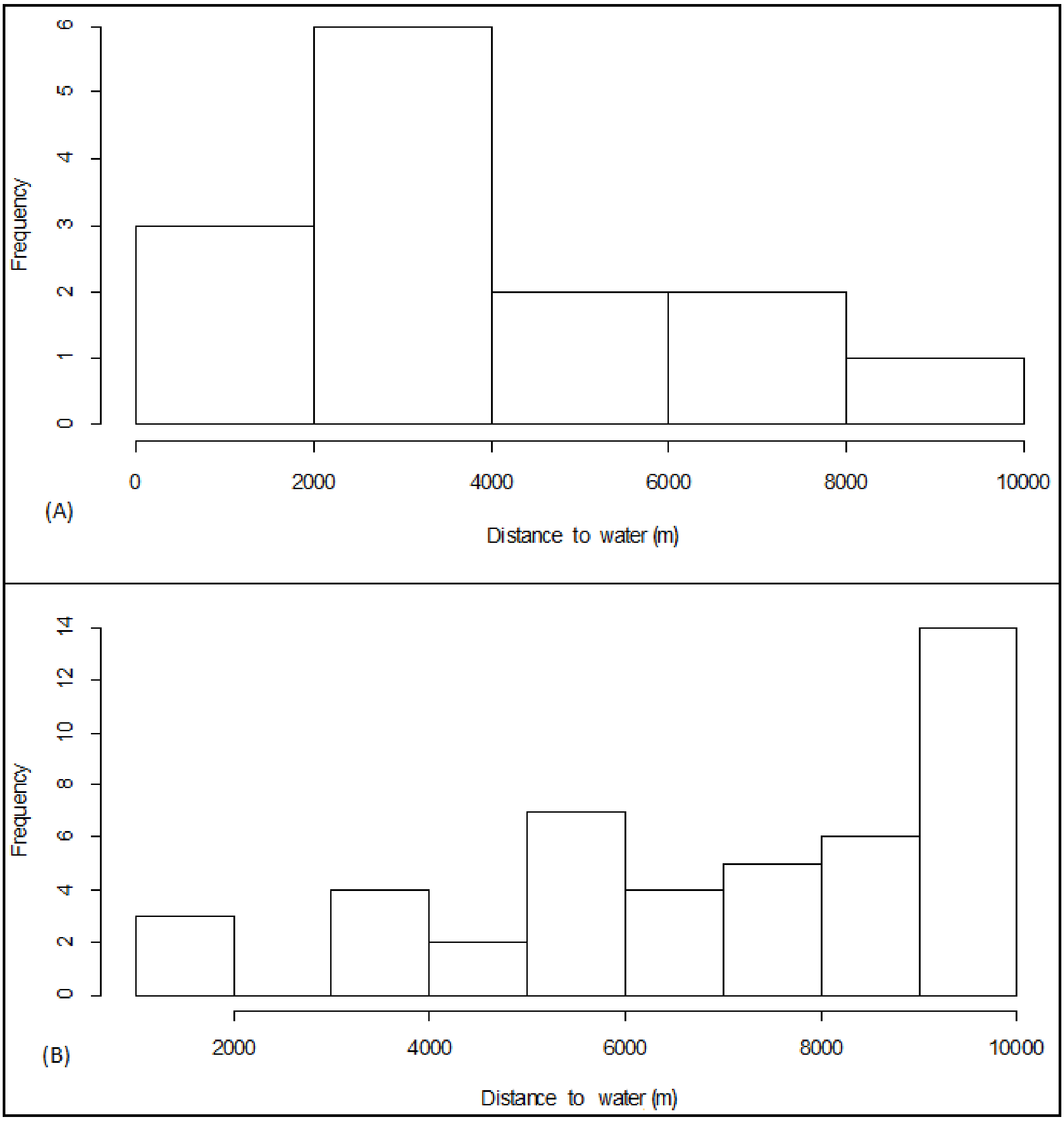

Figure 4. 12: Frequency distribution of wildebeests and zebras in a radius of 10 km around the Sinya water hole (H3) in EWMA during the (A) 2010 dry season and (B) 2013 wet season. Data source: (TAWIRI, 2010, 2013).

4.4 Discussion

4.4.1 ANAPA and KINAPA

It is evident that water related herbivores abundance and distribution varied across space, seasonality and species in both parks. Overall, the herbivore abundance near water sources increased in the dry season and declined in the wet season, and as expected, herbivore abundance was negatively associated with the distance to water in the dry season more so than in the wet season. However, this varied across species with some that were more sensitive to water availability in the dry season (e.g. wildebeest, zebra and elephants) than others (Grant's gazelle, giraffe and impala). Such dry season increase in herbivore abundance around permanent water sources has also been reported in the neighbouring protected ecosystems of Amboseli and Tsavo-Amboseli in Kenya (Kioko et al., 2006; Muruthi and Frohardt, 2006; Western, 1975). Further, Kikoti (2009) suggested that elephants distribution and movement were associated with surface water availability in the neighbouring semi-arid West Kilimanjaro region and that some elephants extensively used the Kitendeni corridor that links the northern side of KINAPA (where water extraction is taking place) and the Amboseli areas in southern Kenya. In this dry area, water is limited with respect to both quantity and quality, and therefore it is likely that some of the water dependent herbivores especially elephants and buffaloes do move to the upstream parks especially KINAPA to access good quality water and hence leading to the observed increase in the dry season abundance in the parks. Therefore, such increase in dry season herbivore abundance is likely reflecting a rise in water demand in the ecosystem where surface water was excessively extracted to support domestic, irrigation, and livestock use.

In addition to being associated with the available surface water, herbivores species in ANAPA recorded a relatively high abundance in the downstream areas (Figure 4.3) implying that they were also influenced by other environmental factors once the water need is satisfied. As existing water extraction in ANAPA took a high amount ranging from 50% to 90% of the available water (see Chapter 2), it would be expected to have higher abundance of water dependent species in the upstream where much more water was available than in the downstream areas that were left with comparatively smaller amount of water after extraction. Elisa et al. (2016) also reported an upstream skewed large herbivore abundance in 2012/2013. However, during that study, there was relatively low amount of surface water following a series of below average annual rainfall recorded in the year of the study and several previous years (see Chapter 2). In addition, there was also a comparatively high water extraction as all the available water was extracted during the dry season. In contrast, during this study there was a relatively high amount of surface water in the wildlife rich areas of ANAPA and the water was not completely extracted so that there was always some water flowing downstream for most of the dry season period.

Water is a dominant driving factor of space use, especially for water dependent species as they often select their habitat based on regular access to water (McNaughton & Georgiadis, 1986; Omphile & Powell, 2002). As in this study, water was always available at both up-and downstream sites, the higher downstream wild herbivore abundance was likely due to both surface water and other environmental factors such as habitat preference. For instance, in KINAPA, most low altitude downstream areas in the parks were largely characterised by an abundant grass, herbs, bushes/thickets, and less steep terrain making them a suitable habitat for several herbivores species such as buffaloes (Van Wieren & Van Langevelde, 2008).

A likely implication for park management of these findings is that water supply must be maintained in the downstream areas, i.e. not all water may be extracted, in order to maintain biodiversity conservation in the long-term. Thus, the current water extraction regime allowing all water to be extracted in the parks is not ecologically sustainable. In addition, the animal water provision by some open chambers along the water pipelines in KINAPA is inadequate, and as a result, there were several cases of water infrastructures damaged by the wild animals. Based on the animal density and the visual observation, the herbivores' water demand was probably higher in KINAPA than in ANAPA, as noted in the excessive trampling and resultant bare soil around the surface water sources along the water extraction pipelines (Figure 4.13). This was further intensified by dry season movement of herbivores especially elephants and buffalo from the lowland semi-arid areas through Kitendeni wildlife corridor to the park (KINAPA) where they accessed good quality water.

Figure 4. 13: (A) Bare land around an open water chamber due to trampling and (B) leakage of a water pipe broken by wild animals searching for water in KINAPA.

Dry season excessive water extraction from the existing water sources in KINAPA has dried up the rivers both inside and outside the park. Consequently, the animals downstream of the extraction points can only access water through occasional open chambers and leakages along the water extraction pipelines. Besides, the existing provision of water along the extraction pipelines in KINAPA is possibly not intended to fully cater for the animals' water needs but rather protection of the water infrastructures and ensuring of un-interrupted water supply to the neighbouring human communities. Such water provision helps to reduce infrastructure damages by the wild animals, especially from elephants looking for water during the dry season. Unfortunately, there were only few and poorly located open water chambers, and therefore animals particularly elephants frequently broke down the water pipelines in searching for quality and sufficient water. For instance, Kamwanga pipeline (4.17 km) that provided access to water for animals in only two chambers, had 5 fractures, while Lerangwa (5.23 km) that provided access to water in 6 chambers had 3 leakages, all as a result of pipe damaged by elephants. Damage of water facilities by elephants in the downstream semi-arid West Kilimanjaro region is also reported by Mariki et al.(2015). Elephants might also be motivated to break the water infrastructures in search for quality water when the stagnant, trampled and defecated water becomes of poor quality often due to bacterial load (Ramey et al., 2013). Dry season, movement of elephants from the semi-arid West Kilimanjaro to KINAPA and their concentration around fresh water sources available in the park, might be a reflection of elephants' adaptation in acquiring quality water, as during the dry season, most of the waters in the semi-arid West Kilimanjaro becomes saline (see Chapter 3) and probably bacteria loaded.

Herbivore abundance around water points in the parks also varied with species. Buffaloes recorded a relatively higher abundance around the existing surface water sources confirming affinity of this species to areas with permanent surface water sources and suitable habitat according to Sianga et al. (2017). Such water availability and favourable habitat might also explain the commonness of buffalo in the parks as compared to the surrounding semi-arid areas. The increasing abundance of eland and dikdik in KINAPA in areas away from surface water is possibly explained by the species' water-independence. These species obtain much of water through feeding and also have the ability to conserve water and therefore minimising the need to drink frequently (Owen-Smith, 1996; Redfern et al., 2003; Rubenstein, 2010; Van Wieren & Van Langevelde, 2008). Higher wet season warthog abundance in ANAPA (Figure 4.4), might be an indication of its high preference for forest glades which were common around water sources, and from which they feed on grass, sedge and herbs and are relatively safe from predators due to better visibility(Kahana et al., 2013). Surprisingly I did not observe wildebeests and zebras in KINAPA during the survey period. Local people and staff also report that these species are usually not present in KINAPA, however in ANAPA zebras exist mainly in the eastern and south-eastern areas where there are large patches of grass vegetation, and also plenty of surface waters. These water dependent species are common in the semi-arid West Kilimanjaro areas especially in the EWMA, Ndarakwai wildlife ranch and NARCO livestock ranch, and therefore they could access KINAPA through Kitendeni corridor, but they did not. Since these species are highly water-dependent (Knight et al., 1988; Rubenstein, 2010), their absence in the park might be linked to the lack of suitable habitat with preferred forage and sufficient visibility required for spotting and avoiding predators. Plains zebras and wildebeests usually prefer open and moderately dry habitats that are in close proximity to surface water and with reasonable grass abundance during the dry season (Rubenstein, 2010).

4.4.2 West Kilimanjaro

Similar to the situation observed in the parks, abundance and distribution of the herbivores in the semi-arid areas of West Kilimanjaro varied across species, space and time, mainly with respect to surface water availability. Overall, an increase in distance away from water sources was significantly($X^2(1, n=2)=5.81$, $p<0.05$) associated with a decrease in herbivore abundance during the dry season. However, in the wet season an increase in distance away from water sources was significantly ($X^2(1, n=2)= 415.65$, $p<0.001$) associated with an increase in the herbivore abundance, and the herbivores were widely distributed as surface water was available in most areas (Table 4.3). Similar findings were also reported by Western and Lindsay (1984) in the adjacent Amboseli ecosystem. Variation across species was also evident, for instance, the abundance and distribution of water-dependent species (wildebeests and zebras) were strongly controlled by the availability of water. Their abundance decreased with an increase in distance from surface water during the dry season, and increased away from the water sources during the wet season as water became available elsewhere (Figure 4.8 and 4.9). This observation is also in agreement with other studies (Knight et al., 1988; Stommel, 2016) which have demonstrated that zebras and wildebeests usually graze on low water content grass and therefore they are closely associated with surface water to meet their water requirements during the dry season. Being hindgut fermenters, zebras also have the capacity to rely on the low nutrient grass available in the often overgrazed areas around permanent surface water sources in the dry areas and therefore they must maintain daily access to drinking water (Rubenstein, 2010; Stommel, 2016). On the other hand, some species did not seem to be associated with surface water. Grant's gazelle for instance strongly increased with distance from water during both the dry and wet seasons (Figure 4.6 and 4.8). However, the abundance of impala and giraffe increased slightly with an increase in distance from water in the dry season, implying these species were only partially dependent on water from the available surface water sources (Figure 4.6). This lack of a clear association with water during the dry season can be explained by the species ability to obtain water from the browse on which they feed. Indeed, the Grant's gazelle, which is mainly a browser and water-independent antelope (Estes, 1967), can meet much of its water requirement through feeding on high water content browse. Similarly, giraffe derives much of its water requirement from its exclusive feeding on browse, and impala which is a mixed feeder obtains part of its water requirement by browsing on succulent forage (McNaughton & Georgiadis, 1986; Spies, 2015). Various species-specific factors such as degree of dependence on water, type of herbivore (e.g. grazer, browser, mixed feeders, omnivore), size and gut morphology, all contribute to different patterns in species abundance and distribution with respect to surface water availability (Redfern et al., 2003). For instance, as browsers extract much of their water needs from high water content forage, they are unllikely to be strongly associated with surface water sources (Redfern et al., 2003; Stommel, 2016).

As a water-dependent species, the abundance of elephants also increased with proximity to the surface water sources during the dry season (mainly encountered within the first kilometre from water source) but it increased away from the surface water during the wet season (Figure 4.10A and B). This finding aligns well with the fact that elephants have large water requirement including drinking (i.e. up to 200 litres a day), and wallowing (IUCN, 2021). The observed higher dry season elephant abundance around the water sources in the West Kilimanjaro is also in agreement with the findings by Kikoti (2009) in the West Kilimanjaro-Natron ecosystem. In search for sufficient quality water during the dry season, the elephants move all around in the ecosystem including KINAPA and the upstream section of the Simba River within village lands. It is also during this period of time (as water becomes severely scarce) when human-elephants conflicts and also human-human conflicts (see Chapter 2) escalate (Kikoti, 2009; Mariki et al., 2015). Therefore, dry season human-elephant conflicts seem to be centred mainly on the availability of surface water in the West Kilimanjaro ecosystem. During the dry season surveys, several incidents of elephants were observed and reported in the village areas particularly, Kitendeni, Tingatinga and Ngereiyani villages. During this period, one person was killed by an elephant in the vicinity of the man-made water holes in Ngereiyani village (personal communication with Mr. Taiko Mollel). In the past, Mariki et al. (2015) have also reported killing of people by elephants as well as retaliatory killing of elephants by villagers in the villages of West Kilimanjaro. Human-elephant conflicts further indicate the magnitude of the existing water crisis and how water availability is critical to both people, livestock and the widlife in the entire Kilimanjaro landscape. This implies that a successful management of human-wildlife conflicts will have to ensure sufficient and quality water is available across the ecosystem especially in the core wildlife areas.

In the West Kilimanjaro ecosystem, it was clear that unlike other herbivore species, elephants occupied a relatively large range depending mainly on the availability of essential resources particularly water and forage. Such observation is also in consistency with observations in other tropical ecosystems (Ndaimani et al., 2017). Previous studies have also indicated that elephants range widely in the entire West Kilimanjaro and adjacent Natron and Amboseli ecosystems (Kikoti, 2009; Muruthi and Frohardt, 2006). During the dry season of this study, the abundance of elephants was relatively high in the semi-arid areas, KINAPA and the associated villages. Some of the elephants moved to KINAPA, probably in search of sufficient and quality water, which was scarce in the semi-arid area. Against this background, it is therefore evident that water is a critical issue for elephants and other wildlife conservation throughout the West Kilimanjaro, KINAPA and ANAPA. However, previously published studies present some contradictory findings on the links between elephants and water. For instance, Kikoti (2009) reported an extensive use by the 7 out of 8 collared elephants of areas around water holes in the EWMA during both dry and wet seasons. Yet, the areas southwest of Amboseli National Park that have no dry season surface water, were also extensively used by the elephants. He also reported that elephants frequently moved into the Amboseli basin during the wet season when water was widespread in the ecosystem. However, this finding is contradictory to what was reported by Western and Lindsay (1984), that elephants concentrate in the Amboseli basin during the dry season primarily to access drinking water from the available swamps. In addition, Muruthi and Frohardt (2006) pointed out that elephants and associated ungulates frequently move from quality grazing land in Tanzania to Kenya's Amboseli basin where they drink in a network of swamps during the dry season.

Surface water availability affected not only the wildlife but also livestock as their abundance increased with increasing proximity to the surface water sources during the dry season and it increased with increasing distance from water during the wet season. The abundance of livestock was far higher (above five-fold) than the abundance of wild herbivores in the West Kilimanjaro (Figures 4.2 and 4.11A). There were few instances where both livestock and wild animals especially wildebeest, zebra, impala and Thomson’s gazelle accessed and drank in the same water sources and time but from different angles in water sources especially within EWMA. In some cases, some wild animals were also observed grazing close to the livestock. However, in most cases the wild animals seemed to avoid accessing the water sources when livestock were drinking. Largely, the wild animals drank water during the morning, evening and night hours when livestock was not present. In contrast, livestock accessed water sources mostly in late morning hours and in the afternoon. The observed situation is partly contrary to the findings reported by De Leeuw et al. (2001) that the distributions of wild herbivores and livestock with respect to surface water sources were negatively correlated, as livestock seemed to displace wildlife from the water sources in the arid rangelands of northern Kenya. In the EWMA and also the Ndarakwai wildlife ranch, there was a close overlapping between wild animals and livestock in the drinking and grazing patterns. Usually wild herbivores are likely to avoid encountering livestock to the extent permitted by the availability of forage and water resources. A study by Valls-Fox et al.(2018) in Sikumi forest in Zimbabwe demonstrated that buffaloes could not afford to avoid encountering the livestock in the course of accessing scarce water resources during the dry season. In this study, the observed close overlapping between the wildlife and livestock was possible in those areas that have some levels of protection and even in the presence of the Maasai livestock keepers because those people traditionally do not engage in game hunting. The Maasai community in the wildlife ecosystems are also said to have a high tolerance towards wildlife in their areas (Muruthi and Frohardt, 2006). However, the situation was not uniform across the entire area of West Kilimanjaro as the wild herbivores particularly elephants and zebras avoided human disturbances when accessing some of the drinking water sources located near village settlements such as Ngereiyani and Ngainyamo water holes (H1, H2 and H5), Kitendeni water troughs (e.g.T4), and some sections (S4 and S5) of the Simba River (see Chapter 2). The wild animals accessed these sources mainly during the night and early in the morning hours. Such behaviour is a common adaptation strategy for wild animals especially elephants (Harris et al., 2008). However, such behaviour may also contribute to human-wildlife conflicts through surprise encounters during the night hours.

As observed in the West Kilimanjaro, livestock and wild herbivores co-exist in the savannah areas. It is for instance known in this kind of environment that cattle and herbivores especially zebras and wildebeests often overlap in resources use (Voeten, 1999). This situation presents an important window of opportunity for biodiversity conservation especially in the era of rapidly growing human population and activities, which call for co-existence between man and nature at the landscape scale to attain sustainable conservation and development. This co-existence is imperative considering the limitations of the existing wildlife protected areas and the need for ecological connectivity and conservation at landscape scale to address impacts associated with anthropogenic activities and climate changes. Nevertheless, such co-existence depends on among other factors, the availability of essential resources notably water and forage, and existence of sustainable use and sharing of such resources. One of the challenges that must be addressed for a sustainable co-existence of livestock and wildlife in the West Kilimanjaro is the overstocking which exerts high consumption pressure on limited water resources, enhances the risks of disease transimission among animals and humans, and causes degradation of land, riparian habitat, and water resources through overgrazing, trampling and siltation that is adversely affecting surface waters especially the Simba and Ngarenyuki Rivers, and waterholes.

Figure 4. 14: Cattle drinking at Kitendeni water trough during the dry season.

Therefore, a sustainable co-existence of livestock and wildlife in the ecosystem should be encouraged through a number of mesures to ensure long-term sustainability, such as limiting the number of livestock and ensuring the partitioning and rotational use of land and water resources in the ecosystem. Further, upstream crop irrigators and domestic water users should stop excessive water abstraction to ensure downstream river flows that supply water to the downstream communities of people, livestock and the wildlife.

Overall, herbivore abundance in the dry season was higher and concentrated around the existing surface water sources compared to areas located away from the water sources. Abundance increased towards the surface water sources, despite majority of them declining substantially and some becoming more saline such as the Sinya mine water hole (i.e. ~7500 ppm) and Ngereiyani water holes (refer Chapter 2 and 3). These surface water sources especially Ngereiyani water holes also measured high concentration of iron (almost 100 ppm), aluminium (about 200 ppm) and nutrients (up 400 ppm) during the dry season (see Chapter 3). Dry season aggregation of both domestic and wild animals populations around these saline and mineral concentrated water sources from which they drank water, implies that water availability mattered more than water quality. However, such high aggregations of animals around scarce and poor quality water sources may increase the risks of disease transimission such as intestinal parasites and contact transmitted diseases, e.g. foot and mouth disease, among and between the wildlife and livestock (Ogutu et al., 2010; Strauch, 2013; Jori and Etter, 2016). However, in the dry season, elephants (and possibly some other herbivores) likely adapted for poor quality and scarce surface water by searching and drinking from other good quality freshwater sources. In the wet season, the overall abundance of both wild herbivores and livestock was more negatively associated with saline water than freshwater. This is because herbivores tend to look for good quality water whenever it is reasonably affordable. For instance, in Ruaha National Park when water is scarce and of poor quality especially with respect to bacterial load, elephants and zebras dig water holes to access good quality water (Stommel, 2016). In Tarangire National Park, elephants are known to avoid water with a salinity concentration of more than 2000 ppm (Gereta et al., 2004) in the presence of alternative less saline water. Drawing on these experiences it is therefore likely that some of the wild herbivores especially elephants, attempt to quit saline water in the West Kilimanjaro region and move upstream to drink fresh water from the Simba River and others sources in KINAPA during the dry season. Elephants appear to be widely ranging in the entire ecosystem (Kikoti, 2009), and their abundance increased substantially around the surface water extraction sites in KINAPA during the dry season. However, remain of some of the wild herbivore species around the high dry season saline waters (H1, H2 and H3 in Figure 4.1) in the West Kilimanjaro might indicate that costs of searching for quality water outweigh benefits gained by those species. This is due to among others, high costs involved in travelling long distance upstream for at least 35 km across areas of human disturbances in order to drink from fresh water sources in the Simba River around Ndarakwai ranch and the neighbouring villages, or move further upstream to Kilimanjaro National Park. In addition to unsuitable habitats, none of these options seems to be easily affordable to most of the herbivores that remain confined to high saline and metal and nutrients concentrated waters. While the actual effects of long-term consumption of poor quality water in the landscape are not known, such consumpation may lead to several physiological stress that adversely affect animals' reproductive health and population growth.

4.4.3 Management implications

This study has provided the evidence that surface water is the main driving factor controlling the abundance and distribution of wild herbivores in the Kilimanjaro landscape. While water quantity seems to be a necessity, water quality is also important. The National

Parks upstream and the semi-arid region downstream are closely linked and must be managed as one. Efforts must be taken to ensure sufficient, quality water is allowed to flow downstream within and outside the parks to sustain biodiversity particularly large herbivores. Studies by Muruthi and Frohardt (2006) and Kikoti (2009) indicate that wild herbivores and elephants spend more of the dry season period in the Kilimanjaro areas including Amboseli National Park in Kenya, mainly because of access to water draining from Mt. Meru (ANAPA) and Mt. Kilimanjaro (KINAPA), and that which is available in the man-made water holes. However, due to excessive water extraction the Ngarenanyuki and the Simba Rivers water from the parks no longer reaches the semi-arid areas in the dry season (Chapter 2). Further, the water extraction is also unsustainable even within the parks. The General Management Plan of ANAPA clearly acknowledges the increased demand for water extraction as the very high ecosystem threat to the park (ANAPA, 2014). Likewise, the General Management Plan of KINAPA categorises over-extraction of water in the park as a high ecosystem threat which is currently the highest rank given to most serious threats in the park (KINAPA, 2016). To ensure a sustainable conservation of biodiversity in the Kilimanjaro landscape, water and land resources should be effectively managed at the ecosystem/watershed level based on an integrated water resource management plan. In line to this, would be to foster water use efficiency through among other ways, establishing improved irrigation systems. Natural downstream water flow should be promoted, and adequate artificial water points established and widely spaced within the ecosystem to ensure ecosystem resilience while mitigating for adverse impacts on vegetation, soils and rare herbivore species. Availing sufficient and good quality water, restoring and maintaining ecological connectivity (wildlife corridors, dispersal areas, and riparian zones) within the land matrix will not only improve biodiversity and water resources, but also alleviate the existing human-wildlife conflicts in the landscape. Ultimately, all this calls for an integration and coordination of development and conservation agenda at both catchment and landscape scale.

4.5 Conclusion

This study examined how surface water availability affects herbivore abundance and space use in the Kilimanjaro landscape. Overall, the wild herbivores were more abundant near water sources during the dry season than the wet season clearly indicating that availability of surface water controlled species abundance and distribution in the ecosystem. Largely, both wild animals and livestock were mainly influenced by water availability compared to water quality especially during the dry season when water was scarce. Due to water scarcity, higher animal abundance was observed around saline and mineral concetrated than fresh water sources in the semi-arid region during the dry season. Animals would however prefer fresh to saline water, where freshwater is available and reasonably accessible. During the wet season animals seemed to prefer less saline (fresh) water and their abundance around such sources was higher than saline water sources. Elephant's movement and distribution might also be reflecting the need for good water quality especially during the dry season as their numbers increased around the relatively good quality (fresh) water sources in the parks. They also damaged water infrastructures, which partly might be linked to the searching for more sufficient and quality water when available water becomes scarce, saline and polluted. Water played a central role as manifested in its influence on the herbivore abundance and distribution. However, the ecosystem is faced by a growing water crisis mainly from excessive water extraction within and outside the parks (Chapter 2). Such crisis is affecting both the people and wildlife through emergence of human-human conflicts, human-wildlife conflicts, damage of water facilities, and changes in herbivores abundance and distribution where animals aggregate around the scarce water sources particularly during the dry season. This may cause physiological stress to herbivores and enhance risks for disease transimission among both wildlife and livestock. The study has shown the critical role surface water plays in shaping herbivores abundance and distribution and has further revealed that the National Parks upstream and the semi-arid region downstream are closely linked and therefore they must be managed as one system. This implies that policies and management practices are likely to be successful in promoting sustainable management of water and biodiversity resources if they focus on the entire landscape while ensuring encompassment of the existing watersheds. However, the existing water crisis is developing quickly and therefore it is a matter of urgency to plan and implement integrated and ecologically sustainable water and land resources management to promote sustainable biodiversity conservation and development in the entire landscape.

4.6 References

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Chapter 5: Impacts of surface water changes on riparian and floodplain vegetation

Abstract

The integrity of surface water bodies depends on the maintenance of the natural hydrological regime, which in turn supports riparian and associated ecosystems. An excessive water abstraction occurs, mainly for irrigation farming, domestic use and livestock watering in the Kilimanjaro landscape, and this threatens the ecosystem as it causes significant water shortages to the downstream communities. This chapter evaluated the consequences of changes in surface water availability on the riparian and fringing floodplain vegetation in the Kilimanjaro landscape. The amount of surface water available, abstracted and hence left for the environment was evaluated in the key wildlife areas. Systematic random sampling was used to quantify riparian vegetation diversity in the abstracted riparian wetlands in Arusha National Park (ANAPA), and herbaceous plant ground cover in the lowland semi-arid areas. The LandTrendr (Landsat based detection of trends in disturbance and recovery) algorithm was employed to assess riparian and adjacent vegetation cover around the surface water bodies in the lowland semi-arid wildlife areas downstream of Arusha and Kilimanjaro National Parks. There was an increase in riparian wetland vegetation diversity in ANAPA since 2013, likely due to an increase of rainfall and water availability during that period within the park. However, in the lowland semi-arid areas, there was a loss in riparian and floodplain vegetation, and was likely due to both water shortage from excessive water withdrawal, mainly by irrigators upstream, and due to excessive trampling and overgrazing by domestic and wild animals concentrated around the remaining scarce water sources during the dry season. The study calls for a careful monitoring and control of water abstraction in order to promote a balanced water use to meet both human and biodiversity needs at the watershed and ecosystem scales. In addition, adequate and properly distributed artificial water points for livestock and wild animal may help alleviate overgrazing in the few natural surface water sources downstream of the parks. Such measures would help conserve the associated vegetation and biodiversity in the lowland semi-arid areas of the Kilimanjaro landscape.

5.1 Introduction

Surface water availability is essential for the maintenance of healthy riparian ecosystems that support a diversity of flora and fauna, and key geomorphological, hydrological and ecological processes (Ward, 1998; National Research Council, 2002; Van Dijk et al., 2013). Surface water supports the development of riparian vegetation including water-loving species such as *Acacia xanthophlea, Pycreus mundtii, Cyperus spp*, that in turn promote water availability, improve water quality, and provide forage and diverse habitat for wildlife (Vesey-FitzGerald, 1974; Hamilton, 2002; Ramberg et al., 2006; Kiwango and Wolanski, 2008; Diop, 2010; Kihwele et al., 2012; Okruszko et al., 2014; Elisa et al., 2021). However, globally, riparian ecosystems are threatened by high water demand from the rapidly growing human population and development activities (Millennium Ecosystem Assessment, 2005; Rebelo et al., 2010). Water abstraction and the regulation of flow is one of the major factors degrading riparian vegetation (Meeson et al., 2002; Richardson et al., 2007; Meragiaw et al., 2018). Excessive abstraction of water and flow regulation substantially reduce water availability and alter flooding regimes, which are required for the maintenance of a productive riparian ecosystem (USDA, 1996). Unfortunately, in many cases, excessive water abstraction is often associated with other human-induced ecological disturbances, especially farming and cattle grazing that exacerbate impacts on riparian and adjacent ecosystems (Richardson et al., 2007; MEMR, 2012).

Such unsustainable water management seriously threatens the riparian ecosystems in the wildlife areas of sub-Saharan Africa. In Cameroon, Drijver and Marchand (1985) documented the effects on riparian ecosystems by a dam and its reservoir in the Benue River, which flooded the riverine forest between Benue and Bouba-Ndjida National Parks, and reduced inundation in the downstream riparian ecosystem areas. The damming of the Kafue River in Zambia caused a displacement of grassland and an explosion in the invasive *Mimosa pigra* plant in parts of the Kafue flats that include Kafue National Park (Sheppe, 1985; Mumba and Thompson, 2005). The water abstraction and damming of the Olifant River, together with industrial activities, have polluted and substantially reduced the flow in the river and adversely affected the aquatic and riparian ecosystems in the Kruger National Park in South Africa (Gyedu-Ababio et al., 2012). The excessive water abstraction from the

Lumi River in Kenya has caused a significant reduction in flow downstream, and consequently a decline of the Lake Jipe water level, which in turn led to the degradation of the riparian vegetation of the lake and of Tsavo West National Park (Ndetei, 2006). The excessive water abstraction in the upstream areas of the Katuma and Great Ruaha Rivers in Tanzania have affected the ecology including degradation of riparian vegetation and disruption of herbivore distribution in the semi-arid Katavi and Ruaha National Parks (Elisa et al., 2010; Stommel, 2016).

Water extraction in the Kilimanjaro landscape

The study focuses on the key surface water sources for wildlife in the Arusha National Park (ANAPA) located upstream, and those in the downstream semi-arid areas (Figure 5.1). Surface water abstraction in the Kilimanjaro landscape has always occurred and some of the tradition irrigation farming schemes date from the 19^{th} century, though water extraction was very small (Grove, 1993; Tagseth, 2008). In recent decades however, human population and associated activities have substantially increased, and as a result, water extraction has greatly increased for the rapidly expanding irrigation farming, domestic water use and livestock watering (Munishi et al., 2009). This has resulted in a substantial water shortage or even the deprivation of water in most of the downstream areas including riparian and floodplain ecosystems during the dry season (Chapter 2). The encroachment by invasive species and the loss of riparian wetland vegetation is not limited to the lowland areas; it also is occurring in the protected upstream areas (ANAPA) where excessive water abstraction takes place (Elisa et al., 2016). Loss of riparian vegetation also contributes to a water shortage in the downstream semi-arid areas of the Kilimanjaro landscape. In those areas, the water crisis is manifested through irrigated crop cultivation, and over-grazing and trampling by domestic and wild herbivores in areas adjacent to the few remaining dry season water sources. This degrades the riparian and adjacent vegetation to the point of causing dry-season desertification in some areas (Allsopp et al., 2007; Munishi et al., 2009; MEMR, 2012). These impacts were reported qualitatively, but have not been quantified until now. This chapter evaluates the impacts of excessive water abstraction on the riparian and adjacent (floodplain) vegetation, and also increased ecological disturbance especially farming, and grazing and trampling pressure by the large wild and domestic herbivores in the Kilimanjaro landscape. The riparian vegetation is referred to as that vegetation community growing along surface watercourses and water bodies, which provide an interface between terrestrial and aquatic ecosystems (USDA, 1996; Richardson et al., 2007; Vesipa et al., 2016). The floodplain vegetation refers to the vegetation occasionally flooded. The study aimed to answer the following questions: (1) Is there difference in wetland vegetation diversity between riparian wetlands with water abstraction and those without abstraction in ANAPA? (2) Is there a difference in vegetation diversity in ANAPA between current riparian vegetation and that of 2013 as assessed by Elisa et al. (2016) due to increased rainfall and water availability in the most recent time period? (3) Is there evidence of downstream vegetation loss in riparian and floodplain areas due to upstream water abstraction, farming, animal grazing and trampling pressures around the few remaining surface water sources? (4) What are the spatial-temporal patterns, and trends of such vegetation cover losses?

To answer these questions, I combined field-based data and satellite imagery in quantifying the change in surface water and associated vegetation change. This involved quantifying of the amount of surface water available, extracted and left for the environment (see Chapter 2). I then quantified the associated change in riparian and floodplain vegetation in terms of species diversity in ANAPA, and herbaceous plant ground cover at Ndarakwai wildlife ranch located in the lowland semi-arid areas. Further, I mapped the long-term loss of vegetation cover around surface water bodies in the semi-arid areas of the Kilimanjaro landscape. This assessment of the impact of surface water change on the vegetation was rendered difficult by the lack of long-term hydrological data but was partly remediated by the use of recent satellite data.

5.2 Methodology

5.2.1 Study Area

The study was conducted in the Kilimanjaro landscape, which is a wildlife-rich area situated in northern Tanzania (Figure 5.1). The area consists of several protected areas including the Arusha National Park (ANAPA) and the Kilimanjaro National Park (KINAPA). Others include the NARCO livestock ranch and the Ndarakwai wildlife ranch, the Enduimet wildlife management area (EWMA), and two wildlife migration corridors. These corridors are Kisimiri (currently highly narrowed by expanding human settlements and farming), and Kitendeni; they respectively link (ANAPA) to the West Kilimanjaro area, KINAPA to the Amboseli National Park in Kenya (Kikoti, 2009). There are several small lakes in the study area, including Lake Amboseli in Kenya, and Lakes Chala and Jipe which are both trans-boundary lakes between Kenya and Tanzania. These lakes are situated in wildlife-rich areas, and they receive most of their water as surface or groundwater from ANAPA and KINAPA. There are also several man-made water holes that supply water to the livestock and wildlife in the semi-arid areas. The semi-arid area largely depends on river water originating from KINAPA, and ANAPA, to sustain the livestock, wildlife and riparian wetland vegetation. In addition, even in the protected areas of KINAPA and ANAPA in the upper catchment, perennial surface water sources from springs, streams and rivers (e.g. the Simba and Ngarenanyuki Rivers), are also excessively abstracted to meet human needs especially irrigation farming and domestic use outside the parks. As a result, the downstream riparian and the floodplain areas, the wetlands and their associated fauna are deprived of water, especially during the dry season. In the study area, a number of livestock (cattle and sheep), and wild animal species drink from the available surface water sources and graze in the surorunding riparian and floodplain areas. The wild animals include elephant (*Loxodonta africana)*, buffalo *(Syncerus caffer*), wildebeest (*Connochaetes taurinus*), plains zebra (*Equus quagga*), Thompson's gazelle (*Eudorcas thomsonii*), Grant's gazelle (*Nanger granti*), Masai giraffe (*Giraffa camelopardalis ssp. Tippelskirchi*), defassa water buck (*Kobus e. defassa),* warthog (*Phacochoerus africanus)*, lesser kudu (*Tragelaphus imberbis*), impala (*Aepyceros melampus*), and striped hyena (*Hyaena hyaena*) (Kikoti, 2009).

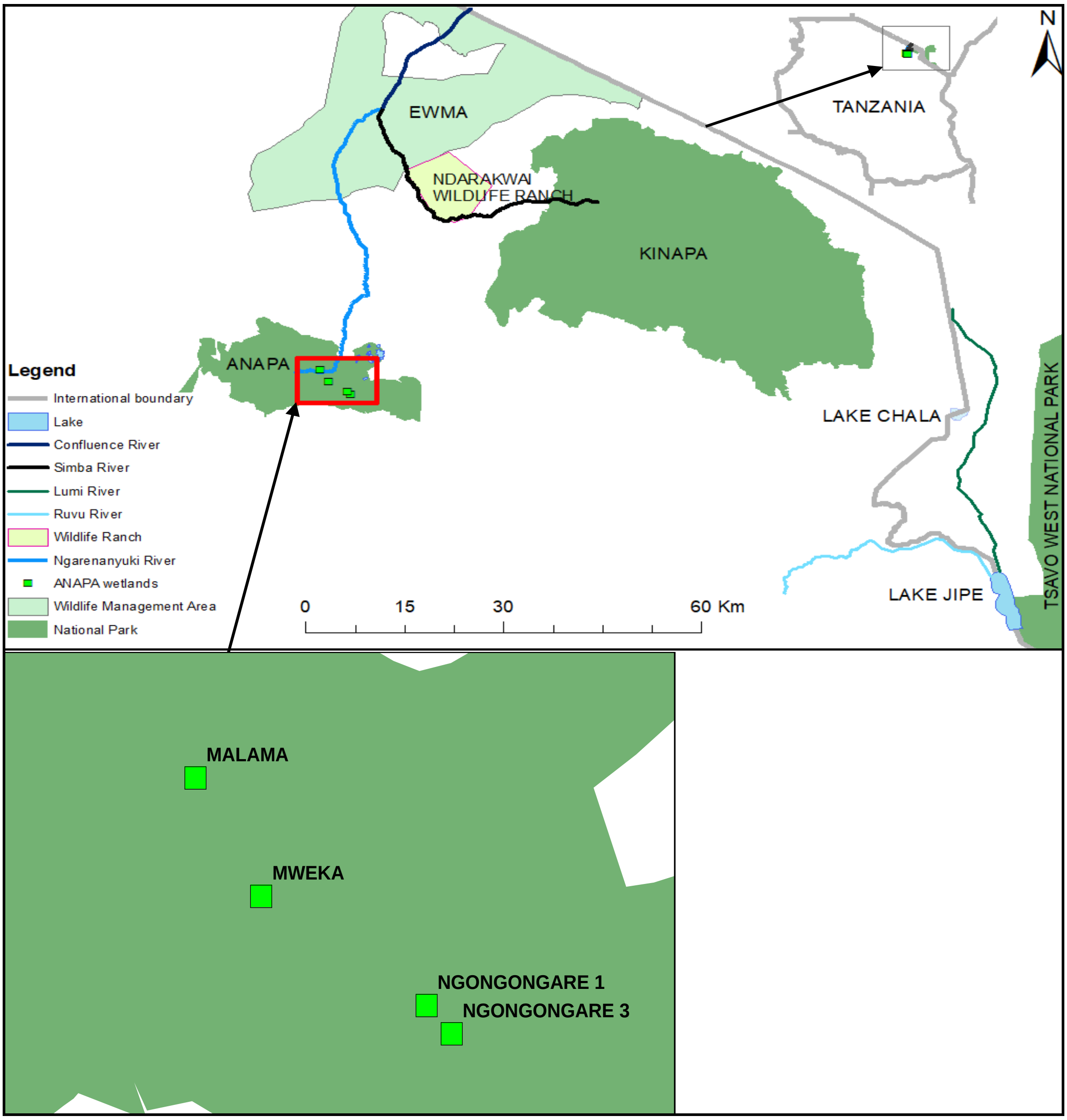

Figure 5. 1: (Top) A map of the Kilimanjaro landscape showing areas where the vegetation assessment was conducted. (Bottom) A zoom-in of the four studied wetlands in ANAPA.

5.2.2 Data collection and analysis

Data on surface water availability and quality were collected over a period of almost 2 years and are presented in detail in the Chapter 2 and 3. During this study, none of the four riparian wetlands in ANAPA were completely deprived of water due to over-abstraction, unlike during the dry season in 2013 as reported by Elisa et al. (2016). In this chapter, additional data on the historical changes in dry season surface area of Lake Jipe is presented to explore how such change might have affected the riparian and floodplain vegetation surrounding the lake. Landsat imagery of the dry season lake surface area since 2000 were obtained from the USGS earth explorer platform, and then processed in ArcGIS to compute area. Collection and analysis of vegetation data is outlined below.

5.2.2.1 Ground data

Data collection

Riparian vegetation diversity and richness in ANAPA

Using a systematic random sampling approach, riparian wetland vegetation diversity in ANAPA was assessed in January and February 2019 (thus in the dry season) in three downstream wetlands (Ngongongare 1, Ngongongare 2 and Mweka) receiving less water, due to abstraction in upstream areas during the dry season. This is the time when water scarcity is a probable limiting ecological factor (Elisa et al.2016; Chapter 2). As water quality and quantity have a potential to affect wetland plant species diversity, dry and wet season water assessment (measuring quantity and quality) was carried as shown in the previous chapters 2 and 3. Data collection for the vegetation assessment was carried out in 2019 during the dry season period comparable to the similar dry season survey conducted in 2013 and presented in Elisa et al. (2016). In addition, a control study was conducted to assess the diversity of wetland vegetation in Malama wetland where there was no water abstraction. The aim is to determine if there was a difference in riparian vegetation diversity between the wetlands with water abstraction and those without abstraction in the wildlife rich areas (i.e. areas with high wild animal density and hence where the animals are frequently sighted according to park rangers). The four study sites (riparian wetlands) were selected based on the advice of the park staff and the previous study by Elisa et al. (2016). Systematic random sampling was used for locating the vegetation quadrats within each wetland, as it is a reliable method for sampling riparian wetland vegetation, and suitable for standardising sampling across variable wetlands as sampling units were evenly distributed across a study site (Barker, 2001; Meragiaw et al., 2018). All the riparian wetlands were located in areas that have similar environmental (i.e. edaphic, climatic and hydrological) characteristics. The maximum distance from one site to the other was 6 km. Hence the sites were in a similar geographic and climate setting, and thus they were comparable. No riparian vegetation assessment was conducted in KINAPA due to the absence of riparian wetlands subject to water abstraction and located in wildlife rich areas.

Ground data collection and analysis in ANAPA focused only on wetland plants due to their high sensitivity (and hence good indicators) to water change (Woldu, 2000). Both obligate and facultative wetland vegetation species were considered in this study. Almost 100% of obligate wetland plants only occur in wetlands under natural conditions, while between 67 and 99% of facultative wetland plants are found in wetlands (Lichvar et al., 2012).

To aid comparison between the findings in 2019 and those of 2013, the riparian vegetation diversity assessment followed the methodology used by Elisa et al.(2016). The survey of the riparian vegetation involved running one 60 m long transect per wetland starting from upstream and following the direction of water. This length of transect was adopted to facilitate comparisons across the wetlands because it represents the maximum length of the smallest wetland. In addition, in each surveyed wetland, a total of 5 quadrats, sized 1 m^2 and spaced 15 m apart, were established along the 60 m long transect following the methodology of Barker (2001). With the exception of the first quadrat, which was randomly placed, all other quadrats in each surveyed wetland were systematically placed, alternating from right to left at 5 m distance from the central transect line. All plants in each quadrat were identified and counted. The identification of the plants was carried out with the assistance of a botanist from the National Herbarium in Arusha. Those plants that could not be fully identified on site were pressed and transferred to the herbarium for identification.

Herbaceous plant ground cover in Ndarakwai wildlife ranch

Assessment of herbaceous plant ground cover (as % area of the ground covered by herbaceous plants) was carried out in Ndarakwai wildlife ranch located in the lowland semi-arid areas and which receives water only from the Simba River (see Chapter 2). The assessment was carried out once during the dry season to determine vegetation cover change due to grazing pressure that builds around the river because of water shortage downstream caused by over-abstraction of river water in the upstream areas. A total of three transects, each 2 km long, and placed perpendicular to the river were established in the wildlife ranch. These transects followed the same animal survey transects described in Chapter 4 to assess the vegetation impact of the dry season increase in herbivore abundance with proximity to scarcely available surface water sources. Two of these transects which were placed 1.5 km apart, started from the river bank and extended northwards into the Ndarakwaki wildlife ranch. The third transect was established as control transect and thus located 5km away from the river. This study used an approach similar to that used by Elisa et al. (2016) in assessing riparian vegetation diversity in ANAPA, and also by Kavana et al. (2019) in assessing herbaceous plant ground cover and diversity in the Serengeti ecosystem, Tanzania. Each transect had 11 quadrats, each sized 1 m^2 and spaced 200 m apart. With the exception of the first quadrat, which was randomly placed, all other quadrats in each transect were systematically placed 200 m apart in a straight line transect. Herbaceous plant ground cover in each quadrat was then visually estimated, and quadrat distance from the river recorded.

Analysis

To quantify and compare riparian vegetation diversity between the wetlands (abstracted and un-abstracted), and between the two different survey years, the diversity index (H) and the species richness index (S) were calculated from the riparian wetland vegetation data using the Shannon-Wiener equations (Wold, 2000). The Shannon diversity index is a useful tool that reflects the rarity and the commonness of species, and the community composition, including the degree of species dominance and diversity in a given community (Wold, 2000). The species richness often provides a reflection of the ecosystem and species sensitivity to environmental changes (Meragiaw et al., 2018). Hence, it was considered a suitable indicator of the impact of surface water change on the riparian vegetation.

The equations and respective computations followed a similar approach used in assessing the composition and diversity of riparian vegetation in the wetlands of Illubabor by Woldu (2000), and along Wolga River of Wonch by Meragiaw et al. (2018) in Ethiopia. The Shannon equation was used in computing vegetation diversity index (H):

$$\text{Shannon Index (H)} = -\sum_{i=1}^{s} p_i \ln p_i \quad (1)$$

where;

s= the number of vegetation species in the community.

Pi= proportion of the individual of the i^{th} species or abundance of the i^{th} species expressed as the proportional of the total cover.

Ln= is the natural logarithm.

Species richness was further used to assess water-related change in vegetation diversity within the wetlands. Species richness index (S) was computed using the following equation:

$$S = \sum_{i=1}^{S} \mathrm{Si} = \Sigma\, \mathrm{Si} \quad (2)$$

Where; Si= the number of individuals in the i^{th} species.

The vegetation indices were calculated per quadrat and then all quadrat index values in each wetland were averaged to compute the mean index value of each wetland.

Further, the T-test and one-way Analysis of Variance (ANOVA) were used to determine statistical differences in riparian vegetation diversity respectively between pairs of abstracted and un-abstracted wetlands and across all wetlands (Bueno et al., 2020). Wetland riparian vegetation diversity was also compared between current (2019) and previous (2013; Elisa et al., 2016) surveys to determine if there was a difference in riparian vegetation diversity between the two. Further, a regression analysis was run to find out the relationship between the amount of surface water available for the wetland environments and wetland vegetation diversity indices in both 2013 and 2019. This was deemed important in establishing the effect of water availability (and hence rainfall) on wetland vegetation diversity, and thus in confirming whether the change in surface water between these two periods could lead to a change in riparian vegetation diversity. Only wetland plant species were subject to analysis.

Data on herbaceous plant ground cover in Ndarakwai were coded in Ms Excel and then subjected to a regression analysis to establish if there was a relationship between the percentage of herbaceous ground cover and the distance to the surface water source, i.e. the Simba River.

5.2.2.2 Landsat Satellite Data

Landsat data was used to explore the spatial-temporal patterns and trends of vegetation disturbance around surface water bodies of the semi-arid areas in the Kilimanjaro landscape. The term ‘disturbance’ as used in this study may mean a reduction or complete removal (loss) of vegetation cover. The semi-arid areas were suitable for Landsat imagery analysis as they had no forest canopy cover, and were largely free from cloud cover. Moderate spatial resolution (~ 30 m) Landsat satellite data were used to examine spatial and temporal patterns of disturbance in vegetation cover within 5 km of surface water bodies (i.e. in the flood plains) in the downstream semi-arid areas. Such resolution is sufficient in detecting changes in a savannah environment. The Landsat data enables the measurement of vegetation changes because of its consistency in capturing temporal-spatial variations at a sufficient temporal and spatial scale. A Landsat based detection of trends in disturbance and recovery (LandTrendr) tool utilising a vegetation index (NBR) was applied in Google Earth Engine (GEE) platform to detect vegetation disturbance. The Normalised Burn Ratio (NBR) was used to examine water-related disturbance in vegetation cover in the semi-arid areas of the Kilimanjaro landscape due to its high sensitivity to inter-annual vegetation changes in the study area. NBR is a vegetation index related to moisture levels (Housman et al., 2021), and it is a most sensitive spectral index that can adequately detect land use and cover changes in dynamic dry tropical landscapes (Rathnayake et al., 2020; De Marzo et al., 2021; Kennedy et al., 2012, 2015; Hermosilla et al., 2018; Nguyen et al., 2018; Komba et al., 2021). That index has also been used successfully in assessing savannah environmental changes due to farming and grazing (Aalto, 2020; Komba et al., 2021) and therefore it was suitable for assessing riparian and adjacent vegetation cover disturbance in the Kilimanjaro landscape.

LandTrendr provides a reliable mechanism to detect vegetation changes (disturbance and recovery) as well as tracking the change continuously over a period of time (Kennedy et al., 2010; Woodcock et al., 2020). In this study, the time-series analysis was carried out in the dry season (late June to late September) covering a period of 21 years from 2000 to 2020. The period from June to September was selected as it falls within the dry season period, during which vegetation phenology is relatively stable, and there is low cloud cover, and hence relatively high and accurate satellite imaging. This is also the dry season, which is the period when surface water in the semi-arid areas is often scarce and thus most of the water-related disturbances to the vegetation occur (De Marzo et al., 2021). Landsat data over this period was deemed sufficient to capture the long-term patterns and the trends in water-based vegetation change in the Kilimanjaro landscape. This assessment period is also similar to a study by Komba et al. (2021) that successfully utilised a LandTrendr tool to asses vegetation disturbances around protected areas in Central Tanzania over a period ranging from 2000 to 2019. An assessment over a period longer than 21 years was not possible because of data quality problems and data gaps before 2000 (Aalto , 2020).

Atmospherically corrected surface reflectance Landsat imagery were accessed and sorted in GEE for detecting changes and trends in vegetation disturbances, to map water-associated vegetation disturbance events in terms of magnitude of disturbance, duration of disturbance, and the year of disturbance. An approach similar to this has been demonstrated to be effective in previous studies (Kennedy et al., 2010; Kennedy et al., 2018; Woodcock et al., 2020; De Marzo et al., 2021; Komba et al., 2021). Specifically, the codes provided in Kennedy et al.(2018) were used to automatically run the LandTrendr algorithm in GEE. Using the LandTrendr algorithm, the short and long-term vegetation disturbances were identified through a temporal segmentation of each pixel, and the vertex that identified the temporal breakpoint per year, and this was followed by the removal of yearly noises from the time series. Using the regression and point-to-point line as specified in the control parameters, the algorithm fitted a straight-line trajectory connecting the vertices (Figure 5.2). The study also adopted the previously given parameter values (Kennedy et al., 2018) which are widely acceptable standards when running LandTrendr algorithm to examine trends in vegetation dynamics (Table 5.1).

Table 5. 1: Parameters used in LandTrendr analysis**.** Source: (Kennedy et al., 2018).

Parameter Name	Value	Description
maxSegments	6	Maximum number of segments to be fitted on the time series
spikeThreshold	0.9	Threshold for dampening the spikes (1.0 means no dampening).
vertexCountOvershoot	3	The initial model can overshoot the maxSegments + 1 vertices by this amount. Later, it will be pruned down to maxSegments + 1
preventOneYearRecovery	true	Prevent segments that represent one year recoveries.
recoveryThreshold	0.25	If a segment has a recovery rate faster than 1/recoveryThreshold (in years), then the segment is disallowed.
pvalThreshold	0.05	If the p-value of the fitted model exceeds this threshold, then the current model is discarded and another one is fitted using the Levenberg- Marquardt optimizer.
bestModelProportion	0.75	Takes the model with most vertices that has a p-value that is at most this proportion away from the model with lowest p-value.
minObservationsNeeded	6	The minimum number of observations required for output fitting.

Depending on the nature of the vegetation change (loss or recovery) indicated by the pixel, the trajectory line may be a single segment, which represents a stable or gradually changing event, or multiple segments for a case where there are several significant vegetation changes over time (Komba et al., 2021). In this study, the processing parameters were set to allow the LandTrendr algorithm to identify the largest vegetation disturbance (loss/reduction) from each pixel and then fit the model to the Normalised Burn Ratio (NBR) spectral index (Figure 5.2). From this fitted segment line, the magnitude of vegetation disturbance, the duration of vegetation disturbance, and the year of detection of such disturbance were computed. The NBR values in savannah often range from almost 0.2 to 0.6 (Komba et al.,2021) and thus the control parameters for the magnitude of vegetation disturbance were set at minimum value of 0.25 and a maximum value of 1.0 which implies a total loss of vegetation. This ensured identification of only the targeted large change events and leaving out the small/subtle magnitude change events that might be confounded with background noise.

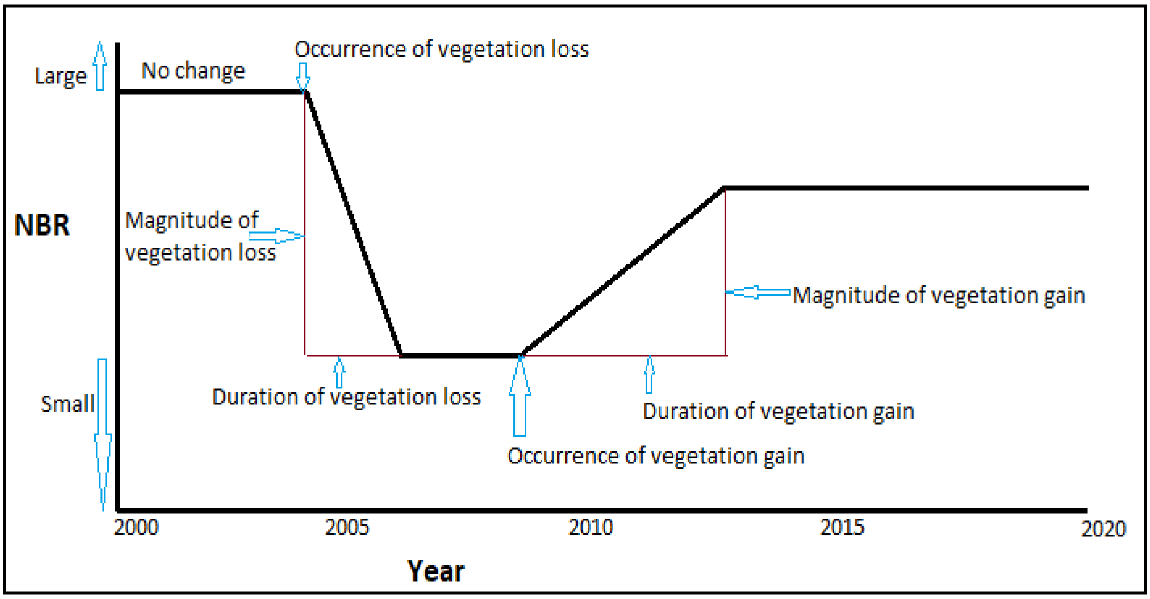

Figure 5. 2: A conceptual model illustrating how LandTrendr performs temporal segmentation for detection of trends in vegetation change, and the information that can be obtained from the output. Source: modified from Komba et al. (2021).

The mapping of vegetation disturbance patterns focused around target surface waters in the semi-arid Kilimanjaro landscape. The focal water sources were the Simba and Ngarenanyuki Rivers that traverse both protected areas and agricultural land, and a section of the Lake Jipe forming part of Tsavo West National Park. Outputs from this analysis included the magnitude and duration of vegetation disturbance, and the year of detection of vegetation disturbance (occurrence of vegetation disturbance). The field data (Chapter 2 and 4), visual observations, in combination with high-resolution Google Earth imagery helped interpret the Landsat satellite data and also suggest reasons for the observed changes. A similar approach was also applied by Rathnayake et al.(2020) and Komba et al. (2021) in Sri Lanka and Tanzania respectively.

5.3. Results

5.3.1 Overview of surface water availability and quality

Based on the 2018 to 2020 water budgets detailed in Chapter 2, there was excessive abstraction of surface water mainly for domestic and irrigation farming in the upstream areas leading to scarcity or deprivation of water in the downstream areas of the Kilimanjaro landscape. The water abstraction in ANAPA took almost 90% of the available water. While a substantial amount of water was being extracted from the assessed wetlands in ANAPA, the remaining dry season water flow in the wetlands was significantly ($p<0.01$, $F=5$, $n=6$) higher in 2019 than 2013 (see Chapter 2). In terms of water quality, all the sites in ANAPA had salinity level below 300 ppm, but the salinity reached 1000 ppm, and sometimes higher, in the lowland semi-arid areas. Surface waters in the study sites of the Kilimanjaro landscape were generally alkaline with pH mostly ranging from 7.5 to 8.5, values that essentially reflect a good quality water (see Chapter 3). Less than 30% and 20% of the available water in the upstream Ngarenanyuki and Simba Rivers, reached the downstream areas (located ~ 20 to 30 km from upstream) at Ngabobo village and the Ndarakwai wildlife ranch. Largely due to excessive abstraction of water in the inflow Lumi River, the transboundary Lake Jipe seems to be on the decline in the dry season period. Results derived from the Landsat data (Figure 5.3) show that the Lake Jipe surface area in the dry season decreased with time ($p<0.01$, $R^2=0.5$, $t=3.69$, $n=12$). Thus, its dry season water level was also decreasing. In turn, this leads to a reduced inundation of the riparian zone of the lake.

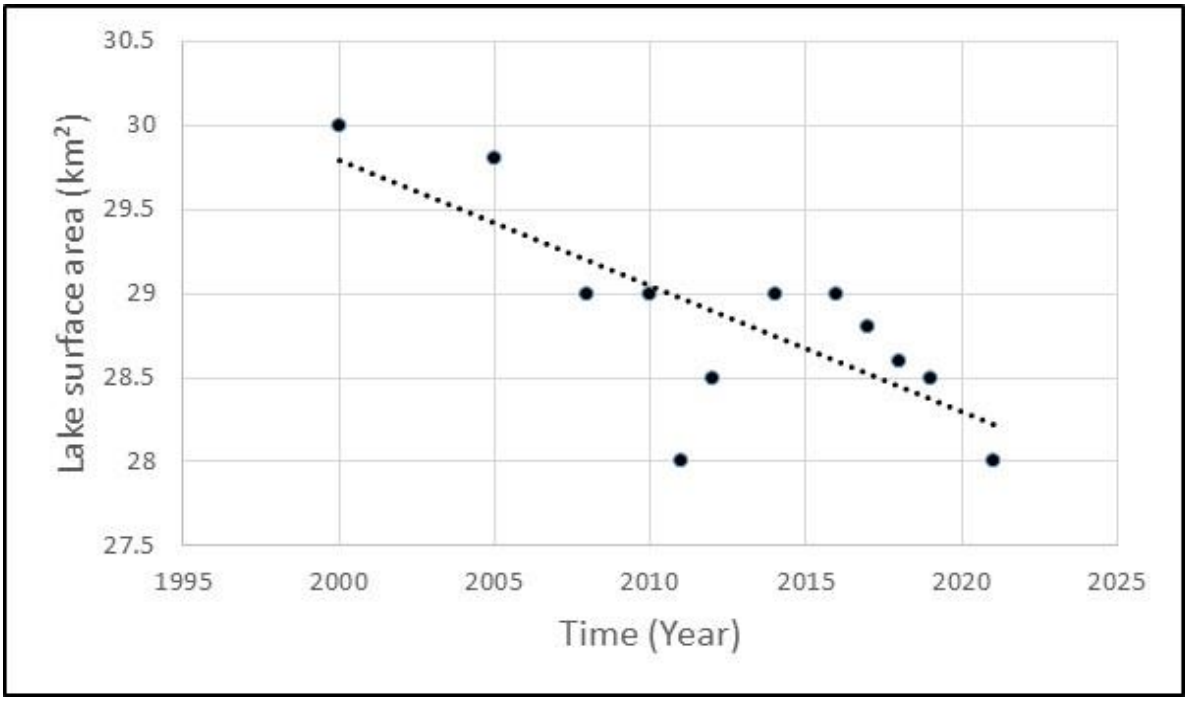

Figure 5. 3: Dry season (August and September) decline in surface water area in the Lake Jipe from 2000 to 2021.

5.3.2 Impact of water abstraction on riparian wetland vegetation in ANAPA

The major wetland vegetation species and their distribution across the four riparian wetlands in ANAPA are shown in Table 5.2. A total of 18 and 22 riparian wetland vegetation species were recorded respectively in 2013 and 2019 in the ANAPA wetlands. Nearly 50% of the wetland species sampled in 2019 were also encountered during the 2013 sampling. The most common species which also appeared in both periods were: *Centella asiatica, Commelina benghalensis, Vigna parkeri , and Cyperus species.*

Table 5. 2: Riparian wetland vegetation species in the ANAPA wetlands in 2019 and 2013.

Riparian wetland vegetation species in the ANAPA wetlands in 2019					
S/n	**Species scientific name**	**Ngongongare 1**	**Ngongongare 3**	**Mweka**	**Malama**
1	*Azolla nilotica*	✓	✓		
2	*Centella asiatica*	✓	✓	✓	✓
3	*Commelina benghalensis*	✓	✓	✓	✓
4	*Cynodon dactylon*	✓	✓		✓
5	*Cyperus exaltatus*	✓	✓		✓
6	*Cyperus maranguensis*	✓			
7	*Cyperus articulatus*		✓	✓	✓
8	*Cyperus laevigatus*				✓
9	*Cyperus rigidifolius*				✓
10	*Ludwigia abyssinica*	✓	✓	✓	✓
11	*Pycreus mundtii* var. uniceps	✓	✓	✓	✓
12	*Sphaeranthus bullatus*	✓	✓	✓	
13	*Polygonum senegalense*	✓			
14	*Leersia hexandra*	✓			
15	*Vigna parkeri*	✓	✓	✓	✓
16	*Sibthorpia europaea*	✓			
17	*Crassocephalum rubens*		✓		
18	*Vosia cuspidata*		✓		✓
19	*Veronica anagallis-aquatica*			✓	
20	*Ranunculus oreophytus*		✓		✓
21	*Rannunculus volkensii*				✓
22	*Ranunculus multifidus*				✓
Riparian wetland vegetation species in the ANAPA wetlands in 2013					
S/n	**Species scientific name**	**Ngongongare 1**	**Ngongongare 3**	**Mweka**	**Malama**
1	*Azolla nilotica*		✓		
2	*Bersama abyssinica*				✓
3	*Centella asiatica*	✓	✓	✓	✓
4	*Commelina benghalensis*	✓			✓
5	*Cynodon dactylon*				✓
6	*Cyperus exaltatus*		✓	✓	✓
7	*Cyperus articulatus*		✓		✓

Table 5.2: continued

Riparian wetland vegetation species in the ANAPA wetlands in 2013					
S/n	**Species scientific name**	**Ngongongare 1**	**Ngongongare 3**	**Mweka**	**Malama**
8	*Cyperus rigidifolius*		✓	✓	✓
9	*Fuirena pubescens*		✓		
10	*Kyllinga erecta*	✓			
11	*Leersia denudata*	✓	✓	✓	✓
12	*Ludwigia repens*		✓		
13	*Ludwigia stolonifera*			✓	
14	*Ranunculus multifidus*				✓
15	*Thunbergia alata*	✓			
16	*Veronica anagallis-aquatica*				✓
17	*Vigna parkeri*	✓	✓	✓	✓
18	*Vigna lateola*		✓		

Figure 5.4 compares the wetland plants diversity index in the four wetlands in 2013 and in 2019. The Malama wetland, which was not subject to water extraction and thus served as a control, recorded the highest wetland vegetation diversity as reflected in a riparian vegetation diversity index of almost 2 in 2019. This was also higher compared to an index of 1 that was measured in 2013. Overall, all wetlands measured higher wetland plant diversity index in 2019 than in 2013. Further, the average wetland vegetation diversity index across all wetlands in 2019 was 1.4, which is higher than the value of 0.8 that was recorded in 2013.

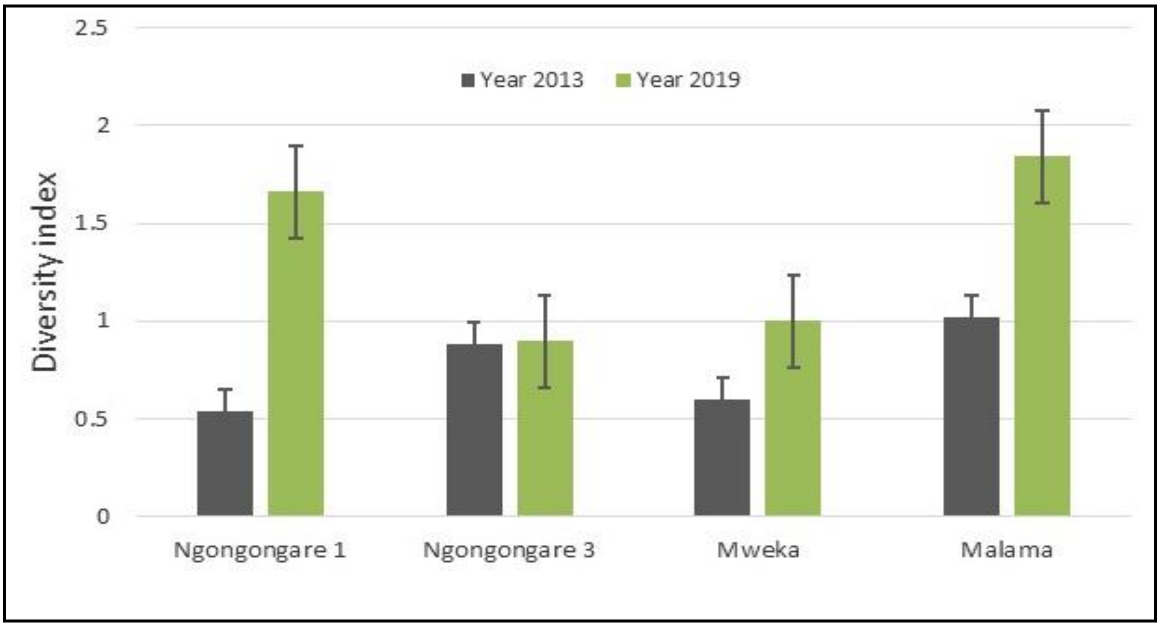

Figure 5. 4: Shannon-Wiener diversity index (±SE, n=5) for wetland plants in the four wetlands in ANAPA in 2013 and 2019.

Statistical analysis for wetland plant species richness, showed that the four wetlands differed significantly ($p<0.05$, $t=4.27$, $n=4$) between 2013 and 2019. There was a higher wetland vegetation diversity, i.e. species richness in 2019 than in 2013. During the field survey of 2013, Elisa et al. (2016) observed that Ngongongare 1 was completely dry in the dry season and recorded no flow, and had one of the lowest wetland vegetation diversity in the park. However, the same wetland recorded the second highest riparian vegetation diversity index in 2019, and significantly higher wetland vegetation richness ($p<0.05$, $t=5.24$, $n=5$) in 2019 during which water was flowing year-round. A similar difference ($p<0.05$, $t=3.01$, $n=5$) was evident in wetland plant species diversity between 2019 and 2013 in Malama wetland, which is a likely indication of improved water availability in 2019. Indeed a regression analysis indicated a significant ($p<0.05$, $R^2=0.7$, $t=3.7$, $n=8$) correlation between the amount of water available for the wetland environments and the wetland vegetation diversity indices in 2013 and 2019. While both diversity index and species richness were higher in 2019 than 2013, overall species richness provided a more robust evidence of improved wetland vegetation diversity in 2019.

5.3.3 Impact of water abstraction on riparian and floodplain vegetation in the low land semi-arid areas

Results in Figure 5.5 show that, dry season herbaceous plant ground cover at Ndarakwai wildlife ranch, significantly decreased with a decrease in distance to the Simba River. This pattern was evident for the transect 1 ($P<0.01$, $R^2=0.75$, $t=5.6$, $n=11$) and transect 2 ($p< 0.01$, $R2=0.77$, $t=5.8$, $n=11$) which were run within the first 2km from the river. In these transects, herbaceous ground cover increased with an increase in distance to the river at a rate of about 0.05%/m. Results from a transect located away (>5 km) from the river, did not indicate any significant pattern in herbaceous ground plant cover in relation to the river ($p>0.05$, $R^2=0.27$, $t=1.8$, $n=11$).

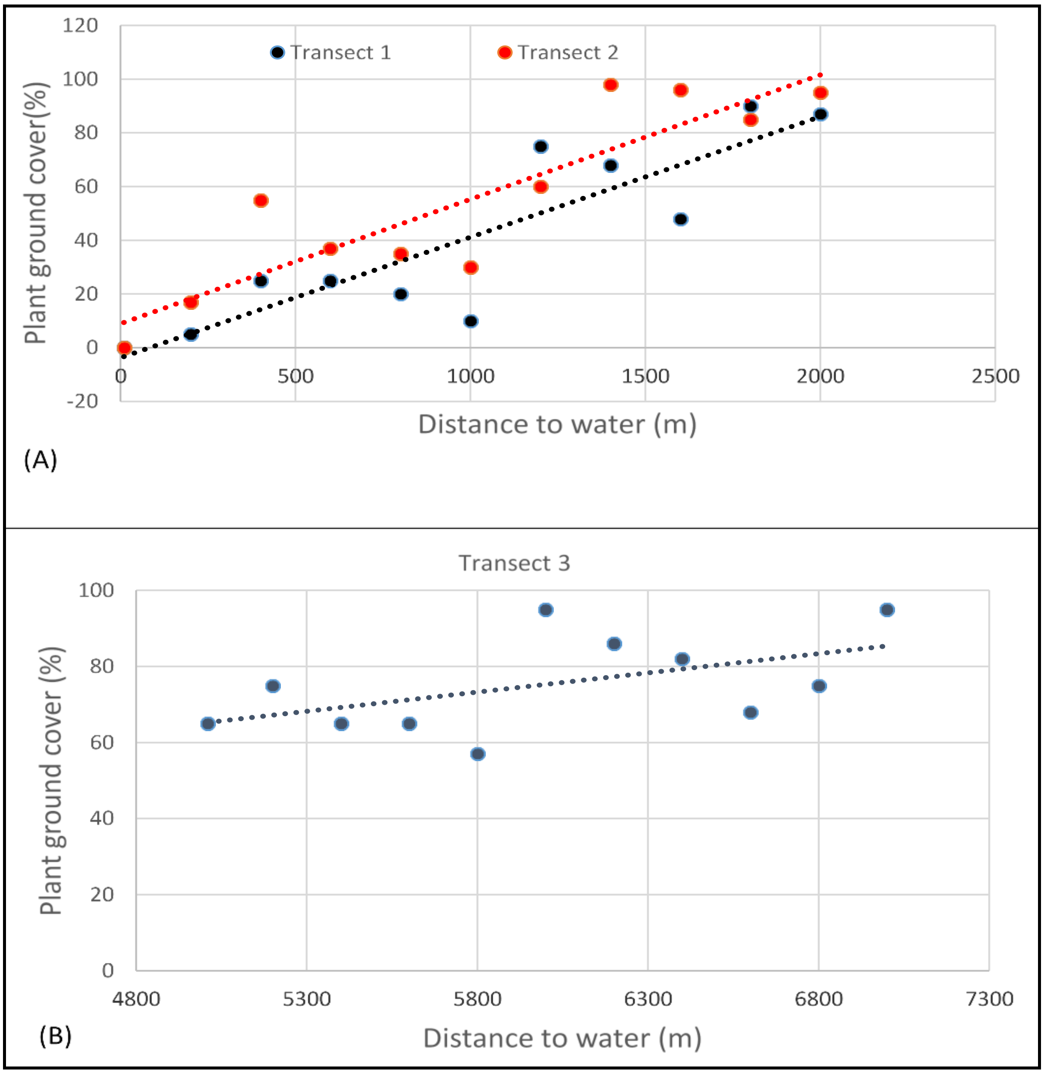

Figure 5. 5:Dry season variation in herbaceous plant ground cover in relation to the distance from the Simba River (A) Transects close to the river (B) Transect located away from the river.

Figure 5.6 shows the spatial patterns and the years of detection of the riparian and floodplain vegetation disturbances within a distance of 5 km (total buffered area 541 km^2) from the Ngarenanyuki River. This river originates from ANAPA and flows through agricultural village land. Vegetation disturbance exhibited both spatial and temporal variability over the assessment period of 21years from 2000 to 2020. The clusters of pixels identifying vegetation disturbaces (loss or reduction) were mainly concentrated close (1-2 km from the river) to the river in the village lands, i.e. the Olkung'wado village, the Madebe hamlet and Ngereiyani village (Figure 5.6). Disturbances were closer to the river channel downstream than in the upstream areas. Vegetation disturbance patterns were further associated with crop irrigation intakes and canals distributed along the Ngarenanyuki River.

More incidents of vegetation disturbances occurred in 2007 than in the other years studied (Figure 5.6 and 5.7). Other significant disturbances occurred in 2004 and in 2015, especially in the upstream village adjacent to ANAPA (Figure 5.6 and 5.7). The output from the LandTrendr analysis, showed that the level of vegetation disturbance along the Ngarenanyuki River was higher in 2013 than 2019 (Figure 5.6 and 5.7). This observation was also reflected in the level of riparian vegetation diversity in ANAPA established from the ground survey (Figure 5.4). On average, the vegetation disturbance events covered an area of about 0.8 km^2 per year within the buffered area around the Ngarenanyuki River. There was less vegetation disturbances further downstream, north of the Ngereiyani village, where the river flows into the Enduimet Wildlife Management Area (EWMA) (Figure 5.6).

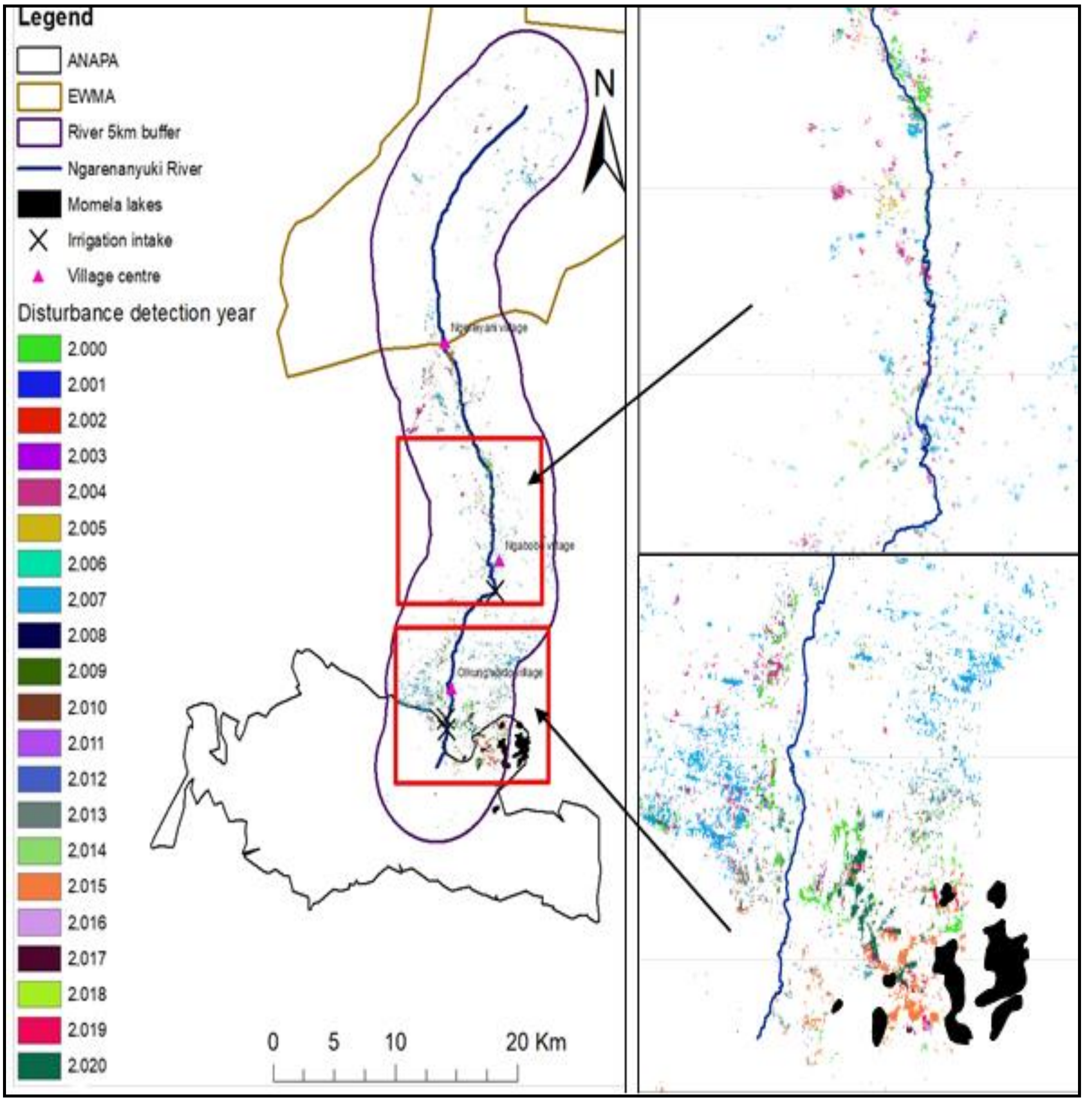

Figure 5. 6: Spatial patterns and year of vegetation disturbance occurrences around the Ngarenanyuki River from 2000-2020. The river originates from ANAPA.

Figure 5.7 indicates the temporal fluctuations of the area of vegetation disturbances from 2000 to 2020 along the Ngarenanyuki River downstream of ANAPA.

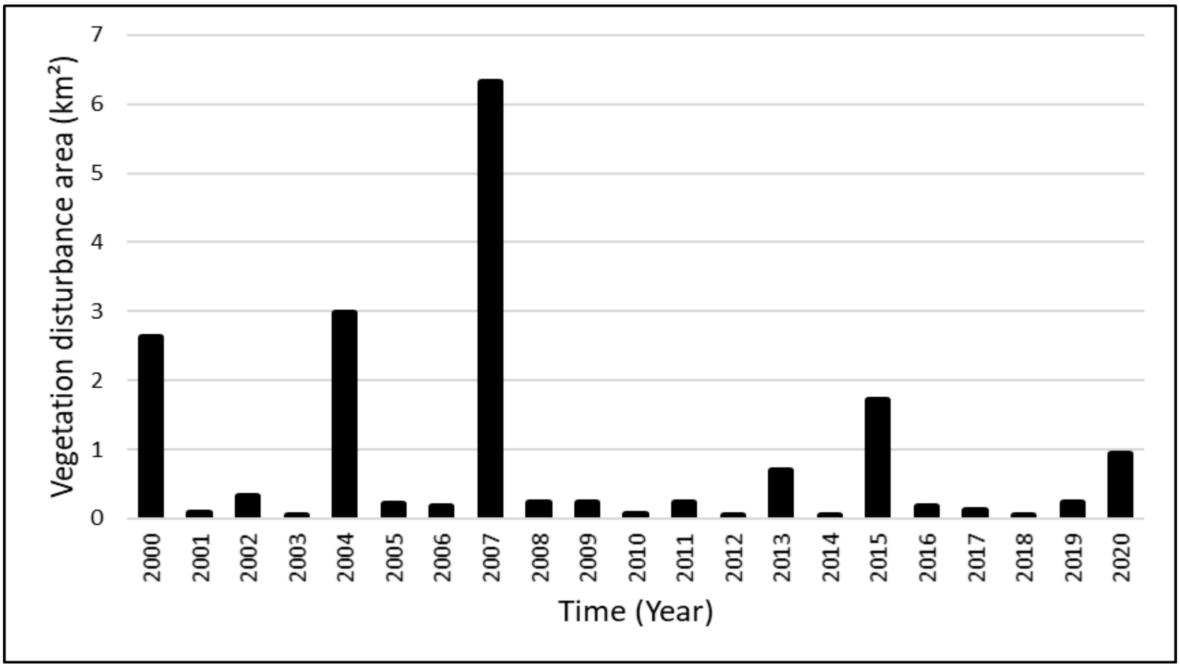

Figure 5. 7: Temporal trends in the area of vegetation disturbances in a 5 km buffer zone around the Ngarenanyuki River, downstream of ANAPA, as computed from the vegetation disturbance detection year data set.

Figure 5.8 shows (top) the magnitude of vegetation disturbance from 2000 to 2020 around the Simba River just downstream from KINAPA, and (bottom) the spatial patterns of vegetation disturbances and the years when such disturbances occurred further downstream where the Simba River traverses the Ndarakwai wildlife ranch and the Enduimet Wildlife Management Area (EWMA). A diverse spatial pattern of vegetation disturbance is evident, and much of it is evident in the upstream West Kilimanjaro forest where the river exits KINAPA. Looking at the years vegetation disturbance was most marked, a spatial-temporal distribution of the vegetation loss/reduction around the the Simba River at Ndarakwai wildlife ranch and EWMA is obvious (Figure 5.8 Bottom). In this case, vegetation disturbance clusters are scattered in the wildlife areas but largely increase towards the river channel. However, in the upstream areas(West Kilimanjaro forest) there was less disturbances towards the river.

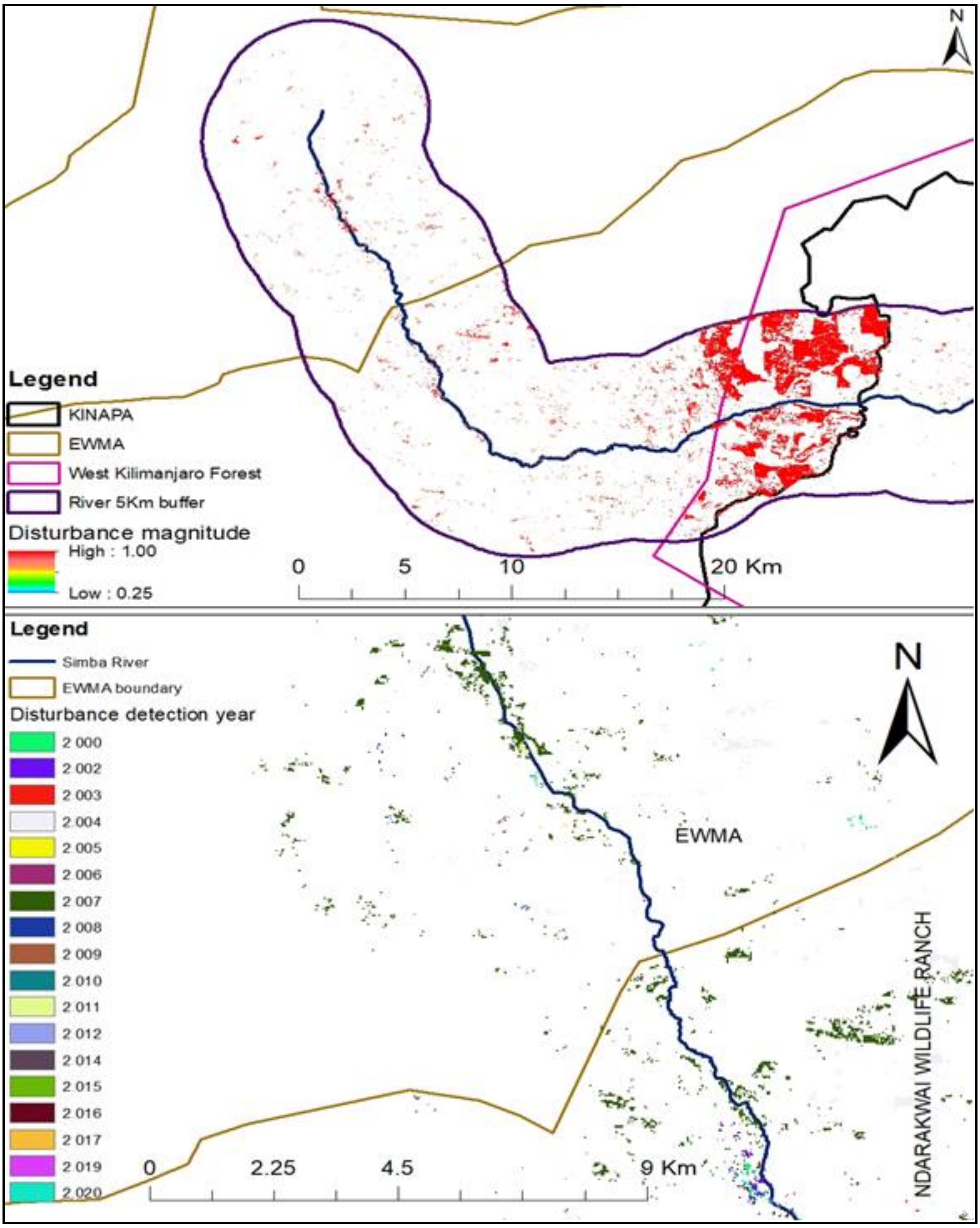

Figure 5. 8: (Top) Spatial pattern of vegetation disturbance severity within 5 km wide buffer along the Simba River from 2000 to 2020. (Bottom) Spatial patterns and year of vegetation disturbance occurrences around a 5 km buffer along the downstream section of the Simba River in Ndarakwai wildlife ranch and EWMA from 2000-2020.

Figure 5.9 shows the temporal fluctuations from 2000 to 2020 of the vegetation disturbance area (covering an area of 295 km^2) around the Simba River in Ndarakwai ranch and EWMA, which is a semi-arid wildlife rich area. The largest area of about 3.5 km^2 that experienced vegetation disturbance occurred in 2007. This was followed by an area of 1.4 km^2 in 2004. For the other years there were minor (< 0.5 km^2) disturbances in the vegetation . The average annual vegetation disturbance area was about 0.34km^{2}..

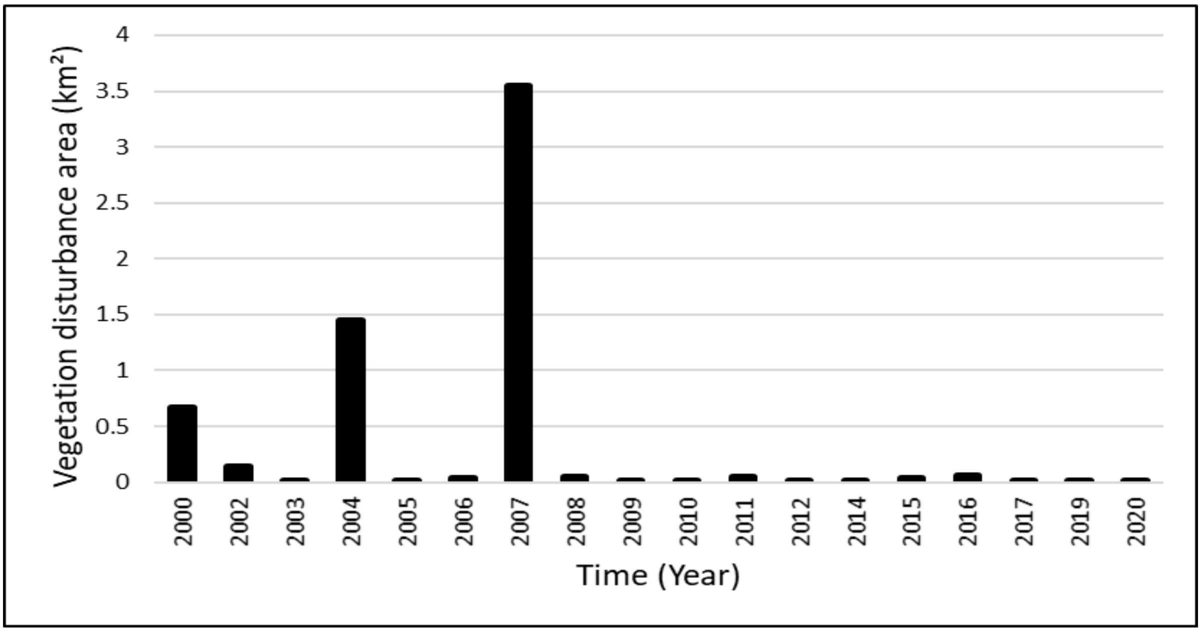

Figure 5. 9: Temporal fluctuations of the vegetation disturbance area in a 5 km wide buffer zone (total area 295 km2) around the downstream section of the Simba River in the Ndarakwai wildlife ranch and the EWMA , as derived from the vegetation disturbance detection year data set.

The spatial-temporal patterns of vegetation disturbance within a 5 km buffer around the southern part of the Lake Jipe on both Tanzanian and Kenyan sides, are shown in Figure 5.10. On the eastern side of the border, in Kenya, the area is protected by the Tsavo West National Park. On the western and south-western side of the border, in Tanzania, there is no protection. There was vegetation disturbance on both sides of the lake, but a much smaller disturbance on the Kenyan side compared to the Tanzanian side of Lake Jipe. The vegetation disturbance was most pronounced in the southern wetland which spans across both Tanzanian and Kenyan sides. The highest vegetation disturbance area (4.25 km^2) was recorded in 2007.

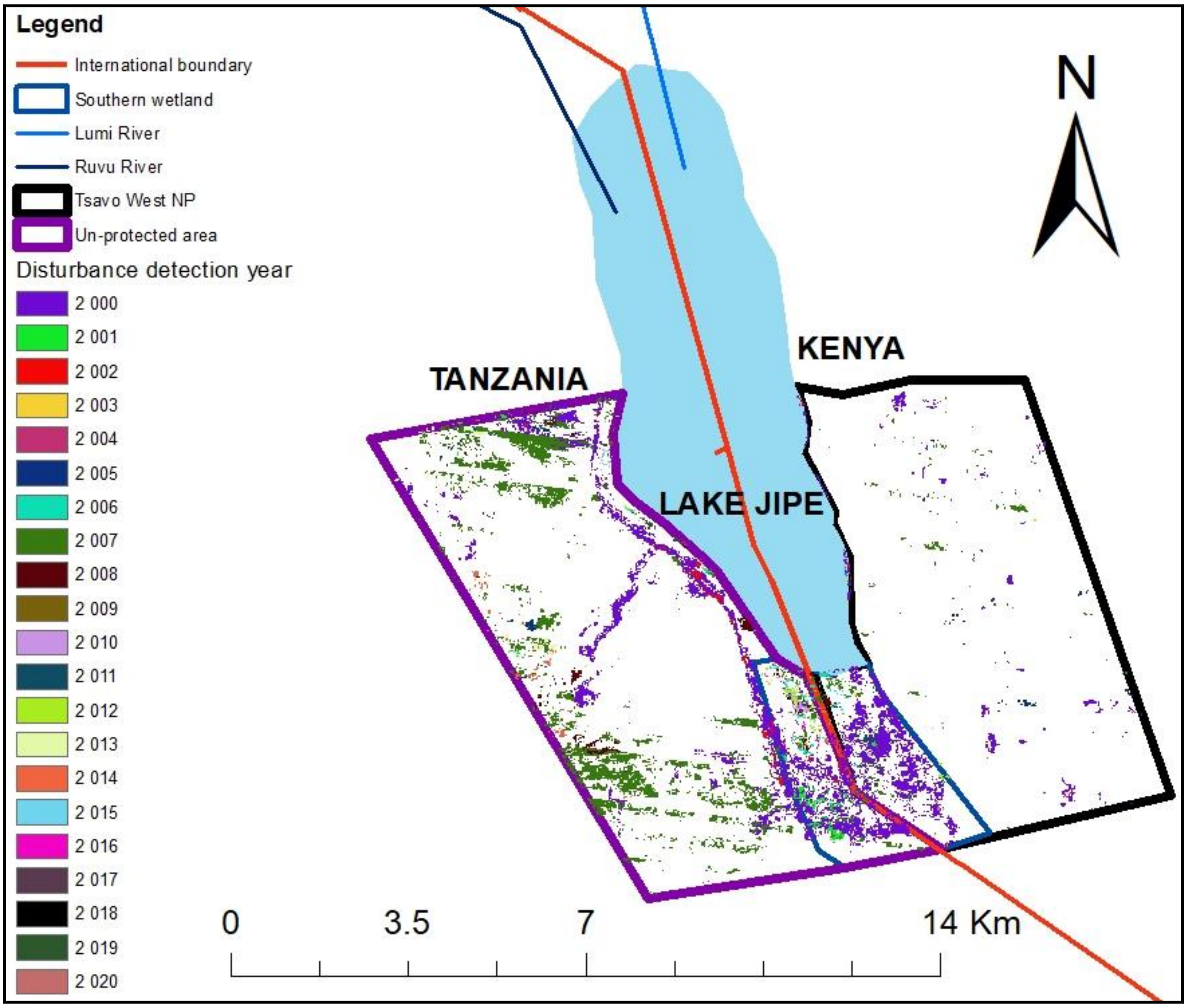

Figure 5. 10: Spatial and temporal vegetation disturbance in a 5 km wide buffer around the southern part of Lake Jipe on both the Tanzanian and Kenyan sides.

The vegetation disturbance area along the Simba and Ngarenanyuki Rivers largely varied in synchrony with mean maximum temperature, suggesting that temperature contributed to the change (Figure 5.11A). In the Ngarenanyuki River, there were also several vegetation disturbance events that coincided with a rainfall change, where for instance, a decrease in rainfall was associated with an increase in vegetation disturbance area, e.g. in year 2007 (Figure 5.11B).

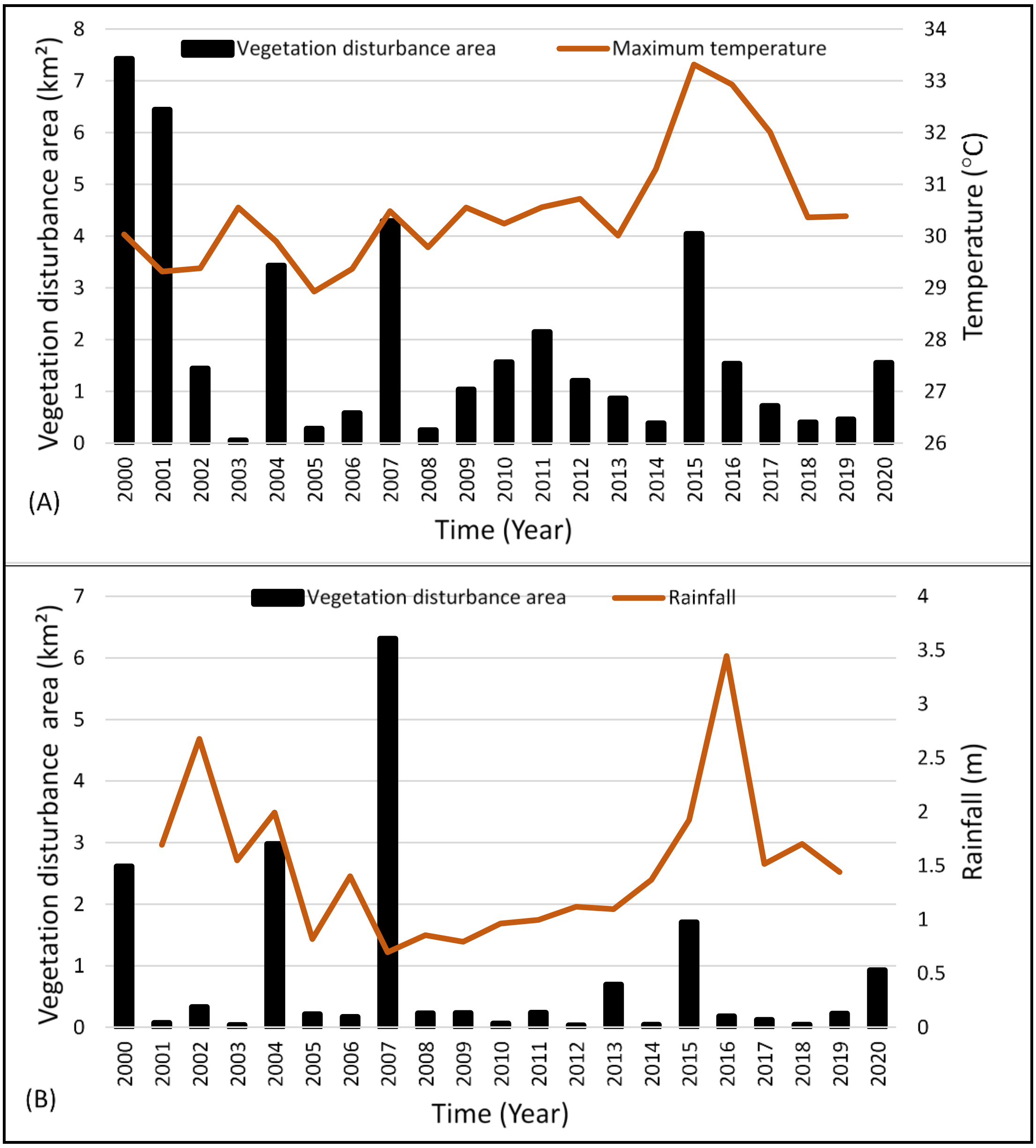

Figure 5. 11: Variation in vegetation disturbance area in (A) the Simba River and (B) the Ngarenanyuki River with mean maximum air temperature and rainfall. Temperature and rainfall data source: (Arusha National Park (ANAPA), 2020; Tanzania Meteorological Agency (TMA)-Moshi, 2020).

5.4 Discussion

The study investigated the change in riparian vegetation diversity in wetlands subjected to water abstraction in ANAPA, and herbaceous plant ground cover in relation to the Simba River at Ndarakwai wildlife ranch which is situated in the lowland semi-arid areas. It also investigated the spatial-temporal patterns of vegetation disturbances around surface water sources in the lowland semi-arid areas of the Kilimanjaro landscape. Together, these studies aimed to examine the impact of reduced water availability on the riparian and adjacent vegetation as a result of excessive water abstraction.

Available riparian vegetation in ANAPA wetlands were mainly from the Cyperaceae, Apiaceae, Commelinaceae, Onagraceae and Leguminosae families. Most of these families are common in East Africa and provide a desirable fodder for livestock and wildlife (Woldu, 2000; Kavana et al., 2019). Based on the results from the field data, the riparian vegetation diversity in ANAPA was higher in 2019 than in 2013, largely, as a result of increased rainfall in 2019 compared to 2013. All wetlands, in particular Ngongongare 1 and Malama, recorded a statistically significant increase in wetland plant species richness between 2013 and 2019, and the highest wetland vegetation diversity was found in Malama wetland which had no water extraction. The observed higher wetland vegetation diversity largely in terms of species richness is likely a reflection of improved water availability, i.e. improved environmental conditions (Meragiaw et al., 2018). The amount of dry season water flowing downstream the ANAPA wetlands was significantly ($p<0.01$, $F=5$, $n=6$) higher in 2019 than 2013 (see Chapter 2). For instance, Ngongongare1 wetland, which was completely dry during the dry season in 2013, recorded substantial amount of water in 2019 during both dry and wet seasons (Chapter 2). An increase in surface water availability was not due to less water being abstracted, as on average, during the study period, extraction in ANAPA took almost 90% of the available water in the dry season, but more likely it was due to an increase in rainfall (see Chapter 2). Hydrology is a major factor that controls wetland diversity and ecological integrity (Auble et al., 1994; Hamilton, 2002); this study also suggests that an increase in surface water availability was a likely reason for the higher wetland vegetation diversity in 2019 compared to 2013. The fluctuations in surface water availability result in an unstable hydrological regime that diversifies local environmental conditions and consequently increase wetland biodiversity (Nielsen et al., 2012). Such instability might also explain why the Ngongongare 1 wetland, which was dry in 2013, recorded the second highest wetland plant diversity in 2019. While wetlands vegetation richness is largely explained by variations in the hydrological regime, studies have also shown that factors such as water quality (e.g. salinity, pH) and soil type may also influence the diversity of wetland vegetation (Goslee et al., 1997; Woldu, 2000). However in this study, the soil type and water pH were unlikely to account for the differences in wetland plant diversity as the surveyed wetlands were all on similar volcanic alkaline soils (GRAIA, 2002) and previous analysis (Chapter 3) indicated no statistical difference in water pH among the surface waters of ANAPA. Other studies have indicated that diversity of wetland vegetation may also be reduced by salinity levels ranging from 1000 ppm to 5000 ppm (Hart et al., 1990; Nielsen, et al., 2003). However, all the wetlands surveyed in this study had in 2019 salinity concentrations that were below this range; Mweka (110 ppm), Malama (300 ppm), Ngongongare 3 (224 ppm), and Ngongongare 1 (227 ppm) (Chapter 3). Further, due to increased dilution that resulted from an increase in surface water, there was a decrease in salinity levels in 2019 when compared to the 2013; the salinity in Mweka decreased by almost 4%, at Ngongongare 3 by almost 36%, and at Malama by almost 20%.

Herbaceous plant ground cover around the Simba River in Ndarakwai ranch was found to significantly decrease with a decrease in distance to the river during the dry season (Figure 5.5). Such decrease was also visibly obvious during the dry season herbivore surveys. There was a substantial reduction due to excessive water abstraction in the Simba River flow as it flowed from upstream to the downstream. For instance, only about 5% equivalent to ~2 m^3/min of the total water available in the upstream (before abstraction) Simba River, reached Ndarakwai during the dry season (see Chapter 2). This situation created a severe water scarcity in the downstream semi-arid wildlife areas, and thus likely forced the wild and domestic herbivores to concentrate around the remaining little flow in the Ndarakwai ranch and EWMA (see Chapter 4). Such high increase in herbivore density around the Simba River in Ndarakwai caused a serious overgrazing and trampling of vegetation, likely leading to significant reduction in plant ground cover towards the river. The observed increase in herbaceous plant ground cover away from the river further confirms that the loss of vegetation cover was associated with water shortages in the downstream semi-arid areas.

Further, the LandTrendr analysis revealed spatial-temporal variations in vegetation disturbances around surface water bodies in the semi-arid lowlands downstream of ANAPA and KINAPA. While there were some patches of vegetation disturbances away from the water sources, most of the vegetation disturbances increased towards the scarce water sources, suggesting that these disturbances were largely linked with the remaining dry season surface water which attracted irrigation farming, livestock and wildlife grazing in the vicinity (Figures 5.6, 5.8 and 5.10). For example, vegetation loss was common around the Simba River in the Ndarakwai wildlife ranch, the Ngarenanyuki River in the rural agricultural lands, Momela lakes within ANAPA, and around southern part of Lake Jipe, especially on the unprotected Tanzanian side but also part of the Tsavo West National Park (Figures 5.6, 5.8 and 5.10). Focusing on Lake Jipe, the vegetation disturbance was much smaller in the protected area of Tsavo National Park in Kenya than in the unprotected areas both north and south of the border. Vegetation disturbance patterns in the Ngarenanyuki River section of the village agricultural lands seemed to be spatially related to areas with irrigation farming, suggesting that such loss was likely attributed to expansion of crop (in particular tomato, onions, cabbage) farming (Istituto Oikos, 2011). Largely this type of farming takes place immediately north of the ANAPA boundary i.e. Olkung'wado village, and in the downstream villages of Ngabobo, and Ngereiyani, and also Madebe hamlet. A similar situation was also evident along the downstream Simba River (Figure 5.8-bottom) where there is irrigation farming. In addition, severe vegetation disturbance(loss) occurred in the upstream (Figure 5.8-top) West Kilimanjaro Forest (part of which is currently a commercial plantation), all of which used to be a natural forest (Mjema, 2015). The natural forest was adversely impacted by deforestation and forest clearing for timber production, fuel wood and fodder collection, livestock grazing, expansion of forest plantation, settlement and farming activities since the 1950s (Mariki, 2015; Mjema, 2015). However, the lower part of this forest remains under forest plantation, where regular commercial tree felling takes place, followed by periods of crop farming until the forest canopy precludes farming due to lack of light. Immediately downstream the forest plantation, along the Simba River and in the neighbouring areas, the observed vegetation disturbance was likely due to extensive small- and large scale commercial irrigation farming for cereals, vegetable and legume production, which largely depends on irrigation water from the Simba River. Farming around the river took place especially in the villages of Mitimirefu, Kalimaji, Tingatinga and Roseline. This irrigation farming excessively abstracted water from the Simba River, leading to severe dry season water scarcity in the downstream areas (Chapter 2). As the Simba and

Ngarenanyuki Rivers flow downstream through agricultural land, they are subject to the effects of the expanding irrigation farming which often takes place adjacent the rivers (Istituto Oikos, 2011). Such unsustainable irrigation farming practice was one of the likely factors behind vegetation disturbance along the rivers. In contrast, the extent of vegetation disturbance in the upper sections of the rivers within the National Parks was insignificant as no farming activities are allowed in the parks. However, there were cases where vegetation disturbance occurred around surface water sources within the wildlife areas. For example, vegetation disturbance increased around Momela Lakes in ANAPA (Figure 5.6), around the Simba River in Ndarakwai wildlife ranch (Figure 5.8), and Lake Jipe in Tsavo West National Park (Figure 5.10). Such increase in vegetation disturbance is probably due to increased grazing and trampling by large (domestic and wild) herbivores seeking water. This is a likely explanation as when water availability is reduced in the ecosystem, it may aggravate grazing pressure around the remaining water sources (Meeson et al., 2002; Mtahiko et al., 2006). During the dry season fieldwork, my visual observations revealed that livestock and wildlife over-grazing and trampling pressure were particularly evident in the semi-arid areas especially around Lake Jipe and Simba River in Ndarakwai wildlife ranch, where the herbivores desperately searched for scarce drinking water. Surface water shortage in the dry season often forces a large number of livestock and wild animals to move to the few remaining water sources in the Kilimanjaro landscape (Ndetei, 2006; Kikoti, 2009). As surface water became scarce largely due to over-abstraction, a substantial dry season increase in animal abundance was evident near the semi-arid water sources and in the upper catchment in KINAPA water sources (Chapter 4). Such animal abundance was associated with overgrazing and trampling that degraded riparian and adjacent vegetation. Cattle heavily grazed the shore of Lake Jipe both inside and outside the Tsavo West National Park, especially the southern wetland, on both the Kenyan and the Tanzanian sides (Figures 5.10 and 5.12). Overgrazing by the dry season influx of livestock around the Lake Jipe is well documented (Ndetei, 2006; Waweru and Oleleboo, 2013). In addition to livestock pressure, the Lake Jipe ecosystem is threatened by other un-sustainable human activities including farming in the Lumi River basin, which is associated with deforestation, siltation and excessive abstraction of water from the Lumi River which supplies water to the lake (Ndetei, 2006; MEMR, 2012; Njiriri, 2016). The lake surface area, and hence the water level, has declined substantially during the dry season period in the last 20 years (Figure 5.3). Because of these un-sustainable human activities, especially over-abstraction of water from the Lumi River, the lake water level has substantially declined and the area has receded from the original 100 km^2 to about 30 km^2 (Ndetei, 2006). There is also an infestation by the invasive plant *Typha domingensis,* a species favoured by the lake shallowness and increased nutrients (MEMR, 2012). Therefore, human-induced water changes, particularly the reduced water flow to the Lake Jipe, might be the principal factor behind loss/reduction of the riparian and adjacent floodplain vegetation community around the lake.

Figure 5. 12: Overgrazing by wildlife and livestock along Lake Jipe in Tsavo West National Park during the dry season in 2019.

During the dry season when water is a limiting ecological factor, animal (especially water-dependent species) abundance and distribution in the Kilimanjaro landscape is largely determined by the availability of surface water (see Chapter 4), as is typical of dry tropical areas (Shannon et al., 2009; Ogutu et al., 2014). The concentration of animals around scarce water sources leads to overgrazing and trampling, which contribute to the degradation of riparian and the adjacent vegetation (Allsopp et al.,2007). This type of environmental degradation is common in the African dry ecosystems characterised by water scarcity. For instance, in the Lake Naivasha ecosystem in Kenya, overgrazing by livestock and wildlife primarily searching for water, has caused a loss of vegetation around the lake, soil degradation, erosion and lake sedimentation (Otiang'a-Owiti and Oswe, 2007).

Substantial reduction in dry season flow to the downstream Simba River section within the Ndarakwai wildlife ranch and EWMA forced a large number of animals to concentrate on the few available water pools, leading to high grazing and trampling pressure on the riparian and associated vegetation. A similar finding was reported for several rivers in Tanzania National Parks, including the Great Ruaha River in Ruaha National Park (Mtahiko et al., 2006; Mnaya et al., 2021).

Further, satellite data reveals a yearly occurrence of vegetation disturbance around surface water bodies in the dry season, with a large inter-annual variability (Figures 5.7 and 5.9). Along the Simba River, some of the vegetation disturbance events (data not shown) have lasted for more than 21 years and they coincide well with the human activities in the West Kilimanjaro forests as reported by Mjema (2015). Human activities in the forest declined in recent years after the government evicted more than 12,000 people in 2007 (Mjema, 2015). This decision was prompted by evidence that deforestation was deemed a threat to rainfall and the integrity of the glacier on Mount Kilimanjaro (Mariki, 2015). However, the eviction of people from the forest resulted in expansion of irrigation farming around the Simba River downstream of the forest, causing further extraction of water and conversion of more lands for crop cultivation (Chairperson Mitimirefu, personal comm). This phenomenon likely contributed to an increase in vegetation disturbances evident in 2007.

Inter-annual vegetation disturbance variabilities might also be contributed by climatic factors especially temperature and rainfall, both of which affect human and animal activities around the water sources (Figure 5.11). More water is consumed for irrigation, domestic use and animal watering during hot and drought years. Animals also often drink more water during this period to compensate for high rate of body water loss and a decrease in vegetation moisture content (Kleynhans, 1996). Therefore, under these hot and dry environmental conditions animals largely depend directly on the available surface water sources. For instance, vegetation disturbance area along the Simba River (Figure 5.11A) largely varied in synchrony with mean maximum temperature where an increase in vegetation disturbance area was associated with an increase in temperature, suggesting that a change in temperature contributes to a change in vegetation disturbance area. Similarly in the Ngarenanyuki River (Figure 5.11B), vegetation disturbance events often coincided with a rainfall change, for instance, a decrease in rainfall was associated with an increase in vegetation disturbance area. This further implies that a change in rainfall affects the amount of vegetation change. However, several observed contradictory patterns between vegetation disturbance and rainfall and/temperature suggest that other factors also play an important role in determining the extent of vegetation disturbance around surface water bodies in the Kilimanjaro landscape.

It is apparent that severe vegetation disturbances (e.g. Figure 5.8) around surface water bodies in the Kilimanjaro landscape have occurred for at least the last 20 years. Adoption of irrigation farming over this period, i.e. in the Ngarenanyuki and Simba River basins, has improved the living standards of the local communities (Istituto Oikos, 2011). However, the irrigation farming practice is unsustainable and over-abstracts water, and consequently has led to serious dry season water shortage in the downstream areas and to the emergence of resources competition including over-grazing and trampling of riparian and adjacent vegetation around the remaining scarce water sources. Reduced flows directly and negatively affect riparian vegetation (Richardson et al., 2007). Reduced dry season flows also aggravate grazing and trampling pressure by the livestock and wildlife, causing degradation of the riparian vegetation (Meeson et al., 2002), and also has the potential to delay riparian vegetation recovery (Vesipa et al., 2016). Unsustainable farming practices lead to a loss of riparian vegetation along river banks, which results in increased siltation which in turn reduce water retention capacity and increase the risk of flooding during the wet season. Ultimately all these changes damage both the riparian and terrestrial ecosystem (Drijver and Marchand, 1985; Ward, 1998; National Research Council, 2002; Ndetei, 2006; Ahmad, 2008; Istituto Oikos, 2011; Van Dijk et al., 2013)

As the riparian and floodplain vegetation represent a vital component of the ecological integrity and ecosystem services in the Kilimanjaro landscape, they must be protected by wisely managing water resources at the watershed scale through ensuring a balanced water and land resource utilisation for sustaining socio-economic and ecological needs. This will also strengthen the landscape resilience to climate change. In addition, there should be supportive human interventions such as widely distributed artificial water sources that provide water to the livestock and wild animals in order alleviate the grazing pressure around the few natural water sources in the landscape.

5.5 Conclusion

The integrity of surface water bodies such as rivers, wetlands and lakes, largely depends on the maintenance of the natural hydrological regime, which in turn supports the riparian vegetation and its associated biodiversity. This study evaluated the consequences of changes in surface water availability on the riparian and fringing floodplain vegetation in the Kilimanjaro landscape. The study revealed that, the diversity of riparian wetland vegetation in ANAPA is largely influenced by surface water availability dynamics as the change in the wetland vegetation diversity reflects changes in surface water availability. Indeed, an increase in surface water availability in 2019 compared to 2013 resulted in an increase in riparian wetland diversity. The lowland semi-arid areas, showed a diverse spatial pattern of vegetation disturbances, however most of these disturbances were concentrated around the few remaining dry season surface water sources. These lowland semi-arid areas have been experiencing substantial water shortages largely due to excessive water abstraction in the upstream areas. Reduced downstream water flow led to reduced inundation of riparian habitat and water resources competition among crop irrigators, and among the large domestic and wild herbivores. The abstraction of water and the removal of the natural vegetation for irrigation farming led to the aggregation of herds of large herbivores that overgrazed and trampled vegetation around water sources while seeking drinking water. This situation ultimately led to degradation of riparian and floodplain vegetation. Thus, the unsustainable water abstraction for irrigation farming and domestic and livestock watering not only exerts excessive pressure on surface waters but also adversely affect biodiversity in several ways including loss of vegetation around surface water bodies.

Therefore, to ensure sustainability of human activities and the environment, the water extraction must be carefully monitored and controlled to foster a sustainable balance of water use between human and biodiversity, and between upstream and downstream water users in the Kilimanjaro landscape. In addition, an adequate and properly distributed artificial water supply must be provided for livestock and wild animals to help alleviate pressure on the vegetation around existing natural water sources. This will also help to alleviate the risk of flooding in the wet season.

5.6 References

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Chapter 6: A synthesis

6.1 A developing water crisis in the Kilimanjaro landscape

Surface freshwater is vital for the survival and health of biotic communities and entire ecosystems, supporting both aquatic and terrestrial ecosystems, which in turn provide a wealth of ecosystem goods and services for humans. However, worldwide freshwater resources are under escalating human pressure, which poses a serious threat to ecosystems (Grafton et al., 2013; McClain, 2013). Human activities are threatening freshwater and related ecosystems globally, not only through over-abstraction but also through alteration of the surface water flow regime, pollution and land/habitat degradation and loss (Dudgeon et al., 2006; Richter et al., 2015). This is the case also of the Kilimanjaro landscape (Figure 6.1).

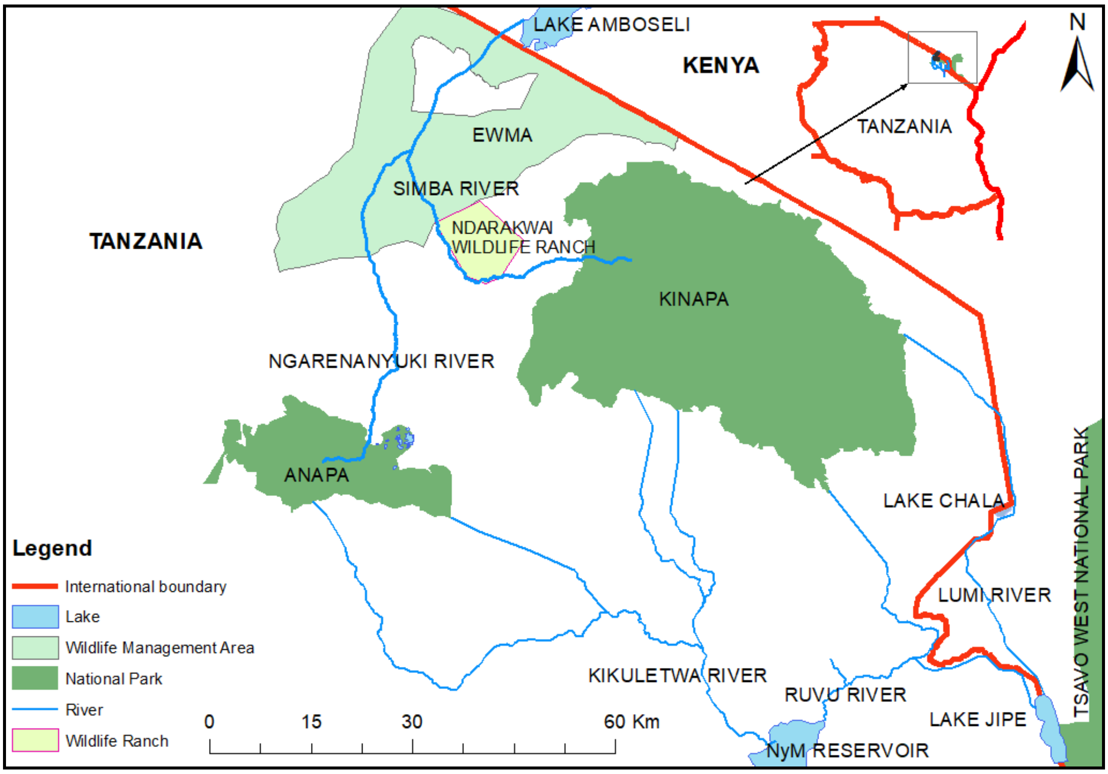

Figure 6. 1: A map showing the study area.

For decades, surface water has been extracted in the wildlife-rich Kilimanjaro landscape, the mountainous part of which serves as 'water tower', supplying freshwater to the ecosystem as well as the neighbouring human community. In common with similar habitats elsewhere in sub-Saharan Africa, the Kilimanjaro landscape is severely impacted by human-induced changes on surface water availability and quality (Chapters 2 and 3). This study demonstrates that the water abstraction at present rates is unsustainable and threatens the long-term availability of water resources both for human use and for the biota in the two National Parks; Kilimanjaro (KINAPA) and Arusha (ANAPA), and in the lowland semi-arid areas. Prior to this study, little was known of the surface water status, the extent of the human-induced change and its impact on the ecological integrity of the Kilimanjaro landscape. This thesis therefore examined the surface water status and the impacts of natural and human-induced surface water change on two key and interrelated components of the natural ecosystem; the riparian plant community and the large herbivore populations in the Kilimanjaro landscape.

In this study, I examined surface water availability in the landscape by measuring the amount of water available and how much is extracted, prioritising the most important lentic and lotic sources for the surrounding vegetation community and wildlife, specifically streams, rivers, waterholes and lakes (Chapter 2). I then examined the surface water quality, focusing on salinity, hardness (calcium and magnesium), fluoride, nutrients, dissolved oxygen, pH, temperature, and heavy metals (Chapter 3). After examining the availability and quality of surface water, and how it is affected by human and natural factors, I assessed how such surface water changes, influence the abundance, and distribution of large herbivores in the Kilimanjaro landscape (Chapter 4). Finally, I examined changes in riparian and adjacent floodplain vegetation resulting from surface water change in the Kilimanjaro landscape (Chapter 5).

6.1.1 Surface water availability

In the upstream sites in the forested mountainous National Parks, upstream of the extraction sites, seasonal changes in surface water availability were generally in phase with rainfall, reflecting the important role played by seasonal precipitation in the provision of water (Chapter 2). The National Parks play a critical role as water towers suppling freshwater to the natural ecosystem and the surrounding local and regional communities. The main water sources are streams, rivers, springs and groundwater. Groundwater from the National Parks is a most important contributor of freshwater for the neighbouring freshwater lakes including Lake Amboseli in Kenya, and Lakes Chala and Jipe, which are transboundary waterbodies shared by Tanzania and Kenya. Therefore, the status of water availability in the Kilimanjaro landscape is of international concern. There was a significant reduction in the river discharge with distance downstream, including within ANAPA and KINAPA downstream of the water extraction sites, and a further reduction in water availability was observed in the lowland semi-arid areas. Water has now become a critically scarce resource downstream of the upland villages during the dry season. Water scarcity was clearly due to excessive water abstraction taking place within the parks and outside the parks in the neighbouring upland villages, which on average took between 70% and 90% of the available water. There is no evidence that annual rainfall decreased in recent decades; in fact, it increased in Arusha National Park from 2013 to 2019 during my two study periods. Therefore, the present shortage of water is likely not compounded by climate change. I have shown, for the first time, that the existing water abstraction in the Kilimanjaro landscape is unsustainable as it causes serious water shortage and in some cases deprivation of water to the downstream areas within the National Parks and in the lowland semi-arid wildlife-rich areas which are also populated by people and their livestock.

In this thesis, I have demonstrated that surface water availability in the Kilimanjaro landscape is mainly influenced by escalating water demands for human use. The largest requirement for water results from irrigation farming, followed by domestic use and livestock watering (Chapter 2). Pressure on water supplies is driven by a rapidly growing human and livestock populations (Mbonile, 2005; Munishi et., 2009). The existing water abstraction rate was excessive, taking on average up to 90% of the available dry season water, and water availability was further decreased by poor water infrastructure such as the absence of water abstraction control gates, and storage tanks plus loss through leakages. The extraction occurred at the expense of the natural environment, affecting both vegetation and the wildlife in both the upstream and downstream regions of the Kilimanjaro landscape. In particular, the key naturally perennial Ngarenanyuki and Simba Rivers that originate from the high rainfall areas of ANAPA and KINAPA and that supply water to the lowland semi-arid areas, had their dry season flow substantially reduced downstream, with water now being available over only a small distance of about 20-30 km downstream from the parks boundaries. As a result, the further downstream semi-arid wildlife areas of the Ndarakwai wildlife ranch and Enduimet Wildlife Management Area (EWMA) were deprived of river water during most of the dry season. In addition, the large rivers in the lowlands were heavily sedimented, resulting in a reduction in water storage capacity and the flooding of the riparian areas. Siltation was noted during my study of the Simba, Ngarenanyuki, Lumi, Ruvu and Kikuletwa Rivers, the freshwater Lake Jipe and the Nyumba ya Mungu reservoir. Siltation was due to deforestation, soil erosion from poor land use including overstocking of livestock and poor farming practices, plus a decrease in flow rates (Ndetei, 2006; Lalika et al., 2015; Ndalilo et al., 2020). This entire situation has resulted in a serious water crisis for the wildlife, livestock and people.

The Kilimanjaro landscape is not alone in experiencing serious water crisis as other sites in Africa downstream of upland forested areas are also subject to water shortages. Similar cases have been reported in other tropical upland-lowland systems such as the slopes of Mount Kenya, the Upper Ewaso Ng'iro North River basin in Kenya and the Tarangire, the Wami and the Katuma Rivers in Tanzania, where over-abstraction of water upstream has led to a marked decline in downstream river flows, causing deprivation of water to people and the wildlife (Gichuki, 2002; Liniger et al., 2005; Mnaya et al., 2021). Further, excessive water abstraction in the Kilimanjaro landscape is also a transboundary issue that needs to be addressed. Indeed, due to excessive water abstraction, the dry season flow of the Ngarenanyuki River now no longer reaches the wildlife-rich Amboseli basin in Kenya (Istituto Oikos, 2011). In the Lumi River downstream flow is now substantially reduced, and this leads to less water entering Lake Jipe, and the Nyumba ya Mungu reservoir in Tanzania. It is especially important to address transboundary water issues as they are likely to have far-reaching social, environmental and geopolitical impacts. For instance, Kenya is proposing daming the Mara River that in the dry season in a dry year would intercept 100% of the river water leading to major environmental and social economic costs for the Serengeti ecosystem in Tanzania (Mnaya et al., 2021). Maybe this proposal could encourage Tanzania to use water diplomacy by restoring the river flows to the Amboseli basin in return for Kenya not drying the Mara River.

6.1.2 Surface water quality

The quality of surface water in the National Parks was generally good and within acceptable standards for the wildlife use (Chapter 3). This finding suggests that the excessive water abstraction in the parks did not significantly alter the water quality. However, the concentration of physicochemical parameters increased downstream and with the dry season and this was largely due to reduced flow and hence dilution, caused by the excessive water abstraction in the upstream areas. In some sites, the lowland semi-arid community and wildlife areas had poor water quality especially during the dry season. Higher concentrations of nitrate, phosphate, heavy metals, fluoride and salinity were mainly due to both agricultural run-off pollution and excessive water abstraction resulting in reduced dilution. The thesis has demonstrated that excessive water abstraction caused not only a severe shortage and deprivation of water to the downstream areas but also degraded the quality of surface water in the lowland semi-arid wildlife-rich areas. Reduced water flows resulted in less dilution, run-off of organic and inorganic materials from irrigated crop fields and increased residence time resulting in increased evaporation (Willis and McDowell, 1983; Abowei, 2010; Moeder et al., 2017). As a result, salinity and fluoride increased markedly, up to 1000 ppm, and almost 30 mg/l respectively in the Ngarenanyuki River, and up to 7700 ppm and 35 mg/l in the water holes within EWMA. Further, the water holes that were occasionally fed by the Ngarenanyuki River contained higher concentration of nutrients especially nitrate, (up to 480 mg/l) in the dry season. Levels of heavy metals especially iron and aluminium were also above acceptable limits for wildlife use, ranging from 10 mg/l to almost 200 mg/l in the downstream section of the Simba River, and in some of the semi-arid water holes (Chapter 3). Such high concentration were likely too high to support a healthy biotic community, particularly following prolonged exposure (Alloway, 2012). If this problem remains un-addressed, then wildlife health may be adversely affected. Adverse effects may also extend to people and livestock which consume the same water sources.

6.1.3 Impacts of surface water change on the ecological integrity

One of the greatest threats to the global biodiversity is the loss of habitat, including those that directly or indirectly depend on surface water (DeLong and Brittingham, 2009). Water plays a significant role to both aquatic and terrestrial species (McClain, 2013; Stommel, 2016). The over-abstraction of surface water often leads to deprivation or substantial shortage and alteration of downstream flow, and to the degradation of water quality, which in turn affects the ecological integrity. I have demonstrated in Chapters 2 and 3 that the dry season over-abstraction of surface water in the upstream areas is unsustainable, and it has caused a water crisis through shortage or deprivation of water in the downstream areas. Some deterioration of water quality in the downstream semi-arid areas was also observed, and this in part was due to the indirect effects of over-abstraction summarised above. I have further shown that this water crisis has adversely impacted the riparian and floodplain vegetation and the wildlife in the downstream areas (Chapters 4 and 5). This resulting water crisis thus leads to an environmental crisis (Duda and El-Ashry, 2000). In times of low rainfall, these crises are the most severe and extend from the upstream forested National Parks (see also Elisa et al., 2016) to the downstream semi-arid areas. A generalised linear mixed-effects model indicates that the abundance of large mammals increased with proximity to surface water sources during the dry season in the lowland semi-arid areas and also in the upstream mountainous Kilimanjaro National Park. As the downstream semi-arid wildlife areas such as the Ndarakwai wildlife ranch and Enduimet Wildlife Management Area (EWMA) were largely deprived of river water, the herds of wild large herbivores and livestock were forced to concentrate around the few remaining surface waters during the dry season period. This behaviour was observed not just in these wildlife areas but throughout the lowland semi-arid areas and the upstream Kilimanjaro National Park. Both saline and freshwater surface water sources were accessed during the dry season. This observation suggests that during the dry season the abundance and the distribution of large herbivores in the Kilimanjaro landscape was largely controlled by the availability of surface water rather than by water quality. However, saline sources were avoided during the wet season as freshwater became abundant.

Elisa et al. (2010) also found that water availability in the dry season was more important than water quality for large herbivores in the Katavi ecosystem. However, in the Ruaha National Park, Stommel (2016) found that both water quality and quantity influenced large herbivore distribution. In the Kilimanjaro landscape, water-dependent and grazer species, e.g., plains zebra and wildebeests were more associated with the available surface water sources than browsers such as giraffe and impala. The water dependent grazer species, especially zebra and wildebeest, remained and hence drank in the few remaining water points, including saline and mineral concentrated water holes and stagnant pools in the EWMA and the downstream sections of the Simba and Ngarenanyuki Rivers during the dry season. In the Kilimanjaro landscape these wildlife species were thus forced to tolerate of poor water quality and aggregated around such sources, a finding that is consistent to the findings from other sites by Rubenstein (2010) and Stommel (2016). The remaining scant water sources was subject to high salinity, fluoride and heavy metals, and further polluted by animal defecation and thus contained a high amount of nitrate. The ingestion of saline water (between 5000 ppm and 6900 ppm) may cause weight loss and reduced milk production in livestock, and it is thus unsuitable for pregnant and lactating animals (Mukhtar, 1998). Some of the water sources in the lowland semi-arid areas also had high fluoride levels, but the impact of high fluoride concentration on wild animals and livestock is poorly known and thus it deserves a further investigation. It is known however that a high fluoride concentration may affect the health and the distribution of biotic community (Kilham and Hecky, 1973; Shahab et al., 2017). An uptake of fluoride at a concentration of or above 35 mg/l is likely to cause fluorosis in animals over a long period (IPCS, 2012). Excess intake of aluminium and iron may impair animal physiology, milk production, species diversity and abundance over a long period of time (Coup and Campbell, 1964; Adeogun et al., 2012) during the dry season in the lowland areas of the Kilimanjaro landscape. It also affects the reproduction of mammals and avian species (Rosseland et al., 1990). There is also increased risk of infectious diseases transmission (e.g. contact-transmitted diseases such as foot and mouth disease) due to large aggregations of wildlife and livestock around scarce surface water sources, while the potential for other diseases (e.g., bacterial) transmission was likely high among both the herbivores drinking from these polluted scarce water sources (Ogutu et al., 2010; Strauch, 2013; Jori and Etter, 2016). However, there are no data on this risk.

The special case of elephants

The observed high nutrients levels may provide a conducive environment for the toxic cyanobacteria which have recently been linked to mass elephant die-off (~350) after drinking in stagnant surface water sources in the northern Botswana in an area similar to the Kilimanjaro landscape that is occupied by humans, livestock and wildlife (Kozlov, 2020). African elephants have high water demand but they are also browsers, and they maintained an intermediate distance to available dry season water sources in the semi-arid areas. Some elephants, in search for good quality water, likely moved from the lowland semi-arid areas through Kitendeni corridor into the upstream KINAPA as reflected by substantial increase in the number of elephants around water sources in the park during the dry season. Elephants are known to desperately search for good quality water when the available water becomes scarce and polluted (Ramey et al., 2013; Stommel, 2016). When water becomes scarce and of poor quality in the semi-arid lowlands of the Kilimanjaro landscape, the elephants search for clean water by moving into the neighbouring villages. This adaption strategy leads to serious human-wildlife conflicts. Such conflicts have culminated in the loss of livestock and wild animals plus human casualties, and huge socio-economic costs in the Kilimanjaro landscape (Kikoti, 2009; Mariki et al., 2015).

Developing human conflicts for water

The recent shortage and deprivation of water in the downstream areas is starting to cause human conflicts between the upstream users and the downstream users during the dry season in the Kilimanjaro landscape. This is evidenced by a recent conflict between the upstream Mitimirefu village and the downstream Tingatinga village along the Simba River (Chairperson, Mitimirefu village, personal comm.). There has also been several incidents of conflicts (including two major conflicts, one in 1990s and the other in late 2000s) between a downstream Ngereiyani village and upland Ngabobo and Olkung'wado villages due to excessive water abstraction taking place in the Ngarenanyuki River at the upland villages (Chairperson Ngereiyani village, personal comm.).

The wet season situation

This thesis also provided data showing that the large herbivores abundance increased with an increase in distance to water during the wet season. This is in direct contrast to the observations during the dry season. Thus, the thesis provides a clear evidence that accesss to water (which was mainly affected by abstraction) was the main factor underpinning animal abundance and distribution in the landscape during the dry season. This finding is in agreement with Kikoti (2009) who found that the large herbivores in the Loliondo area in Tanzania move away from areas of dry season surface waters to the drier areas of Lake Natron in the wet season. However, this observation also suggests that other environmental factors such as quantity and quality of forage may influence herbivore abundance and distribution, provided that the water need is at least largely satisfied. Therefore, the availability of surface water during the dry season appears to be the key factor controlling the abundance and distribution of herbivores in the Kilimanjaro landscape.

Vegetation changes

The thesis also provided a comparative view of the wetland vegetation in ANAPA between 2013 and 2019 (Chapter 5). Compared to 2013 (Elisa et al., 2016), there was in 2019 an increase in wetland vegetation diversity in the riparian wetlands of ANAPA. This corresponded with an increase in surface water availability. However, analysis of a short term data on dry season herbaceous plant ground cover around the Simba River in Ndarakwai wildlife ranch, and a long-term time series of Landsat images from 2000 to 2020, revealed a different picture downstream of the National Parks, in the lowland semi-arid areas of the Kilimanjaro landscape. I found a substantial reduction and loss of vegetation during the dry season in the riparian and the adjacent floodplain around the Simba and Ngarenanyuki Rivers which are the two main rivers draining the northern and western slopes of ANAPA and KINAPA respectively. I found a similar dry season disturbance of vegetation in the floodplains around Lake Jipe, both on its Kenyan and Tanzanian sides, though the vegetation disturbance was lesser in the protected area of the Kenyan Tsavo West National Park. From the Landsat images, I was also able to quantify the significant decrease of the wetted area of Lake Jipe due to a decrease in water level during the dry season over the period 2000-2020.

These findings demonstrate that the environment in the downstream areas is degrading due to a shortage and/or deprivation of water brought upon by an excessive abstraction of water in the upstream areas. This once again highlights the need to implement remediation measures, in this case to ensure sufficient water availability to maintain riparian and floodplain vegetation diversity. It is stressed that the substantial disturbance in riparian and adjacent floodplain vegetation documented in this thesiswas mainly caused by excessive water abstraction in the upstream areas within the National Parks and at the upland villages. This over-abstraction led to a water shortage and/deprivation, and associated overgrazing and trampling of vegetation by the wild animals and livestock primarily seeking for the scarce water resources in the downstream semi-arid areas. Further, the unsustainably practised irrigation farming using Lumi River water that supplies water to the Lake Jipe, has caused a significant decline in the size of the lake, infestation by the invasive southern cattail *Typha domengensis*, and the lake southern wetland to be substantially reduced due to availability of water and grazing pressure (Ndetei, 2006; MEMR, 2012; Ndalilo et al., 2020).

A call for remediation measures

These problems endanger the sustainability and the resilience of the Kilimanjaro landscape, including the National Parks. Indeed, the riparian and the adjacent floodplain vegetation not only provide an important forage and habitat for the wildlife, but they also support the hydrology and quality of surface water and ensure stability of the shoreline and riverbanks. The destruction of the vegetation results in the degradation of the ecosystem. Thus, these findings emphasise the developing water crisis which calls for urgent remediation measures to ensure sufficient water availability to sustain riparian and floodplain vegetation diversity, and the wildlife in the Kilimanjaro landscape.

6.2 Future projections

The temperature in Africa is predicted to rise 1.5 times more than the global annual average (Christensen et al., 2007). A substantial increase in temperature is also predicted in the Kilimanjaro landscape (Otte et al., 2017; Kishiwa et al., 2018) which will likely increase water loss by evaporation. Thus, the Kilimanjaro landscape may become affected by climate-related water stress on top of the already occurring human-induced water shortages (IUCN, 2007). The Kilimanjaro landscape is currently experiencing a rapid growth in human and livestock populations from natural increase and immigration (NBS, 2002, 2012; Mbonile, 2005; Munishi et al., 2009). This exerts ever increasing pressure on the water and associated natural resources (Munishi et al., 2009). The human population is expected to double in 20 years in the rural areas of the Kilimanjaro landscape (Mbonile, 2005). This will result in even more pressure on the water resources in the future. Human activities such as irrigation farming are projected to increase dramatically, and about 70% of irrigation water demands in the Kilimanjaro landscape will be unmet (Kishiwa et al., 2018). In addition, the currently and projected high livestock population implies that under a business-as-usual scenario, there will be more water consumption and also an increased degradation of riparian habitat as the livestock and the remaining wildlife concentrate around key water sources during the dry season (Mbonile, 2005). These changes will severely compromise the ecological functioning of the Kilimanjaro landscape.

The study has demonstrated how the ecology of the Kilimanjaro landscape is adversely affected by the existing excessive water abstraction both in the water towers (the forested mountainous National Parks) and in neighbouring downstream vilages. To the best of my knowledge, this is the first insight into the water-related ecological impacts with a focus on the wildlife-rich areas in the Kilimanjaro landscape, following a similar but short study conducted by Elisa et al. (2016) in the Arusha National Park.

6.3 Recommendations

The study proposes a number of solutions to improve water resource management and mitigate the ecological impacts of water abstraction, and then points out relevant areas that call for further research work. With the increasing human demands for water, sustainable development requires that water be allocated wisely to meet both human and environmental needs (McClain, 2013). This study recommends the following solutions to improve water resources management and mitigate the ecological impacts of water abstraction.

6.3.1 Scientific information, monitoring and control

In the Kilimanjaro landscape water allocation and abstraction within and outside the parks is often carried out without sufficient knowledge of the amount of water available in the different seasons, and on the potential environmental impacts of water abstraction. We need to understand the ecological water requirement so as to properly allocate water for a sustainable environment and development (McClain, 2013). Further, there is no monitoring and control of existing abstraction practices to determine their sustainability. Therefore, water allocation must be based ona rigorous data-set and, additionally, no water project should be established without being subjected to environmental impact assessment (Mbonile, 2005). Water abstraction must be subject to regular monitoring and strict control to ensure equitable sharing of quality surface water between the upstream and the downstream users including the wildlife and their environments (Mnaya et al., 2021). This is also a key means of addressing water-related conflicts (Gichuki, 2002). The government (and hence its relevant institutions) has a role to ensure (including through the allocation of funds) monitoring and the enforcement of sustainable water resource management regulations (Holler, 2014). However, the government should facilitate and encourage active involvement of the local communities and the protected areas authorities in each step of managing water resources to ensure success and sustainability of water resources management in the landscape. The National Park managements and the local communities should collaborate closely with the basin water authorities who are mandated to manage water resources including issuance, suspension and varying of water rights, and monitoring of water abstraction in the country (Mbonile, 2005). Where feasible, there should be an introduction of allowable abstraction, by setting up a cap on the amount of water that can be abstracted in existing and future water abstractions, to ensure sufficient and good quality water is allowed to flow to the downstream environments. Setting and enforcing caps for allowable abstraction will also serve as a key incentive for water use efficiency (Le Quesne et al., 2010) that is an essential component of any water management strategy.

6.3.2 Sustainability policies and implementation

The African water vision 2025 recognizes the need to allocate sufficient water for environmental sustainability in all nations and river basins (African Union, 2009), and it has been integrated into several national legal frameworks. Tanzania is among the countries with some of the globally best national water policies on matters related to sustainability after undergoing substantial policy reforms in the 1990s, and consequently it has adopted the Integrated Water Resources Management (IWRM) approach to water resources management (van Koppen et al., 2016). However, in practice, little has been achieved with respect to water governance and law enforcement for ecologically sustainable water resources management due to, among others, the inadequacy of knowledge, resources, capacity building, social and political will (Le Quesne et al., 2010; McClain, 2013; Mnaya et al., 2021). Indeed, there is a weak enforcement of the water and environmental laws in the management of water resources in Tanzania. The excessive water abstraction and the associated unsustainable farming and livestock keeping adjacent to the water sources, essentially violate the national environmental policies that mandate the provision of environmental flow and protection of catchment areas, including riverine and wetlands (URT, 1997, 2002). There is one exception to that statement, namely the the experience from the Katuma River in Katavi National Park. There, the Tanzanian governement intervened in restoring and conserving the river by removing illegal weirs and restoring the river chanel in 2016 (Elisa et al., 2021). From that experience we learned that the use of state governance is essential in controlling unsustainable water abstraction and practises such as the extensive use of numerours portable pumps, the use of unlined irrigation canals, the lack of gates in irrigation canals, the illegal abstraction of river water for irrigation farming, and the overstoking of liverstock next to rivers and lakes. This lesson needs to be applied to the Kilimanjaro landscape.

6.3.3 Community participation and capacity building

As a common good, water is effectively managed by the local communities that benefit mostly from it (Distaso and Ciervo, 2011; Dell'Angelo et al., 2016). There is therefore a need for capacity building to enable the local and indegenous communities to be able to actively participate in managing water resources and the environment in their areas. According to Ostrom's design principles, there should be among others,strong local water management institutions such as water user associations (WUAs) to ensure effective water resources management (Ostrom, 1990; Dell'Angelo et al., 2016). This would enable them to take an active responsibility in managing the local water resoutces (URT, 2002). According to the Tanzania water policy, the government is supposed to facilitate formation and functioning of WUAs. Unfortunately, most of the local water governance institutions in the country such as the WUAs are weak due to lack of capacity (Kabote and John, 2017). As the government is already under-resourced to effectively manage all water resources by itself, it is imperative to build the local institutions' capacity by providing them with professional, technical and logistical support (Holler, 2014). There are examples elsewhere in East Africa where empowered WUAs have been effective in monitoring, controlling and ensuring sustainable water abstractions in community lands. Such an example is in the Mount Kenya region (a landscape similar to that of Kilimanjaro), where WUAs have proved effective in managing water resources through raising community awareness on the water status, improving water flows and managing water use conflicts (Ehrensperger and Kiteme, 2005).

6.3.4 Coordination and integration

A coordinated and intergrated approach is a prerequsite for the effective management of water and biodiversity resources in the Kilimanjaro landscape. The problems are complex and thus they need this integrated approach, because it is a transboundary landscape consisting of several regionally shared water sources whose sustainability calls for a coodinated regional governance and integrated ecosystem management between Kenya and Tanzania (MEMR, 2012).

6.3.5 Water use efficiency

Water abstraction in the Kilimanjaro landscape is characterised by poor water infrastructure which results in the waste of substantial amounts of water (Mbonile, 2005; Istituto Oikos, 2011). For instance, some of the irrigation canals have no control gates and are not lined. Further, the recently established large-scale domestic water project that abstract water from the upstream Simba River and conveys that water to the Longido district is poorly constructed as it lacks in-transit storage tanks, leading to frequent leakage of the conveyance pipe as resulting water pressure is too great (Chairperson, Mitimirefu Village, personal comm.). In addition to the existence of poor water infrastructure, water is also wasted in flood irrigation that is ill-suited to this semi-arid area where water is a precious scarce commodity (Istituto Oikos, 2011). Therefore, one of the ways to address the existing over-abstraction of water is to ensure water use efficiency by improving water infrastructures and adopting efficient crop irrigation techniques such as drip irrigation and to grow crops that require less amount of water. This is also in line with the national environmental policy of Tanzania which emphasises the need for efficient irrigation water use (URT, 1997).

6.3.6 Exploit wisely alternative sources of water

In addition to establishment of water holes that are appropriately distributed across the semi-arid areas of the landscape, flood and rainwater may be harvested and stored in above or below ground water tanks. Such water will supplement water needs for irrigation and livestock watering during the dry season, and hence help in alleviating pressure on the existing natural water sources.

6.4 Limitations and further study

This study aimed to assess the ecological impacts of natural and human-induced changes in surface water in terms of its availability and quality, with a focus on the wildlife-rich areas in the Kilimanjaro landscape. While the aim has largely been achieved, there were several limitations. (i) The short and atypical seasonal patterns of rainfall, in particular the timing and extended length of the dry season. To address this limitation, and to address management needs, there is a need for a long-term monitoring program. (ii) The difficulty in quantifying the changes in water volume of river-fed water holes, which were fully under human control and operated on an intermittent basis only. The solution requires the use of manual gauging by trained local people or automatic loggers. (iii) The lack of long-term water level data limited the evaluation of water budgets and availability in the freshwater lakes to a short time period (20 years). For the future where water resources will hopefully be managed, a long-term ecohydrological monitoring programme needs to be implemented. Additionally, not all water sources, especially those in KINAPA, could be studied due to limited resources given the large expanse and difficult terrain of the park. However, based on the local knowledge, the study sites for this thesis are believed to be sufficiently representative for the wildlife-rich, leeward sides of Mounts Kilimanjaro and Meru. These regions, and the lowland semi-arid areas that they supply with water, have received minimal attention prior to this study.

It is suggested that further studies should include the following:

(i) The long-term assessment of the key surface water sources dynamics with respect to quantity and quality. This is important for understanding of key patterns and trends of surface water variations. Such an assessment is needed to improve the sustainable management of water and biodiversity. This study has provided the baseline knowledge and understanding on which to base such a future assessment.

(ii) An examination of economically viable methods to supply at least some of the demands for water in the dry season from water harvesting and storage during the wet season. Such methods could include underground/above ground tanks, plastic liners, and the sustainable use of groundwater.

(iii) Identifying the best mechanisms that can serve as strong incentives for water use efficiency in the upstream areas that will ensure a reasonable amount of water flow in the downstream areas. The aim is to devise ways that promote equitable water resource sharing to meet human and biodiversity needs in the upland-lowland system

(iv) The impacts of high fluoride on the wild herbivores and livestock in the semi-arid areas, especially in the West Kilimanjaro region. The aim is to understand the effects of high fluoride concentration on animal physiology and behaviour in the Kilimanjaro landscape.

(v) The behavioural and physiological change in large herbivores in response to deteriorating water quantity and quality in the lowland semi-arid wildlife-rich areas. The aim is to further understand the current surface water quality status in terms of biological and physicochemical characteristics, and how it might be affecting animal health, behaviour and productivity.

(vi) A detailed assessment of water-related animal migration and dispersal, the resulting human-wildlife conflicts, and possible mitigations measures.

(vii) Considering an optimisation of socio-economic and ecological productivity, examine the best ways for water allocation among different users and uses in the Kilimanjaro landscape.

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