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EGERTON UNIVERSITY

ASSESSMENT OF THE INFLUENCE OF BUJAGALI DAM ON THE SPATIAL-TEMPORAL FISH ASSEMBLAGES OF THE UPPER VICTORIA NILE RIVER, UGANDA

Master of Science thesis

by

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April 2016

DECLARATION AND RECOMMENDATION

DECLARATION

This thesis Report is my original work and has not been submitted or presented for examination in any institution.

Joshua Tazibirwa

Reg. No

Signature ____________________________ Date ____________________

RECOMMENDATION

This work has been submitted for examination with our approval as University supervisors.

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ABSTRACT

Large hydropower plants are known to be engines of economic development which unfortunately are also associated with a number of unpredictable environmental drawbacks at varying spatial-temporal scales. In this study, the temporal influence of Bujagali hydropower plant on the fish populations of the upper Victoria Nile river in Uganda was assessed. Historical data obtained through regular monitoring surveys from two upstream and one downstream transect between 2006 and 2015 was analyzed together with contemporary data obtained from the same areas and another further downstream transect. Contemporary data was collected between December 2015 and February 2016 using three gillnet fleets. The fleets each consisting of 13 pieces of 1"-8″ mesh sizes were set for two consecutive days each month, at three independent sampling sites located in each transect. Contemporary and historical data sets were merged and analyzed in R, PRIMER and SPSS softwares using ANOVA, average hierarchical clustering and non-metric multidimensional scaling (nMDS) techniques. A total of 1,377 fish specimens representative of 40 species and weighting 72.7 kg were recovered in the contemporary study. Abundance was dominated by haplochromines (67.3%) and together with *Lates niloticus*, *Synodontis afrofischeri* and *Mormyrus kannume* contributed almost 90% of the overall abundance. *M. kannume* and haplochromines combined contributed about 60% of the overall biomass, and along with *L. niloticus*, *Bagrus docmak*, *S. afrofischeri* and *S. victoriae* made up over 95% of the overall biomass recovered in the sampling period. A general temporal decline in biomass and diversity was observed both upstream and downstream of the dam. Remarkable temporal shifts in biomass dominance from small to large bodied individuals were also observed in the upstream areas and the reverse was true for the downstream regions. Riverine species *Barbus altianalis*, *S. afrofischeri* and *M. kannume* were the most vulnerable to damming but the latter two also showed high degrees of resilience to flow and water quality alternations respectively. The W-statistic values obtained through the Abundance Biomass Comparison (ABC) method were all negative and increased with time, suggesting an increasing disturbance from the dam on fish populations in the river. The results of this study indicate Bujagali dam as having notably changed the natural setting of fish populations in the study area. However a better understanding of the role of other human pressures in the observed dynamics is important for proper management decisions. *B. altianalis, S. afrofischeri* and *M. kannume* are recommended as the most suitable indicator species for dam impact monitoring.

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LIST OF ABBREVIATIONS AND ACRONYMS

ANOVA	Analysis of Variance
BEL	Bujagali Energy Limited
EIA	Environmental Impact Assessment
FAO	Food and Agriculture Organization
FIRRI	Fisheries Resources Research Institute
GPS	Global Positioning Systems
IPGL	International Post Graduate training in Limnology
MRC	Mekong River Commission
NaFIRRI	National Fisheries Resources Research Institute
NAPE	National Association of Professional Environmentalist
NBI	Nile Basin Initiative
PASW (SPSS)	Predictive Analytics Software
PRIMER	Plymouth Routines In Multivariate Ecological Research
TGD	Three Gorge Dams
WCD	World Commission on Dams

CHAPTER ONE
INTRODUCTION

1.1 Background

Large hydropower plants were until recently viewed as the most environmental friendly sources of energy to empower economic development. This assertion is in fact driven by a number of empirical roles they have played in human development. Hydropower plants store large volumes of water for human domestic and industrial consumption, provide large quantities of electricity for industrial development and above all are very crucial in the agriculture sector where about 12-16% of the world food production is directly accounted to them (Richter *et al*., 2010). Such mounting proof their positive impact on human societies is very hard to deny.

Nevertheless, a number of studies have suggested the benefits accrued from huge hydropower projects are often offset by the equally large environmental and social costs (Lejon, 2012). Hydropower schemes modify natural riverine flow regimes, coming with various concomitant effects (Bergkamp *et al*., 2000). For example flow regime modification alters the normal downstream transport of sediment and nutrients, leading to reduced primary productivity and general morphological setting of rivers (Powers *et al*., 2014). Hydropower plants fragment habitants, block migratory routes and cause fluctuations in thermal regimes in both upstream and downstream sections (Yang *et al*., 2012). In fish populations, such changes in riverine conditions are often manifested as reduced recruitment successes, alternations in species diversity, abundance and biomass, species extirpations and general reduction in commercial fish catches (Agostinho *et al*., 2008; Zhang *et al*., 2012).

The nature, scale and severity of impacts from large hydropower depend on many factors (Ward and Stanford, 1983). These factors can range from the geographical location of the affected system to the dam's location and form of operation among others (Bergkamp *et al*., 2000). For this reason, it is often largely difficult to predict the scale and severity of the impacts from hydropower plants. Of recent therefore, the early detection of these impacts has become one of the commonly used approaches to dam's ecological impact monitoring and management (Roni *et al*., 2005). The Abundance Biomass Comparison (ABC) tool is one of such methods which has proven very effective in detecting the impacts of hydropower plants at every early stages of their development (Penczak and Kruk, 1999). Originally developed for marine ecosystems, this method was built on the evolutionary principle of r and k-selection mechanism (Warwick, 1986). In undisturbed conditions, slow growing but large bodied k-selected species will flourish while the small opportunistic individuals will take over a system that has been disturbed. Unlike other ordination methods that may only detect changes in community structure without showing the direction the communities are taking (positive or negative), the ABC has proven successful in doing both (Yemane *et al.*, 2005). Besides being effective, the method is cheap, easy to apply and does not need the availability of baseline information for it to be used (Bianchi *et al*., 2001; Liu *et al*., 2013, Piperac *et al*., 2015). Considering the lack of baseline data for most hydropower projects on the African continent, the ABC tool presents the best opportunity for examination the response of fish populations to damming. However despite such advantages, its application on the continent's riverine systems is still limited.

Bujagali hydropower station is a 25 m high run-of-the-river power scheme whose approval and construction was received amidst controversy from different stakeholders (Luwa, 2007; NAPE, 2007; NAPE, 2015). The station is located on the upper Victoria Nile in the East African country of Uganda. Taken as a whole, the Nile river harbours over 128 fish species, some endemic and important to only sections of the river (Witte *et al*., 2009; Wohl, 2011). The upper Victoria Nile in particular is home to a number of species such *Neochromis simotes* which are threatened with extinction (FIRRI, 2000; Atkins, 2001; NaFIRRI, 2006). The fishery also offers a source of income and livelihood to a substantial portion Uganda's population. For example, the Victoria and Albert Niles reportedly produce more than 6,000 metric tonnes of commercial fish catches annually, generating over five million US dollars for the country (FIRRI, 2000; Mbabazi *et al*., 2012). However with Uganda's power demands by 2025 expected to be 2000 MW (Adeyemi and Asere, 2014), it is prudent the county also taps the equally high hydropower potential of this portion of the river. Unfortunately this is being done with little concerns for the both the short and long term environmental implications of such projects. Using Bujagali as a case study, this study aimed to investigate the spatial and temporal implications of such projects on the fish populations of the river. The study also intended to examine the effectiveness of the ABC tool in detecting the impacts of hydropower plants on large tropical river ecosystems in Africa. Such a study is vital in generating information needed in the current and future decision making process on energy development in the country

1.2 Statement of the Problem

The temporal and spatial scales of hydropower project impacts are seldom precisely determined yet they are important in guiding decisions on impact mitigation measures. Because of their complex nature, impact monitoring strategies for hydropower plants require that adverse array of methods are employed at differing time and spatial scales. In Uganda, impact monitoring for Bujagali hydropower plant is undertaken in a 10 km stretch of the river. However, it still remains unclear on what is happening in further upstream or downstream sections beyond this monitored stretch. Although monitoring studies are yet to reveal a significant impact of the dam on fish populations, the conclusions are based on impact monitoring techniques that may not effectively detect the relatively small changes in fish populations that occur in the early stages of dam construction and operation. On addition to examining if Bujagali hydropower project's impacts may possibly be manifesting beyond the monitored river stretch, this study also aimed at applying another impact monitoring tool on the same data sets to test its effectiveness in detecting changes in the fish populations due to the construction and existence of the dam. The information generated will be useful in guiding decision-making processes aimed at improving the effectiveness of current and future impact monitoring approaches while also guiding on priorities to consider in choosing mitigation measures.

1.3 Objectives

1.3.1 General objective

The overall objective of this study was to assess the influence of Bujagali hydropower dam on the spatial-temporal fish assemblages of the upper Victoria Nile River.

1.3.2 Specific objectives

The specific objectives of this study were to:

i) Determine the species abundance of fishes at Kalange, Reservoir, Buyala and Kirindi at the Bujagali Hydropower Station.
ii) Determine the size structure and diversity of the fish populations at the sample sites
iii) To determine and compare spatial-temporal changes in the abundance, diversity and size structure of fishes in Bujagali dam in the period 2006-2016.

iv) To determine the degree of impact from Bujagali dam on the abundance of fishes in the upper Victoria Nile river in the period 2006-2016

1.4 Hypotheses

i) There is no significant difference between the fish species abundance at Kalange, Reservoir, Buyala and Kirindi.
ii) There is no significant difference between the contemporary size structure and diversity of fishes at Kalange, Reservoir, Buyala and Kirindi and those of the year 2006.
iii) There is no significant difference in the annual changes in abundance, diversity and size structure of fishes in Bujagali dam impact monitoring sites in the period 2006-2016.
iv) There is no significant difference in the degree of impact on the abundance of fishes in the upper Victoria Nile from Bujagali dam in the period 2006-2016.

1.5 Justification

The upper Victoria Nile in Uganda is home to a number of fish species such as *Neochromis simotes* and other un-described haplochromine cichlids adapted to particular habitats whose distribution is currently only known to occur in that section of the river. Therefore, altering such habitats may lead to loss of some species before they are described. Moreover, it also harbours other fish species known to occur in other water bodies in Uganda but are currently facing extinction due to various forms of pressure. The Upper Victoria Nile ecosystem is therefore critical in fisheries biodiversity conservation and all necessary efforts should be geared towards keeping its ecological integrity intact. Studies have also identified five commercially important fish species for the fishery of the Upper Victoria Nile, of which three (*B. altianalis*, *M. kannume* and *B. docmak*) prefer lotic environments. Bujagali dam is located in the fast flowing section of the river and is thought to modify the flow regimes and may possibly make the environment unfavourable for some of these species. Studies aimed at looking at how such riverine specialist species are responding to the dam are crucial in the development and implementation of mitigation measures while also emphasizing the spatial scale at which the measures should be implemented.

CHAPTER TWO
LITERATURE REVIEW

2.1 History of damming of rivers

Although human damming of riverine environments dates back to a long time in history, it was not until the 20th century that this practice intensified to the concern of aquatic ecologists (Mauch and Zeller, 2008; Mohammed-Aslam and Balasubramamian, 2010; Conniff *et al*., 2012). Initially, dams were built for the purposes of either increasing available water for agricultural and domestic use or flood control, however with industrial and technological advancement in the 19th century, dams were now being built mainly for hydropower generation (Henshaw, 2011; Alexander *et al*., 2012).

Bergkamp *et al*. (2000) classifies dams depending on their mode of operation as storage/impoundment and run-of-the-river dams. According to the World Commission on Dams (WCD), impoundment hydropower dams are the ones that hold back and store river water in a reservoir with or without river diversion, while run-of-the-rivers hydropower dams do not store river water behind them and operate with or without channelling part of the flowing river water (Dursun and Gokcol, 2011). Since run-of-the-river hydropower schemes like Bujagali form either only small or no reservoirs, it is generally conceived that they are ecologically less damaging than their counterparts (Bujagali Energy Limited (BEL), 2006). However, there is growing number of studies that suggest run-of-the-river dams may have as just disastrous impacts to fisheries as do storage dams (Anderson *et al*., 2015). For example, on the Mun River in Thailand, the construction of Pak Mun, a run-of-the-river hydropower station had by the year 2000 resulted in the disappearance of over 16 of the 112 migratory and rapid habitat dependent species of fish (World Commission on Dams (WCD), 2000).

2.2 Impacts of dams on physico-chemical parameters and primary producers

The physical and chemical components of any riverine environment are the first to be impacted upon its damming before the effects can be manifested in any form of biota. For example, the frequency, degree and magnitude of flood peaks and sediment transport downstream have been reported to reduce after the damming of most rivers. In China, 94% of the 142 x 10^9 kg sediment flux into the Mekong river basin is expected to be trapped behind dams after the completion of all eight dams between Gonguoqiao and Mengsong (Kummu and Varis, 2007), while the natural flooding rates of more than half of USA's rivers have been reduced by about 25% due to water held back behind dams (Fitzhugh and Vogel, 2010). Changes in natural flow dynamics are known to subsequently interfere with nutrient transport thus inhibiting the primary productivity of the system. For example in the USA, nitrogen and phosphorous transport loads downstream of dammed rivers in the Missouri basin has been reported to reduce by 16% and 33% respectively (Brown *et al.*, 2011). This upstream nutrient retention often affects the natural distribution of primary producers as observed in the Yangtze River in China where algal density in the three Gorge Dams reservoir increased from $0.25\text{-}32.70\times10^4$ cells L^{-1} to 2.73×10^6 cells L^{-1} a year after closing of the dams (Zeng *et al.*, 2007).

As flow regime is a critical determinant of thermal dynamics in lotic environments, its modification concomitantly results in changes in the temperature regimes of the river both in the upstream and downstream areas. Upstream of the rivers, the reservoirs created by storage and some run-of-the-river dams are always deep. For instance the small reservoirs created by Bujagali hydropower station in Uganda have maximum depths of over 30 metres (NaFIRRI, 2015). Mixing in such deep reservoirs will always be limited, releasing water of different temperatures downstream depending on the different seasons. A case in point is the Xiahushan reservoir on East river China, where release water has a temperature that deviates from that of the natural river by a margin of 2.3°C depending on the season (Yang *et al.*, 2012).

2.3 Hydropower dam impacts on fish

The most obvious direct and short term effect of dams on fish is the blockage of their migratory routes. This is especially a problem for the migratory fish species which migrate upstream to spawn in the fast flowing waters (Holden, 1979). Although upstream fish passages have been suggested to eliminate this problem, deficiencies in their effectiveness have been reported in some studies. In some cases, dams may cause delays for fish in accessing suitable spawning and feeding sites; for example, Thorstad *et al.* (2003) observed delays of about four days in migrating Atlantic salmon populations in Nidelva River, Norway. Such delays at barrier hydropower plants in Allier river, France have reportedly been associated with an almost 45% increase in mortality in migrating Atlantic salmon spawning populations during summer time (Baisez *et al*., 2011). Hydropower plants also affect riverine fish populations by deterring their recruitment success. This can be through a high juvenile mortality rate in turbines as juvenile fish try to migrate downstream (Keefer *et al*., 2012). Recruitment success can also decline due to reduction in downstream flood peaks as reported in the Parana river, Brazil, where almost 50% reduction in fish recruitment success has been observed (Agostinho *et al*., 2001).

Sometimes dam-induced changes in the hydrology and flow regime of rivers may make the environment hostile for fishes that thrive in running waters. Such disturbances are in most cases manifested as declines in commercial fish catches, changes in species composition and in extreme cases, localized or regional extinctions. For instance, in the Volga river, Russia, commercial fish catches have declined from over 80% in the pre-damming era to less than 5% over a period of years (Górski *et al.,* 2011). Moreover recent studies are associating ecological changes in upstream areas of dammed rivers to downstream dam reservoirs. For example, some authors believe the disappearance of *Notropis stramineus* and *Cyprinella lutrensis* fish species from some tributaries of Little and Red rivers in Oklahoma USA is possibly due to the damming of the rivers (Mathews and Marsh-Mathews, 2007; Franssen and Tobler, 2013). These reports follow related observation by Greathouse *et al*. (2006) who observed food abundance in tributaries upstream of dammed rivers in Puerto Rico as being tenfold less than that of their counterparts on un-dammed rivers.

As much as most studies have mainly documented negative influences of dams on riverine fish populations, a number of others have also shown how dams can improve the productivity. For example in Australia and Brazil, Tonkin *et al*. (2014) and Albrecht *et al*. (2009) observed the growth and recruitment of *Macquaria australasica and Brycon gouldingi* in Dartmouth and Serra da Mesa dam reservoirs respectively to be related to the reservoirs' filling. The fisheries enhancement has also been observed in tailwaters behind the dam. In the tailwaters of Cumberland river Wolf Creek Dam USA, Dreves *et al. (*2014) reported an increase of trout fish catches per unit effort from 50 to 300 fish/hour, 20 years after the river was impounded. Such reservoir and tailwater enhancements are believed by some authors to compensate for the dam-induced reduction of the riverine fisheries production (van Zwieten *et al*., 2011).

On the Nile River, lessons on how the dams could affect its fishery can only be obtained from observations on Aswan high dam in Egypt, as it is probably the only dam on the river whose impacts have been monitored long enough (Jackson and Marmulla, 2001). Upon completion of the Aswan in 1965, annual total fisheries yields downstream declined from about 25,000 tonnes to only 5,000 tonnes by 1975. However, twenty years later, the total fisheries production had risen again to over 20,000 tonnes, for which the cause are still uncertain (McCall, 2008). The location of a dam on the river is one of the most crucial factors in determining the degree of its effects. It is highly unlikely that a dam like Aswan located in the extreme downstream section of the Nile River may have similar effects as do those located in extreme upstream sections of the same river (Ward and Stanford, 1983).

In the developing countries like Uganda, fisheries are pivotal in the economic development and supporting the livelihoods of a substantial portion of their human populations. For example, Uganda's fisheries industry employs about 1.2 million people, generating over 12.5% of the agricultural GPD in the country (Musinguzi *et al*., 2015). With the continuing decline in the lacustrine fishery amidst an increasing population growth, proper management and enhancement of fluvial fisheries could perhaps offer a cheaper and more sustainable alternative. Despite the growing knowledge and scientific information on how hydropower plant development may alter productivity and fish production, this topic has received very little attention. For example, in the socio and environment impact assessment studies for the under-construction Isimba dam, almost no attention was given to the ecological implications of the dam especially with regard to the river fishery (Ministry of Energy and Minerals Development (MEAMD), 2013). Whereas the economic benefits that come with hydropower developments are known, a better understanding of both the spatial and temporal scales of their effects is crucial; especially for a country like Uganda whose human population dependence on the natural environment is still very high.

CHAPTER THREE
MATERIALS AND METHODS

3.1 Study area

Bujagali is a 250MW production capacity low-head ROR power plant located on the Upper Victoria Nile Uganda East Africa. As far as fisheries biodiversity is concerned, the Upper Victoria Nile is of special importance as it harbours a number of species thought to be either extinct or extirpated from the main Lakes (Balirwa *et al*., 2003). The dam is also located about eight kilometres downstream of two storage hydropower stations Nalubaale (formerly Owen falls) and the bypass canal created Kiira dam. About 40 and 100 kilometres downstream of Bujagali dam are two other large run-of-the-river (ROR) dams Isimba (180MW) and Karuma (600MW) which were under construction by the time of the study. The design of the Bujagali power plant is in such a way that the turbine is close to the weir structure, which allows water to be returned to the weir pool thus depletion of the river's flow is avoided (Robson, 2013).

Construction of the Bujagali was first proposed in the mid 1990s and the first ecological baseline surveys undertaken in 2000 at a quarterly interval in the months of February, May, August and November. After a period of delays, the construction of the dam was later commissioned in September 2007 after another ecological survey was undertaken in April 2006 to supplement the four surveys done in 2000. In both the 2000 and 2006 ecological baseline surveys, sampling was undertaken at each of the four transects; Kalange-Makwanzi, Buyala-Kikubamutwe, Kirindi-Matumu and Namasagali-Bunyamira (hereafter called Kalange, Buyala, Kirindi and Namasagali) which was the furthest downstream sampling transect. Upon inception of its construction in 2007, sampling for impact monitoring was undertaken at all baseline survey transects. However in 2008, Kirindi and Namasagali were eliminated from the monitored transects, allowing the activity to continue at only Kalange and Buyala. Between 2006 and 2008, surveys were undertaken once annually in the months of April (2006 and 2007) and September (2008). However since 2009, the impact studies have been undertaken bi-annually in April and September) at all the three transects.

Although it is clear that the closing of the dam in 2012 may have created a reservoir which extended up to Kalange, the location of this transect close to the fast flowing tailwaters of Nalubaale and Kiira dams makes it behave as a transitional zone (Thornton *et al*., 1990). For that reason, sampling for monitoring reasons was also allocated to a section in the immediate upstream vicinity of the dam wall where an island was blasted to make a storage area. The conditions in this transect (hereafter called the Reservoir transect) were characteristic of reservoirs created behind storage dams thus being considered as an independent transect.

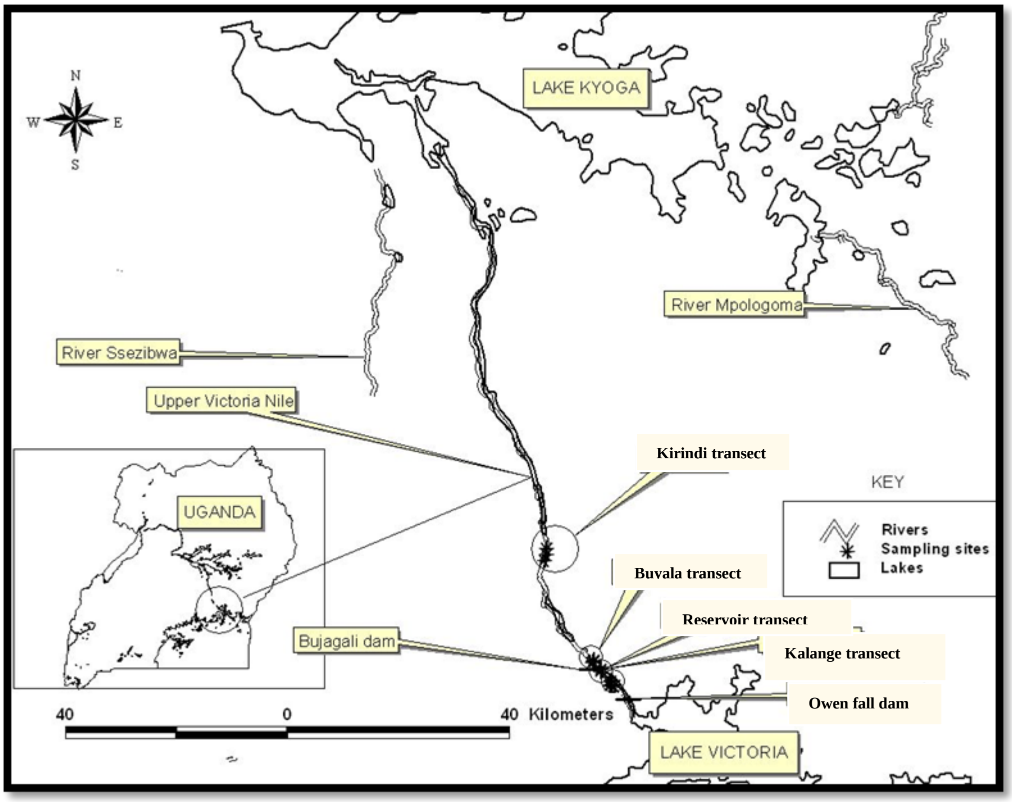


Fig 1: Location of the sampling transects and sites on the upper Nile river (Modified from: NaFIRRI, 2006)

3.2 Description of the sampled transects

3.2.1 Kalange.

This transect was located about 6km upstream of Bujagali dam at GPS coordinates 36N 0516559E. According to FIRRI (2000), the transect originally had five islands, but all were submerged as the water levels increased following the closure of Bujagali dam. The five islands' vegetation was dominated by *Tremor orientalis* and *Ficus* tree species and locals claim they also used to be inhabited by a small human population that practiced subsistence agriculture. By the time of this study however, the last remnant of the islands, Njaaba was dominated by *Vossia cuspidate* vegetation with a number of submerged tree stumps and logs reflecting the terrestrial vegetation that used to exist before the dam. Agriculture took place on both the eastern and western banks Kalange transect though the former seemed more cultivated than the latter. Apart from the agriculture land in the background, the immediate shores of the East bank were dominated by *Vossia cuspidata* which changed to forest/shrub vegetation type as you moved further into land. The western bank was however covered by only trees forming a forest. Both the eastern and western banks were gently sloping though the latter was steeper. Three sampling points were used as sampling positions in this transect, that is; the submerged (now *Vossia cuspidata*) Island and two sites on the western and east banks, adjacent to the *Vossia cuspidata* island. The western bank position was about 100m from Njaaba while the eastern one was about twice as Njaaba from the western bank.

3.2.2 Reservoir

Following the closing of the dam, a 4.5km^2 reservoir was created immediately upstream of the dam. Because the physical conditions in the reservoir are different from areas further upstream, NaFIRRI decided to consider it as an independent transect (NaFIRRI, 2013). This transect is located about one kilometer upstream of the dam at GPS coordinates 36N 0514499E. It was characterized almost by standing waters which velocity was also dependent on the dam operation downstream. Its eastern bank were mainly flat land covered with shrub-forests which covered up to 70% of the stretch, with the rest being agriculture cultivated land. The immediate shores of the eastern bank were dominated by *Vossia cuspidata*, while those of the western bank were rocky against a completely shrub vegetated background. The plant composition of the margins of the western bank was dominated by one single species which we could not identify. Three locations were being used as sampling points in the Reservoir that is; on the western bank, eastern bank and in the seemingly calm open waters of the reservoir between the two bank points.

3.2.3 Buyala

This transect was located 36N 0514575E, about one kilometer downstream of the dam. Both of its banks were steep with a mountainous background. The western bank vegetation was about 40% covered with sedges at the immediate shores but further away from the water, the vegetation changed to forests and farmed land. About half of the eastern bank of this transect was used for agriculture although just a few meters from the *Vossia cuspidata* dominated shore margins were trees that formed a significant part of bank's vegetation. Three points were also sampled in this transect. The eastern sampling location ended in rocky rapids downstream, while the western point lied against a forested background with rocks at the immediate water margins. The current in this transect was faster than that in the Reservoir and Kalange

3.2.4 Kirindi

Although this transect had a number of islands which disrupted the river's flow, it is where the fastest currents forming a series of rapids were observed. The transect was located about

24km downstream of Bujagali dam at GPS coordinates 36N 0506200. The vegetation at the immediate shores of the eastern bank was *Vossia cuspidata* against a background dominated by agriculture land. In between the *Vossia cuspidata* and cultivated land were shrubs mixed with patches of *Cyperus papyrus*. By the time of this study, sand mining was taking place in some areas of the river and it was being landed at a beach on the western bank. One sampling point was located on the eastern banks of Damba Island, where the marginal vegetation at the Island land –water interface was *Vossia cuspidata* against a shrub dominated background. About 200metres further downstream lied another sampling point on the eastern bank of the river. This one was dominated by *Cyperus papyrus* with shrub forests just behind them and its background was all cultivated land where maize was the major crop. The third sampling point was on the western bank almost adjacent to that on the eastern bank. The main vegetation at the immediate water margins was *Vossia cuspidata which* changed to shrubs further offshore.

3.3 Sampling and data sources

For consistency and compatibility, we replicated in this study the sampling procedures as used by the National Fisheries Resources Research Institute (NaFIRRI) (2006-2015) to obtain data on the contemporary status of fish populations in the study area. Sampling in each transect was undertaken at three sampling points in each transect as described above (Fig. 1). Three fleets of multifilament gillnets composed of panels of mesh sizes 1"-8" in which pieces of 1"- 6" increased at 0.5" while those of 6"-8" at 1" were set overnight for two consecutive days each month in each of the sampling points in each transect. Sampling was undertaken between December 2015 and February 2016. At each sampling site, the fleets were set parallel to the water flow to limit on gillnet losses. The fleets were set in the evening between 1800 hrs and 1900 hrs and retrieved the next morning between 0600 hrs and 0700 hrs.

To enhance data availability, seining was employed once in each of the last two months of sampling. A 30 m long 8 mm mesh-size net held at the beach at one end as the other was spread out to the maximum possible length to allow hauling without losing the fish within its enclosure was employed. On the basis of their suitability for seining, sites for seining purpose were identified in each of the transects. At the upstream transects Kalange and the Reservoir, two inshore sites; one on the eastern and another on the western bank were identified as suitable for seining. At Kirindi two sites on either banks were also identified used as seining points during the study period. Seining was undertaken between 0600 hrs and 0700 hrs on only one of the two consecutive days in which sampling was undertaken in each transect every month. The seine net was hauled twice at each seining site in a transect. However at Buyala, no suitable seining point could be identified.

Monitoring data for the period 2007-2015 and ecological baseline survey data of 2000 and 2006 were obtained from the NaFIRRI archives and used as part of the complete time series data for spatial and temporal analysis of the fish population assemblages' attributes of interest.

3.4 Sample analysis

Fish specimens recovered from each transect per sampling were identified to the lowest level possible, separated, counted and their numbers recorded by species as described in Greenwood (1966). In addition, the total lengths (Fork length for species with forked caudal fins) of each species recovered from experimental fishing and seining were measured using a measuring board and recorded to the nearest one decimal place in centimetres. Total and individual weights of all specimens from each recovered species were also taken and recorded in grams. Weigh measurements were taken using either spring balances or digital weighing scale (calibrated to read to the nearest three decimal points) depending on the size of the specimen (s).

3.5 Data analysis

All data was entered in MS Excel and arranged by stations and years as the primary classification variables. Species data was entered as count and weight. Files for use with all software were saved in CSV format to facilitate portability.

3.5.1 Contemporary relative abundance and biomass, diversity and size structure

Relative abundance and biomass

The biomass and abundance indices of each species were calculated as catch per net per night. For this reason, only data obtained from gillnetting was used in the determination of the relative abundance and biomass of each species. The same measure is by NaFIRRI in the ongoing monitoring studies to assess changes in the fish populations of the dammed river section. The biomass and abundance count data for each species recovered from an individual transect were treated as independent samples. The total weight and numbers of each species recovered from a given transect in the three months sampling period were calculated using MS excel. The gillnets were divided into three size groups; Small (1"-2.5"), Medium (3"-4.5") and Large (5"-8") according to FIRRI (2000) and NaFIRRI (2006). The relative abundance and biomass for each species were then calculated depending on its vulnerability to a size category. Using the species list generated through the monitoring studies, each species was assigned a gear category with the corresponding total number of nets the species could be vulnerable to per sampling day (Table 1). Since sampling at each transect was undertaken for a total of six days in the whole sampling period, the relative abundance and biomass of each species per transect were calculated as total number or biomass of each species obtained in the three month's sampling period divided by the sum of gillnets it is vulnerable (Table I) and divided by the total number of days (6) sampled at each transect.

Table 1 Fish species gear size allocation and the number of gillnet pieces used in the calculation of their relative abundance and biomass. The list was generated from results of studies in the same area by FIRRI (2000) and NAFIRRI (2006).

Species	Gillnet size category	Number of gillnets
Bagrus docmak	Large	39
Barbus altianalis	Large	39
Barbus paludinosus	Small	12
Brycinus jacksonii	Small	12
Clarius gariepinus	Large	39
Haplochromines	Small	12
Labeo victorianus	Large	39
Lates niloticus	Large	39
Mormyrus kannume	Large	39
Oreochromis leucostictus	Medium	24
Oreochromis niloticus	Large	39
Oreochromis variabilis	Large	39
Synodontis afrofischeri	Medium	24
Synodontis victoriae	Medium	24
Tilapia zilli	Medium	24

Size structure

For determination of the size structure of the dominant fish species recovered in this study, all total length data for each species recovered from each transect was treated as an independent sample and tested for normality in SPSS using the Shapiro-Wilk test at the 95% confidential interval. One way ANOVA was used to test for significant of differences among the sites. If the data was normally distributed. Data set which violated normality assumptions log transformed and re-tested for normality. If data sets from all the sites still failed the normality test, then the non-parametric Mann-Whitney test was used to test for significance among the sites. However if one or more of the sets passed or failed the normality test, then both the parametric and non-parametric tests were applied to test for significance between them.

Diversity

As a measure of difference in distribution patterns upstream and downstream, Shannon-Wiener diversity indices for each transect were calculated as in Agrawal and Gopal (2013). The relative abundance of the species for each transect in this study was used in the calculation of the index. Since the sample size was small, non-parametric Kruskal-Wallis test was used to test for significance differences among the sites.

3.5.2 Spatial-temporal trends

Abundance and biomass

To examine spatial-temporal variations in abundance and biomass, we merged the contemporary data obtained in this study with that obtained between 2006 and 2015. As consistent data between 2006 and 2016 was not available for all the sites sampled in this study, spatial-temporal analysis for all the parameters was centred at only furthest upstream Kalange and immediate downstream Buyala transects. Although sampling was undertaken only once each year between 2006 and 2008, twice between 2009 and 2015 and thrice in the current study, possible effects due these difference where neutralized by adjusting the days sampled per transect in each year to account for the extra effort. For that matter, two, four and six days were considered in the calculation of abundance and biomass for each species in the period 2006-2008, 2009-2015 and this study respectively. Data sets for each transect was treated in a similar way as for the contemporary data sets. The overall abundance and biomass for each year were then obtained by calculating the total sum of relative abundances and biomass of all species. The Kruskal-Wallis test was used to test for significance of differences among the years.

Spatial-temporal changes in the two parameters were further examined by developing ecological similarity matrices for each transect using the Bray-Curtis similarity coefficient. Bray-Curtis similarity coefficients were calculated in PRIMER6 software as follows

$$D = 1 - 2\frac{\sum_{i=1}^{S}\min(a_i, c_i)}{\sum_{i=1}^{S}(a_i + c_i)} \qquad \text{...Equation 1}$$

Where D= Bray-Curtis coefficient, a = relative abundance of i^{th} species at site a, relative abundance of i^{th} species at site c and S= total number of species.

Data used in the calculation of the coefficients was log transformed in the form LogX+1. Group average hierarchical sorting strategy of cluster analysing and non-metric multidimensional scaling (nMDS) were then used to visualise the differences and similarities among the years (Clarke, 1993). ANalysis Of SIMilarity (ANOSIM) was used to test for significant differences among the observations and SIMilarity PERcentage (SIMPER) analysis applied to determine the species most responsible for the similarities/dissimilarities among the years and sites.

Diversity

Spatial and temporal trends in diversity were determined by computing the Shannon-Wiener diversity indices for each year and transect using Excel. The indices were calculated as follows from the numbers specimen obtained from eat species in a year and transect according to Agrawal and Gopal (2013).

$$H' = -\sum_{i=1}^{R} p_i \ln p_i \quad \text{..} \textit{Equation 2}$$

Where H' = Shannon's diversity index, R = Total number of species, i = the i^{th} species, In = natural logarithm and $pi = \frac{ni}{N}$ in which N = Total number of individuals of all species and n_i = number of individuals of the ith species

Size structure

For annual size structure variation, all total length size data for each species recovered from all transects in a given year was pooled together and treated as an independent sample for the year in question. To determine annual variation in size structure of a species at a given transect, each year's size data for each species at a transect was also taken as an independent sample. ANOVA and/or Kruskal-Wallis tests were used to determine significance of differences among the years. The data sets underwent the same treatment as for the contemporary date before ANOVA or Kruskal-Wallis test could be done on them. Since size structure data for the other species were not consistently available over the years, temporal changes in size structure were only determined for the species defined by NaFIRRI (2006) as most dominant for which at least more than 30 specimens had been recovered in each year of sampling.

3.5.3 Spatial-temporal trends in dam impact on fish populations

For determining of spatial and temporal degree of dam impact on fish populations, the count biomass and abundance data were employed in the construction of the Abundance Biomass Comparison curves (ABC) (Warwick, 1986) in PRIMER6 software. Both data forms were also first square-rooted to reduce the effects of high abundances on the overall output. For similar reasons as for temporal size structure and diversity indices, temporal and spatial variation in the dam's degree of impact were only determined at the most upstream Kalange and the immediate downstream Buyala transects. Fish species were first ranked according to their importance based on relative abundance and biomass on a scale of 1-10 with increasing order of importance. The ranks for each species were then plotted against the species' respective percentage cumulative dominance as biomass or numbers on a similar scale according to Yemane *et al.* (2005). The area between the two curves (relative biomass and relative abundance) for each year were as;

$$\text{W-Statistics} = \frac{\sum_{i=1}^{s}(B_i - A_i)}{50(S-1)} \qquad \text{.......... } \textit{Equation 3}$$

Where B_i is the relative biomass of the i^{th} species, A_i is the relative abundance of the i^{th} species and S = number of species.

The value of the W-statistic values obtained for each year indicated the stress levels the population was facing, which is also an indicator of the impact they were facing from the different pressures. The stress levels recorded in 2006 were then subtracted from each year starting from 2007 to 2016. The differences between 2006 and each year were then converted to absolute percentages and taken as a measure of the degree of dam impact on the fish population in each respective year. This data was the one used in the analysis of the spatio-temporal variation in the dam's disturbance of fish populations in the river.

CHAPTER FOUR
RESULTS

4.1 Contemporary fish species abundance, diversity and size structure

A total of 1,377 fish specimens of fresh weight 72.7 kg were recovered from all transects during the sampling period by both gillnetting and seining. The specimens were representative of 40 species belonging to eight families. Haplochromine cichlids comprised 24 species and contributed 67.3 % and 18.7% of the abundance and biomass respectively recovered through both seining and gillnetting. However, since some haplochromine specimen were not clearly identified, all species falling under this group are reported as a single taxon, Haplochromines. A total of 543 fish specimens of fresh weight 19.2kg were recovered from the most upstream Kalange transect, making it the most productive during the study period. The Reservoir was the least productive, registering about 12% of the overall abundance recovered from both seining and gillnetting in this study. Buyala recorded the least biomass most likely because only gillnetting was undertaken in this transect. Gillnet catches constituted 69.7% and 90.3% by number and weight respectively (Table 2 and 3). Approximately 52% and 61% of the overall fish abundance and biomass respectively recovered in this study were from the upstream transects.

With haplochromines excluded, the most dominant species by numbers with decreasing order of importance were; *L. niloticus, S. afrofischeri* and *M. kannume*. The three species together contributed about 15% to the overall abundance (Table 5). *M. kannume* recorded the highest relative biomass of 118g/net/night, but together with the haplochromines, the two contributed over 60% of the total biomass from all species. *L. niloticus, Bagrus docmak, S. afrofischeri* and *S. victoriae* together contributed about 35% to the overall biomass, making the contributions by other species recovered during the study period insignificant (Table 5). No significant differences in either abundance or biomass of any species existed among the sites (Kruskal-Wallis P=0.392 at 95%CI).

The largest specimens of the numerically most dominant species *L. niloticus* and *M. kannume* were from the downstream transects Kirindi and Buyala respectively (Table 4). There were significant differences in the mean total lengths of *L. niloticus* recovered at the Reservoir and those from Kalange and Kirindi (Mann-Whitney test p <0.05).

The total mean length of *M. kannume* recovered at Kalange were also significantly different from those recovered at the Reservoir and Buyala (Mann-Whitney $p<0.05$). The highest mean total length of *S. afrofischeri* was recorded at Kalange and no significant differences in the mean size of the specimens was observed among the transects (Kruskal-Wallis $P=0.392$ at 95%CI). The mean total length of *O. niloticus* from Kalange were significantly higher than those recovered from Kirindi (Mann-Whitney test $P<0.05$).

Comparison of the relative biomass before and after inception of the dam construction in 2006 showed that overall biomass at Kalange, Buyala and Kirindi transects all decreased by over 70% (Table 5). The independent sample t-test showed significant differences in the catch rates of *B. altianalis*, *L. niloticus* and *M. kannume* ($p<0.05$). In terms of individual transects, the highest decline in relative biomass was over 97% at Buyala. The highest decline in catch rates by an individual species was by *B. altianalis* which reduced from 97.6g per net per night from all transects in 2006 to 0.8g in 2016 (Table 5). To eliminate the effects of gillnet losses at Kirindi, gillnet-loss corrected biomass were calculated for this transect. However, the results also indicated the biomass in 2016 was significantly lower than that in 2006 (Table 5).

Species diversity was highest at the Reservoir (1.7) and lowest at Buyala (0.6) and no significant differences were observed among the diversities of the four transects.

4.2 Spatial-temporal changes

4.2.1 Species abundance and biomass

Abundance at the upstream Kalange transect was dominated by haplochromines until the commissioning of dam operation in 2012 when it declined by over 40% as *S. afrofischeri* increased by more than 70% (Fig 2 i and ii). Although there was a remarkable gradual increase in haplochromine abundance after 2012, it appeared the dam operation provided better conditions for the performance of the large bodied species *M.kannume*, *L. niloticus* and *S. afrofischeri* upstream. This shift is also depicted by a similar change in biomass dominance, from the haplochromines to the three species after 2012 (Fig 2 i and ii).

At the downstream Buyala transect, an assemblage dominated by large bodied species *L. niloticus*, *M. kannume* and *S. afrofischeri* between 2006 and 2012 shifted to one of small bodied haplochromines from 2013 onwards (Fig 2 iii and iv). This suggests different conditions in the upstream and downstream transect after commissioning.

When abundance for all species recovered in each year were aggregated, there was no observed significant change throughout the years save for the sharp increase and equal decline in 2009 and 2010 respectively at Kalange. In a similar way, abundance at Buyala transect showed no significant changes over the years but underwent alternating increases and declines (Fig 3A). However an over 80% decline in biomass was observed between 2006 and 2016 at both transects (Fig 3B). Nevertheless, the decline at Buyala transect was generally greater than that at upstream Kalange. No significant differences in abundance and biomass were observed among the years (Kruskal-Wallis P=0.440).

There were no significant difference in the abundance of fish among the years and sites (PERMANOVA P=0.412 and 0.552; ANOSIM P>0.05, R= 0.202 and 0.045). However differences were detected in the temporal variation of biomass among the years and the two transects considered in the examination of temporal changes (PERMANOVA P= 0.031and 0.037; ANISOM P=0.01; R=0.33).

At 60% similarity, annual biomass at Buyala transect could be clustered into two major year groups in which 2013 belonged to none (Fig 4A). The dissimilarities between 2013 and the 2006/2014 and 2007-2016 year groups was about 50% and 43% (Table 6). However in terms of abundance, the years could only be clustered into groups at 50% similarity (Fig 4B). this implied species abundance at this transect was more sensitive to environmental change than biomass. The year 2006 did not belong to any of the groups suggesting it was the most dissimilar. The 2007-2012 and 2013-2016 groups were each about 52% dissimilar from 2006 (Table 6).

At upstream Kalange transect, ecological dissimilarities among the years based on biomass occurred above 70% similarity. This reflected less disturbance in the upstream areas as compared to the downstream. Nevertheless a similar grouping as in the downstream transect were also observed. However unlike Buyala where years 2006 and 2013 did not appear in any group, each of the two years appeared in independent groups at Kalange (Fig 4C). In terms of abundance, the years were grouped into two clusters at 60% similarity with only 2013 and 2014 appearing in a different group (Fig 4D).

Therefore basing on the nMDS results, fish populations in the study area could be categorized into three phase; the baseline populations of 2006, the construction phase populations of 2007-2012 and the dam closing and operation populations between 2013 and 2016. The percentage dissimilarity among the three groups suggest dam operation caused more disturbance than its construction (Table 6). Nevertheless it appears that since 2014, the ecosystem has been stabilizing towards the construction phase but not the baseline conditions (Table 6).

Table 2Total number and weight of fish species recovered through gillnetting from each transect between December 2015 and February 2016

Family	Species	Total number				Tot	Total fresh weight(g)				Tot.
		Upstream		Downstream			Upstream		Downstream		
		Kalange	Reservoir	Buyala	Kirindi		Kalange	Reservoir	Buyala	Kirindi	
Bagridae	*Bagrus docmak*	7	4		6	**17**	1450	696		6632	**8778**
Cyprinidae	*Barbus altianalis*				2	**2**				1248	**1248**
Alestidae	*Brycinus sadleri*		2			**2**		18			**18**
Mormyridae	*Gnathonemus longibarbis*	2	1			**3**	88	182			**270**
Cichlidae	*Haplochromines*	245	41	300	18	**604**	3561	438	3196	448	**7643**
Mormyridae	*Marcusenius grahami*				1	**1**				36	**36**
Centropomidae	*Lates niloticus*	55	44	22	7	**128**	5639	5065	1522	1922	**14147**
Mormyridae	*Mormyrus kannume*	19	33	8	27	**87**	2164	16244	2738	6405	**27551**
Cichlidae	*Oreochromis niloticus*	5	1	1		**7**	588	668	6		**1262**
Cichlidae	*Oreochromis variabilis*	4				**4**	310				**310**
Synodontidae	*Synodontis afrofischeri*	39	11	26	1	**77**	1488	382	590	38	**2498**
Synodontidae	*Synodontis victoriae*	4	13	1	1	**19**	296	1196	116	96	**1704**
Cichlidae	*Tilapia zillii*	5		4		**9**	64		144		**208**
	Total	**385**	**150**	**362**	**63**	**960**	**15647**	**24889**	**8312**	**16825**	**65673**

Table 3: Total number and mean fresh weight biomass (g) of fish species recovered through seining between December 2015 and February 2016.

		Total number				Mean fresh weight (g)			
		Upstream		Downstream		Upstream		Downstream	
Family	**Species**	Kalange	Reservoir	Kirindi	**Total**	Kalange	Reservoir	Kirindi	**Total**
Cyprinodontidae	*Aplocheilichthys pumilus*		1		**1**		2		**2**
Cichlidae	*Haplochromines*	130	11	182	**323**	3246	84	2650	**5980**
Centropomidae	*L. niloticus*			1	**1**			70	**70**
Cichlidae	*O. leucostictus*			1	**1**			60	**60**
Cichlidae	*O. niloticus*			17	**17**			32	**32**
Cichlidae	*O. variabilis*	9	3	22	**34**	112	70	206	**388**
Cichlidae	*T. zillii*	19		21	**40**	238		300	**538**
	Total	**158**	**15**	**244**	**417**	**3596**	**156**	**3318**	**7070**

Table 4 Mean total lengths (cm) ±SE of the most dominant species recovered by both seining and gillnetting between December 2015 and February 2016 in the study area. The sample size (n) are showed parentheses. Mean total lengths were only calculated for species with n ≥ 5

Transect	***B. docmak***	***L. niloticus***	***M. kannume***	***S. afrofischeri***	***O. niloticus***
Kalange	22.8±2.8 (7)	20.2±1.1 (55)	22.2±1.3 (19)	13.3±1.3 (39)	16.8±3.6 (5)
Reservoir		14.7±1.3 (44)	31.5±2.2 (33)	11.5±0.7 (11)	
Buyala		17.9±0.8 (22)	31.8±2.8 (8)	12.1±0.2 (26)	
Kirindi	29.5±0.5 (6)	23.4±7.1 (8)	25.4±2.9 (27)		8.4±0.5 (17)

Table 5 Comparison of 2006 and 2016 relative abundance as catch per net per night (g) for some of the dominant species . The catch rates for 2016 are averages of the actual rates for the three consecutive month's sampling undertaken for the year. In parentheses are the number, n (with asterisks) from which the rates were computed and the gillnet-loss corrected catch rates for Kirindi transect.

Species	**Transect**							
	Kalange		**Buyala**		**Kirindi**		**Total**	
	2006	**2016**	**2006**	**2016**	**2006**	**2016**	**2006**	**2016**
B. altianalis	25.2(2*)	0.0	63.9(10*)	0.0	8.5(1*)	1.77(5.3, 1*)	**97.6**	**1.8**
B. docmak	0.0	2.1(2*)	0.0	0.0	11.2(4*)	9.4(27.5, 2*)	**11.2**	**11.5**
Haplochromines	303.9(216*)	31.5(82*)	284.9(404*)	14.8(100*)	7.7(15*)	14.3(6.8, 6*)	**596.5**	**60.6**
L. niloticus	20.6(19*)	8.0(18*)	141.5(114*)	2.2(7*)	12.4(7*)	2.8(10.3, 2*)	**174.5**	**13.0**
M. kannume	144.1(33*)	3.1(6*)	284.3(69*)	3.9(3*)	124.1(63*)	9.1(35.6, 9*)	**552.5**	**16.1**
Total	**493.8**	**44.7**	**774.6**	**20.9**	**163.9**	**37.5 (85.5)**		

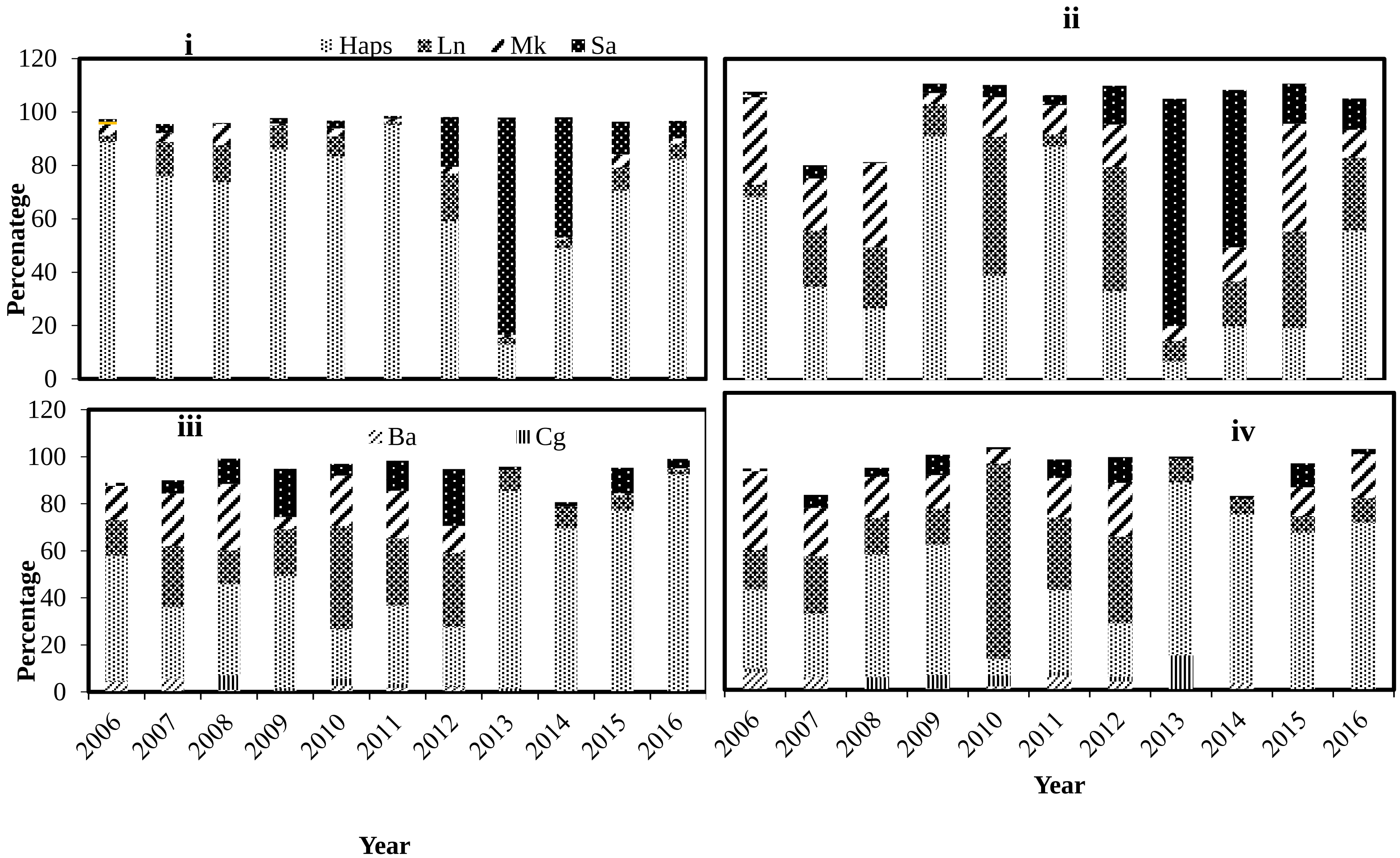


Fig 2 Variations in the percentage relative abundance (i & iii) and biomass (ii & iv) of the most dominant fish species recovered from upstream Kalange (i & ii) and downstream Buyala (iii & iv) transects over the period 2006-2016. Haps = Haplochromines; Sa= *S. afrofischeri*; Ln = *L. niloticus*; Ba= *B. altianalis*; Mk = *M. kannume* and Cg = *Clarias gariepinus*.

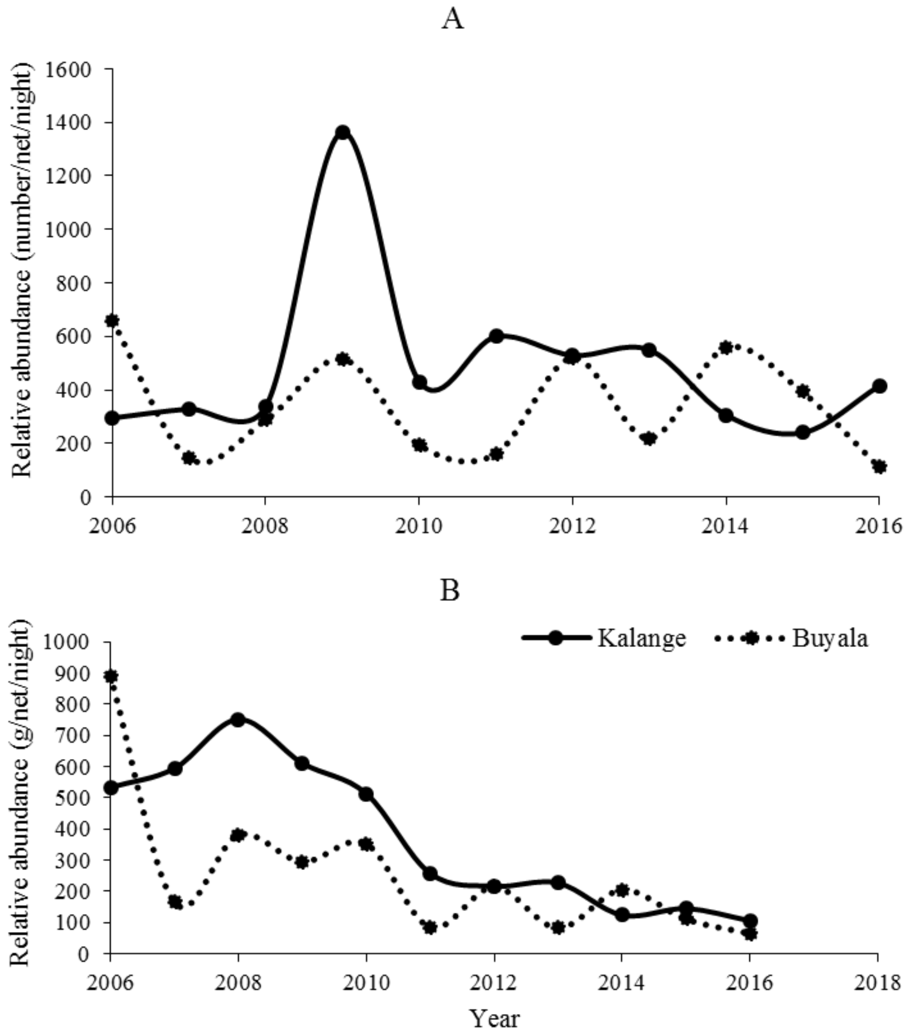


Figure 3 Annual variation in the overall (all species aggregated) relative abundance by number (A) and biomass (B) between 2006 and 2016 at upstream Kalange and downstream Buyala transects in the study area

A total of 11 major species were revealed to account for about 90% of the observed differences among the year groups (Table 6). In the upstream areas, the biomass of four species *B. altianalis*, *M. kannume*, haplochromines and *S. afrofischeri* were the most affected by the dam construction and closing. In terms of abundance, the most affected species upstream were haplochromines, *S. afrofischeri* and *L. niloticus*. In contrast, the same event appears to have affected mainly cichlids *T. zilli*, *O. variabilis* and *O. niloticus* in addition to *B. altianalis* and *M .kannume* in the downstream areas (Table V). With both transect aggregated, the overall biomass and abundance of all but *T. zillii*, *O. variabilis* and *O. niloticus* species generally declined below the 2006 to 2011 average after dam closing in 2012 (Fig 5).

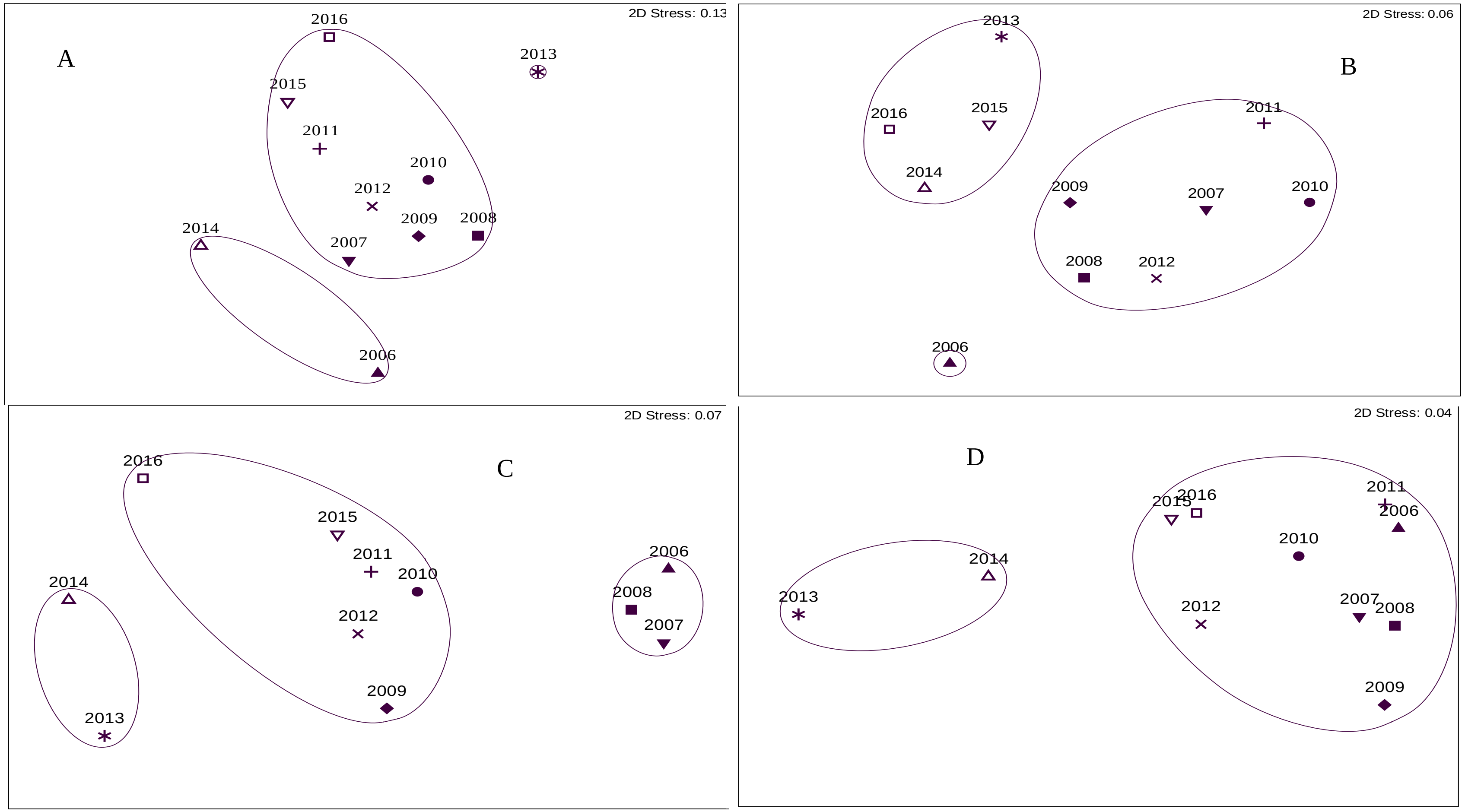


Figure 4 Comparison of non-metric multi-dimensional scaling visualization of the different phases the fish populations in the study area have undergone since the ecological baseline survey of 2006 at downstream Buyala (A and B) and upstream Kalange (C and D) transects. The circling lines are group average hierarchical cluster dendrograms fitted to show similarity at 60% (A and D), 70 %(C) and 50% (B). A and C are biomass visualizations and B and D are visualization of abundance.

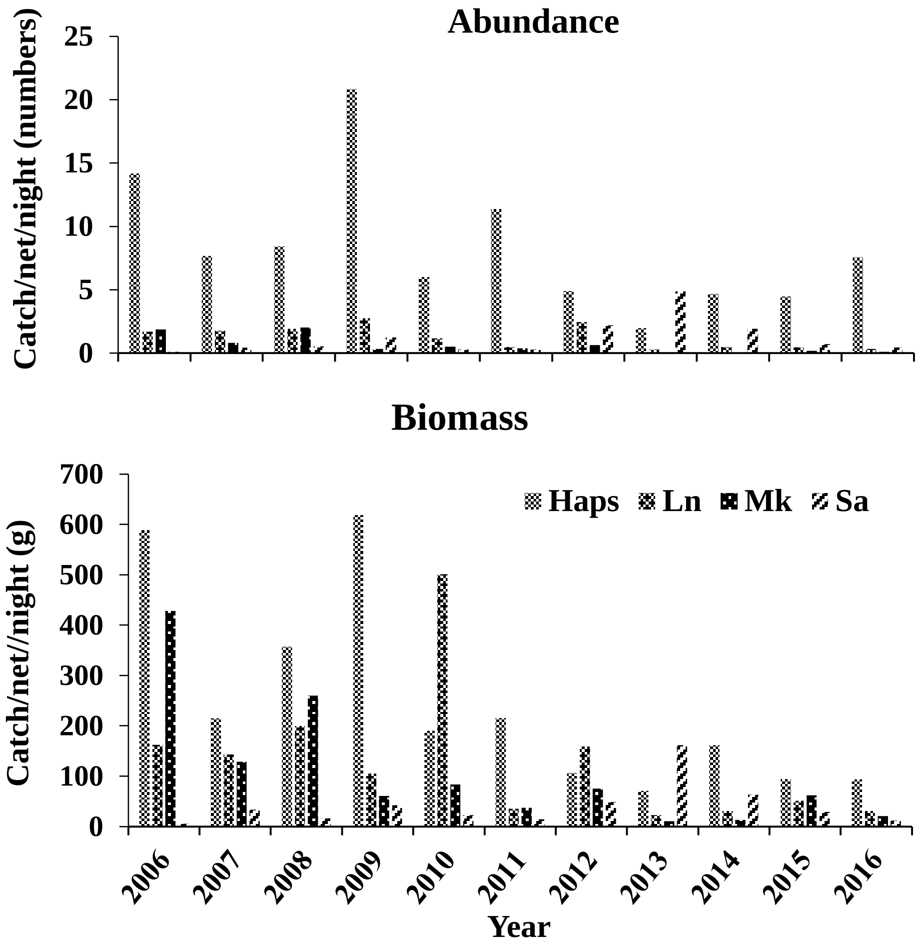


Figure 5 Variation in abundance and biomass of the species accounting for most of the dissimilarities among the different year groups. Haps = Haplochromines, Ln = *L. niloticus,* Mk = *M. kannume* and Sa = *S. afrofischeri*

4.2.2 Species diversity

Without and without haplochromines, the diversity at Buyala transect shows a similar trend over the years (Fig 6A). After a drastic decline in 2007, the diversity underwent a period of concurrent gradual increases and decreases at a two years interval. Peak diversity in this transect was recorded in 2012 and the least in 2016. A general declining trend in the diversity at this transect was observed.

At the upstream Kalange transect, diversity was generally higher than that at Buyala over the period 2006-2016, both with and without haplochromines (Fig 6A and B). However unless Buyala, the temporal patterns of diversity with and without haplochromine were different. With haplochromines considered, the diversity at this transect immediately increased upon the onset of dam construction in 2007 before starting to decline in 2009.

The least values were recorded in 2011 (Fig 6B). In contrast when haplochromines were not considered, 2011 recorded one of the highest diversities in the period 2006-2016. On addition, diversity indices declined between 2006 and 2009 recording least values in 2013 (Fig 6B).

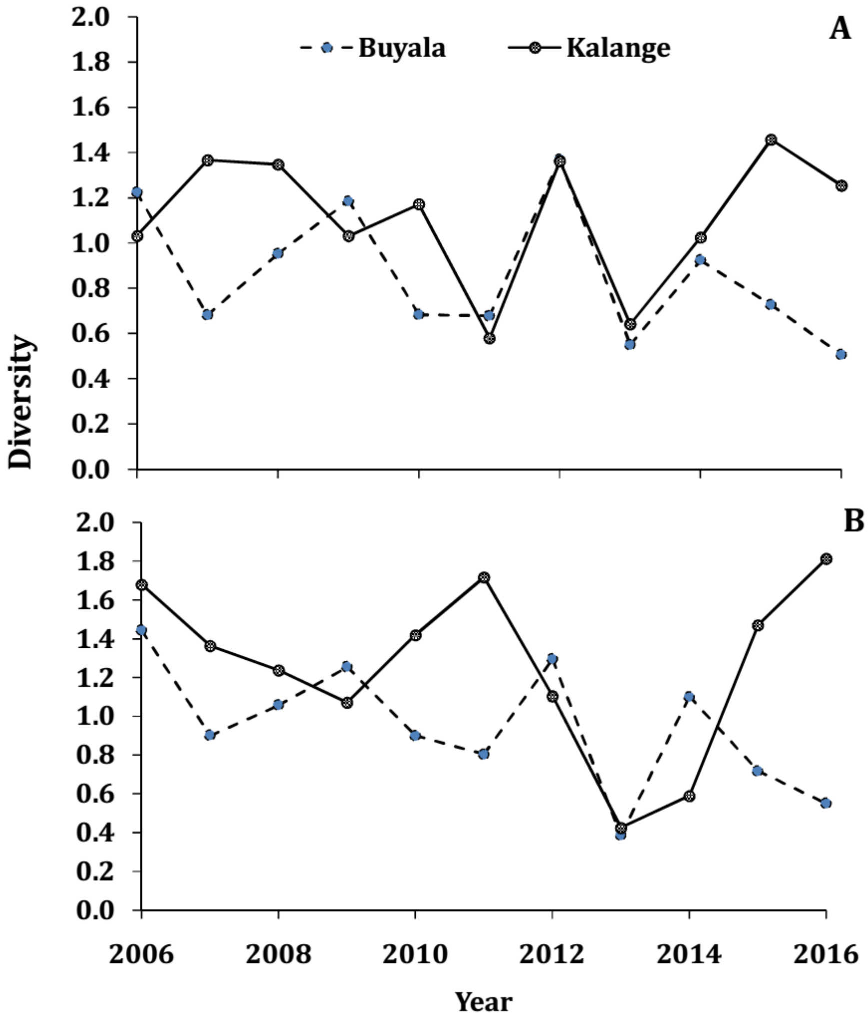


Figure 6 Annual variation in Shannon-Wiener diversity indices for Kalange and Buyala transects between the period 2006 and 2016 (A) with haplochromines included and (B) without haplochromines.

Table 6 Percentage contribution of the different species to the average dissimilarity among the different year groups in the upstream (Kalange) and downstream (Buyala) transects. In parentheses are the average dissimilarities between groups. Ba = *B. altianalis*, Bd = *B. docmak*, Cg = *C. gariepinus*, Haps = haplochromines, Ln = *L. niloticus,* Mk = *M. kannume*, Ol = *O. leucostiscus*, On = *O. niloticus*, Ov = *O. variabilis*, Sa = *S. afrofischeri*, Sv = *S. victoriae* and Tz = *T. zilli*. Groups written with a forward slash (/) contain only the stated years while those with a dash (-) have other years in between them in no particular order (see Fig 3)

	Biomass						Abundance (numbers)			
	Upstream			Downstream			Upstream	Downstream		
Species	2006-2008 vs 2009-2016 (30)	2006-2008 vs 2013/2014 (47)	2009-2016 vs2013/2014 (30)	2006/2014 vs 2007-2016 (42)	2006/2014 vs 2013 (50)	2007-2016 vs 2013 (43)	2006-2012 vs 2013/2014 (53)	2006 vs 2007-2012 (52)	2006 vs 2013-2016 (53)	2007-2012 vs 2013-2016 (51)
Ba	26	20	-	9	14	8	-	8	10	-
Bd	7	-	-	4	5	8	-	-	-	-
Cg	-	5	9	7	8	11	-	-	-	3
Haps	6	11	14	7	6	4	37	33	18	34
Ln	7	-	10	7	7	10	12	11	21	18
Mk	10	11	7	15	12	24	6	14	25	19
Ol	6	-	-	4	4	-	-	-	-	-
On	5	5	7	11	11	-	-	-	-	-
Ov	6	7	11	7	12	7	-	-	-	-
Sa	6	11	15	8	-	12	37	9	4	12
Sv	8	7	-	-	-	4	-	-	-	-
Tz	6	9	13	12	12	5	-	16	14	6

4.2.3 Species size structure

Temporal variation in size structure was only determined for of the three numerically most dominant species *L. niloticus*, *M. kannume* and *S. afrofischeri*.

4.2.3.1 *L. niloticus*

The mean total lengths of this species recorded at the upstream Kalange transect were generally higher than those in the downstream Buyala transect for all the years between 2006 and 2016 (Fig 7). At Kalange, the highest mean total length for the species was 21.2 ±10.8SD recorded in 2015 and the least was 16.8±4.2SD cm recorded in 2009 and 2012 (Fig 7). At Buyala, the highest mean total length was 18±6 cm recorded in 2006 and 2010 and the least was 13.3±4 cm recorded in 2014 (Fig 7). With all size data for the years between 2006 and 2016 lumped together, the mean total length at Kalange 18.5±0.3 cm (n=672) was higher than downstream Buyala 16.3±0.23 cm (n=726). One-way ANOVA revealed highly significant differences among the mean total length of the species recovered from both transects (ANOVA, $p<0.05$).

With both transects aggregated, the highest mean total length of the species was 19±4.5 cm recorded in 2008 and 2016, while the least was 14.3±6 cm in 2014 (Fig 8). Comparison among the years revealed that the mean total length in 2014 as the most significantly lower than most other years ((ANOVA $p<0.05$).

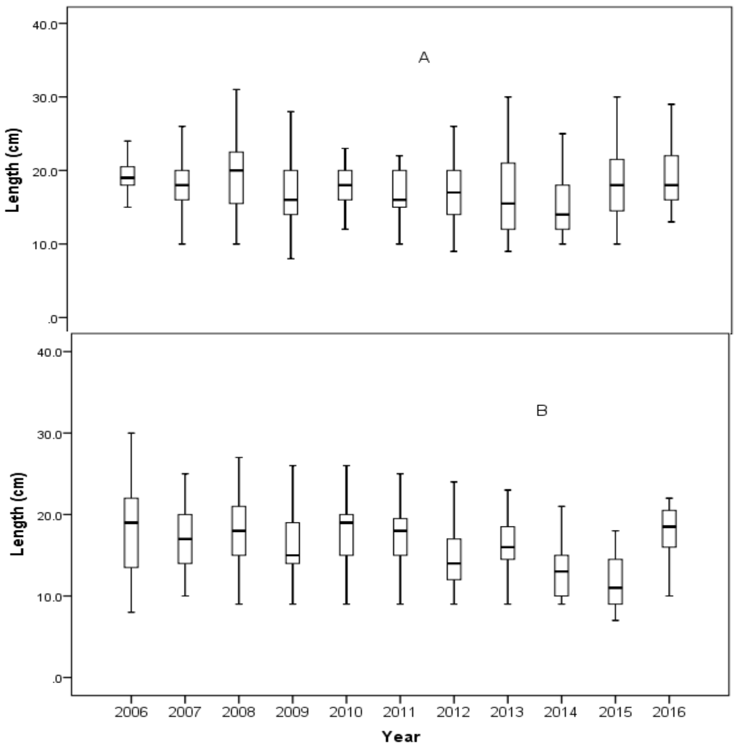


Figure 7 Annual variation in the mean total length of L. niloticus in the upstream (Kalange, A) and downstream (Buyala, B) transects of Bujagali dam in the period 2006-2016. The sample size (n) for each year at Kalange in increasing order of years were; 19, 95, 84, 123, 70, 21, 171, 24, 21, 19 and 25. The n for Buyala in the same order as Kalange were; 104, 45, 52, 110, 54, 55, 192, 20, 51, 27 and 16

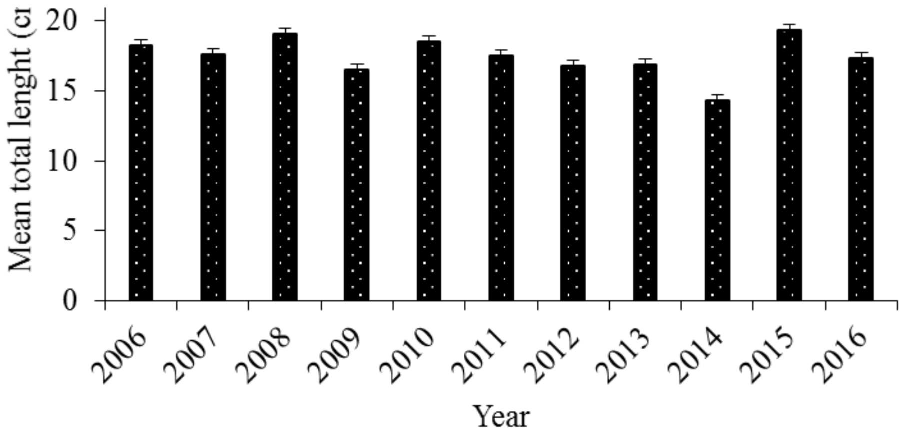


Figure 8 Variation in the annual mean total length of *L. niloticus* recovered between 2006 and 2016 from the study area. Error bars are standard errors of the mean. The sample number (n) for each year from 2006-2016 are as follows with increasing order of the years: 123, 140, 136, 233, 124, 76, 441, 44, 72, 76 and 41 respectively

4.2.3.2 *M. kannume*

Like *L. niloticus*, the mean total length of *M. kannume* recovered from the downstream Buyala transect of 25.6±0.9 cm (n=268) was smaller than that at Kalange. No significant differences in the length of the species were observed between the transects (ANOVA p>0.05). However when each transect was considered on an annual basis independently, the largest specimens at upstream Kalange transect were of mean total length 38±8.3 cm recorded in 2014 and smallest was 22.0±7.2 cm recovered in 2009. In a similar manner, largest specimens of the species recovered in the downstream Buyala transect of 41±25 cm were in 2015 while the smallest were 19.0±4.2 cm recorded in 2011 (Fig 9). There was a general decline in the mean total length of *M. kannume* between 2006 and 2013 at both the downstream and upstream transects indicating unfavorable conditions for the species in this period as compared to two years after closing the dam in 2012 (Fig 9).

With both transects combined, the highest mean total length was 31.9±11 cm recorded in 2014, while the least was 21.2±6.5 cm in 2011 (Fig 10). A decline in the total length means of the species was observed between 2006 and 2011, after which it started increasing before peaking in 2014 (Fig 10). The mean total lengths of most post-closing years were significantly different from other years (ANOVA P<0.05). Most notable is 2014, which mean total length was significantly higher than all years except 2015. This observation contradicts that on *L. niloticus* in which 2014 recorded the least mean length.

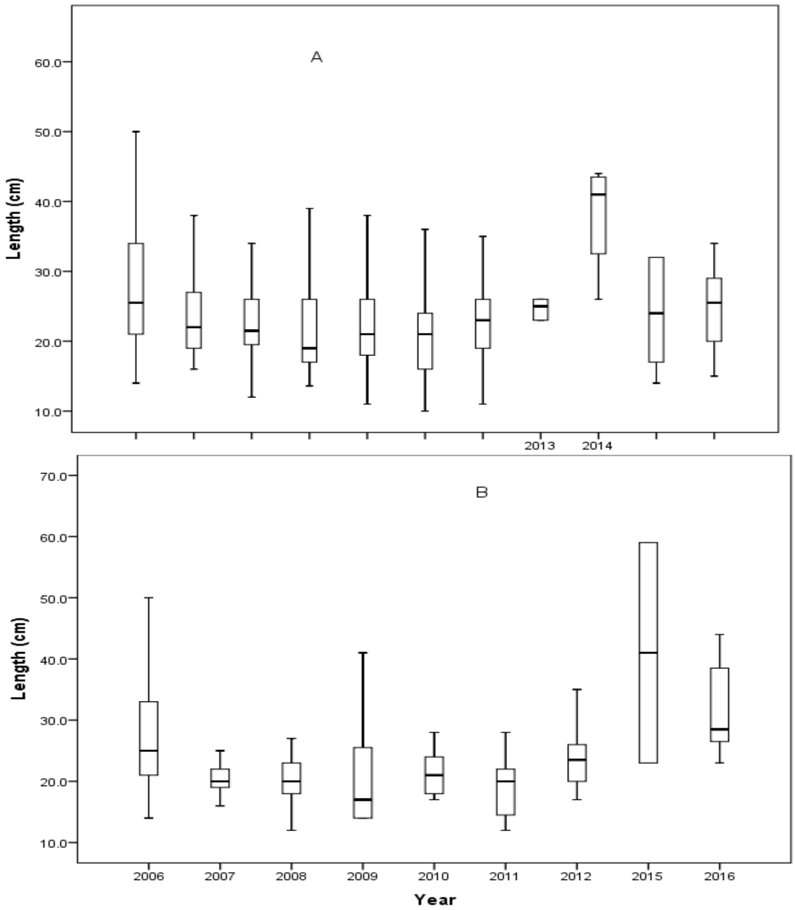


Figure 9 Annual variation in the mean total length of *M. kannume* upstream (Kalange, A) and downstream (Buyala, B) of Bujagali hydropower station. The sample size (n) for each year at Kalange in increasing order of years were; 33, 23, 55, 23, 32, 20, 26, 10, 4, 8 and 18. The n for Buyala in the same order as Kalange were; 69, 24, 45, 19, 30, 27, 44, 2, 8. No specimens were recovered in 2013 and 2014.

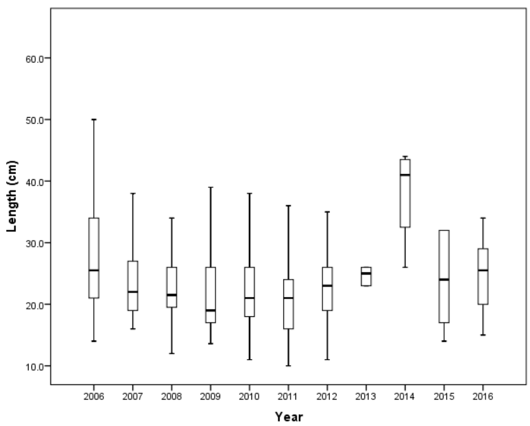


Figure 10 The mean annual total length of *M. kannume* recovered from all transects in the study area in each year between 2006 (06) and 2016 (16). The sample number (n) from which the values were calculated were as follows with increasing order of the years: 102, 47, 100, 42, 62, 47, 121, 48, 42, 77 and 50.

4.2.3.3 *S. afrofischeri*

Between 2006 and 2016, the highest mean total length of the species recovered from upstream Kalange transect was 14.6±0.82 cm (n =15) in 2007 and the least was 10.6±2.3 cm (n=46) in 2014. At the downstream Buyala transect, the largest specimens recorded were 14.5±1.4 cm (n=6) in 2007 while the least were 10.2±1.7 cm (n=56) and 10.3±3.0 cm (n=24) recorded in 2012 and 2008 respectively (Fig 11). Comparison of the overall mean total length between 2006 and 2016 at upstream Kalange and Buyala transects showed the length at the latter transect to be significantly lower than the former (ANOVA $p<0.05$; Kalange (n) = 541 and Buyala (n) =225).

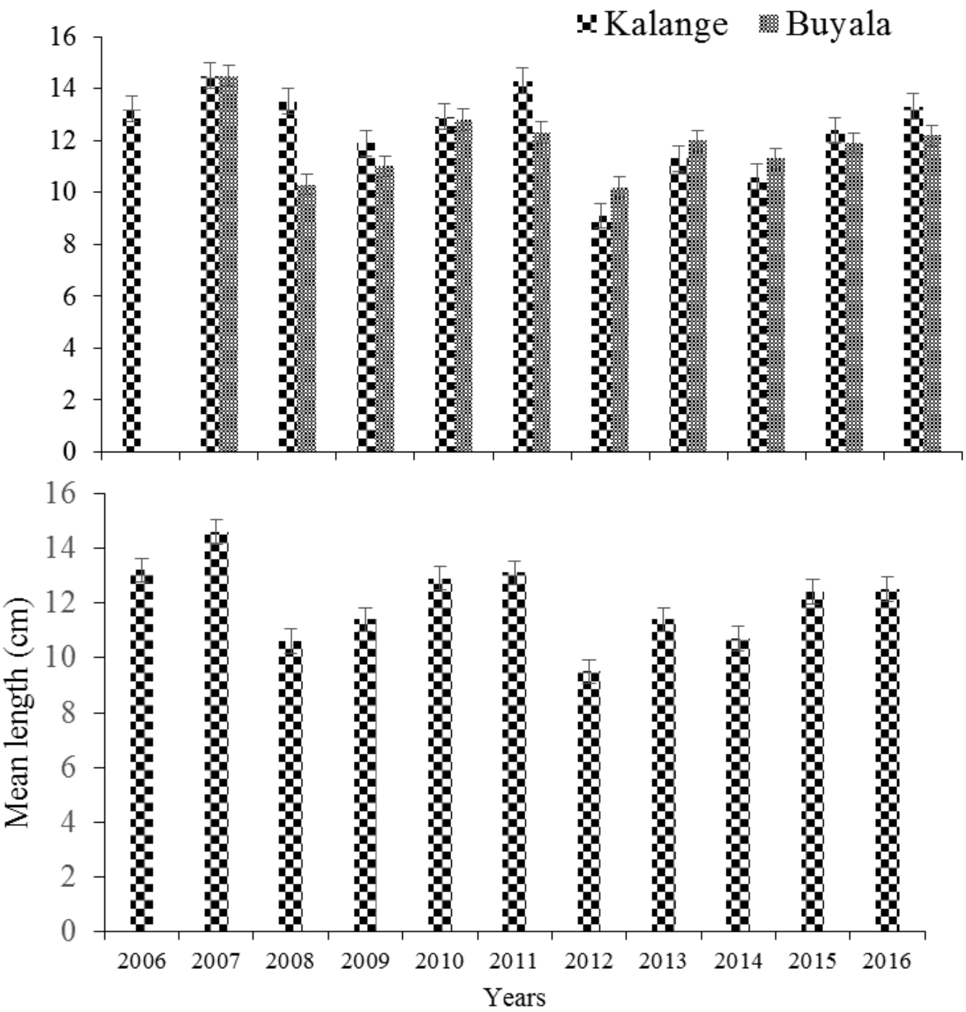


Figure 11 Annual changes in the total mean length of *S. afrofischeri* in the upstream (Kalange) and downstream (Buyala) (above) and trends in the same parameter over the same period with the transects aggregated (below). The errors bars are standard errors of the mean. The sample size (n) for each year at Kalange in increasing order of years were; 5, 15, 2, 46, 19, 11, 111, 222, 46, 40 and 24. The n for Buyala in the same order as Kalange were; 6, 24, 70, 6, 14, 56, 2, 6, 18, 23. No *S. afrofischeri* specimens were recovered at Buyala in 2006.

When spatial differences were not considered, the highest mean total length of the species in the period 2006-2016 was 14.6±1.0 cm (n =21) recorded in 2007, while the smallest was 9.5±2.4 cm (n= 167) in 2012 (Fig 11). Variation in the annual total mean lengths of *S. afrofischeri* reflects the three phases the fish populations have undergone since the inception of dam construction. There was a general increase in the mean size of *S. afrofischeri* between 2007 and 2012 and a sharp decline after 2012 (Fig 11). ANOVA showed total length of 2012 as significantly lower than that of all the other years ($p<0.05$) However, no significant differences in mean lengths of *S. afrofischeri*, *O. niloticus* and *B. docmak* were observed among all sites ($p>0.05$). The Shannon-Wiener diversity index indicated that diversity was highest at the Reservoir (1.7) and lowest at Buyala (0.6).

4.3 Impact analysis of Bujagali dam

The W-statistic from the abundance biomass comparison (ABC) were used as measures of the degree of impact on the fish populations in the river. By comparing the post impoundment years to the baseline of 2006, the difference in the W-statistic value was taken as the 'additional' impact attributed to the dam. Since the fish population in the study area naturally comprises of mainly small bodied individuals and a few exceptionally large bodied ones, values for both parameters (abundance and biomass) were square-rooted before analysis to reduce on the effects of very high abundance or biomass.

With both transects aggregated, most of the annual ABC curves had the abundance curve above the biomass curve, hence negative W-statistic values (Fig 14). This implied stress conditions for the fish populations in the study area. Similarly, most of the W-statistics for upstream Kalange and downstream Buyala transects were also negative. At Buyala transect, a positive value was only recorded in 2007 while at the Kalange positive values were only recorded in 2011 and 2015 (Table 7). The statistics were generally more negative downstream (Buyala) than the upstream (Kalange). This indicated more disturbance downstream of the dam than upstream. Trends in the statistics showed increasing stress since 2006 peaking between 2011 and 2013 (Table 7). The non-parametric Kruskal-Wallis test did not detect any significant difference in the stress among the years and between the sites ($p>0.05$)

A correlation between the W-statistic values and the length, biomass and abundance revealed a significant negative and positive relationship between in the length of *L. niloticus* and the biomass of *S. afrofischeri* ($P<0.05$) (Table 8).

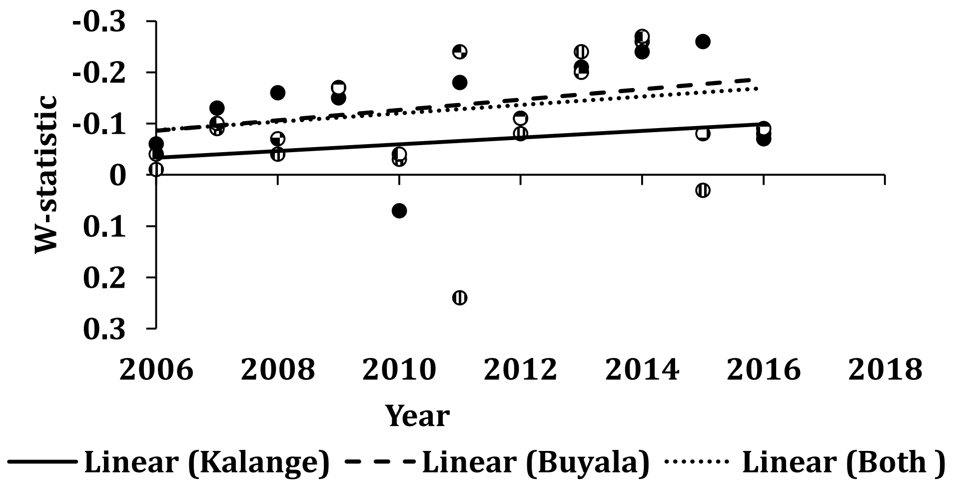


Figure 12 Temporal trends in W-statistic (disturbance/stress) on fish populations in the upstream transect (Kalange), downstream transect (Buyala) and both transects aggregated together (both) between 2006 and 2016. The linear regressions were fitted using the least square method to show the direction of the trends.

However, the biomass of *L. niloticus* and *B. altianalis* and the abundance of *M. kannume* and B. altianalis were also over 40% negatively related to stress (Table 8).

Table 7 Comparison of variation in disturbance/stress (W-statistic) at upstream Kalange and downstream Buyala transects between 2006 and 2016

Year	Kalange	Buyala
2006	-0.01	-0.06
2007	-0.09	-0.13
2008	-0.04	-0.16
2009	-0.17	-0.15
2010	-0.03	0.07
2011	0.24	-0.18
2012	-0.08	-0.11
2013	-0.24	-0.21
2014	-0.26	-0.24
2015	0.03	-0.26
2016	-0.08	-0.07

A regression analysis showed that whereas the abundance (numbers) of *L. niloticus* and *M.kannume* were almost equally negatively affected by the disturbance (Fig 14A), effect of stress on the biomass of the latter was negligible (Fig 13B).

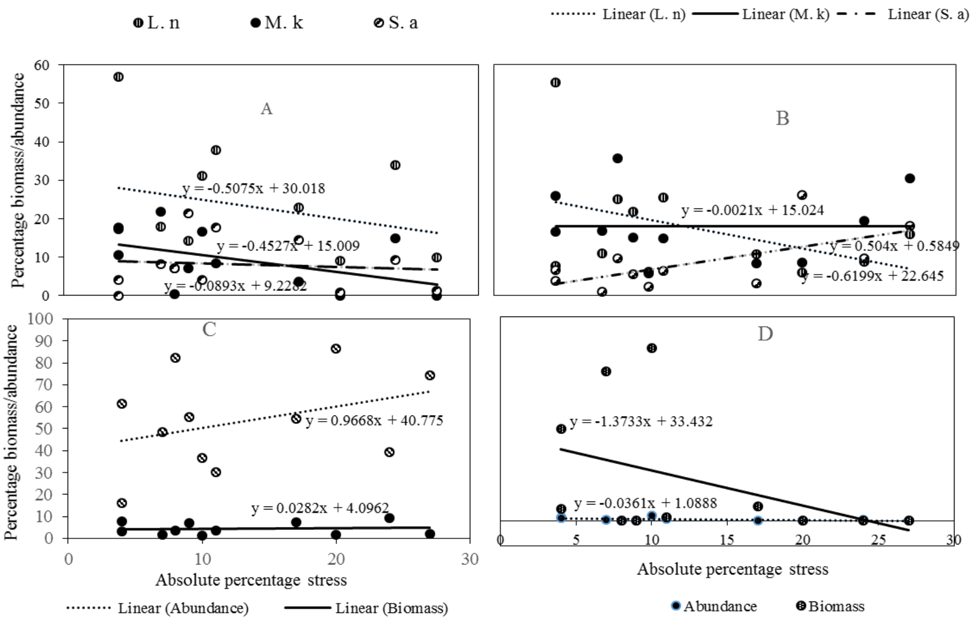
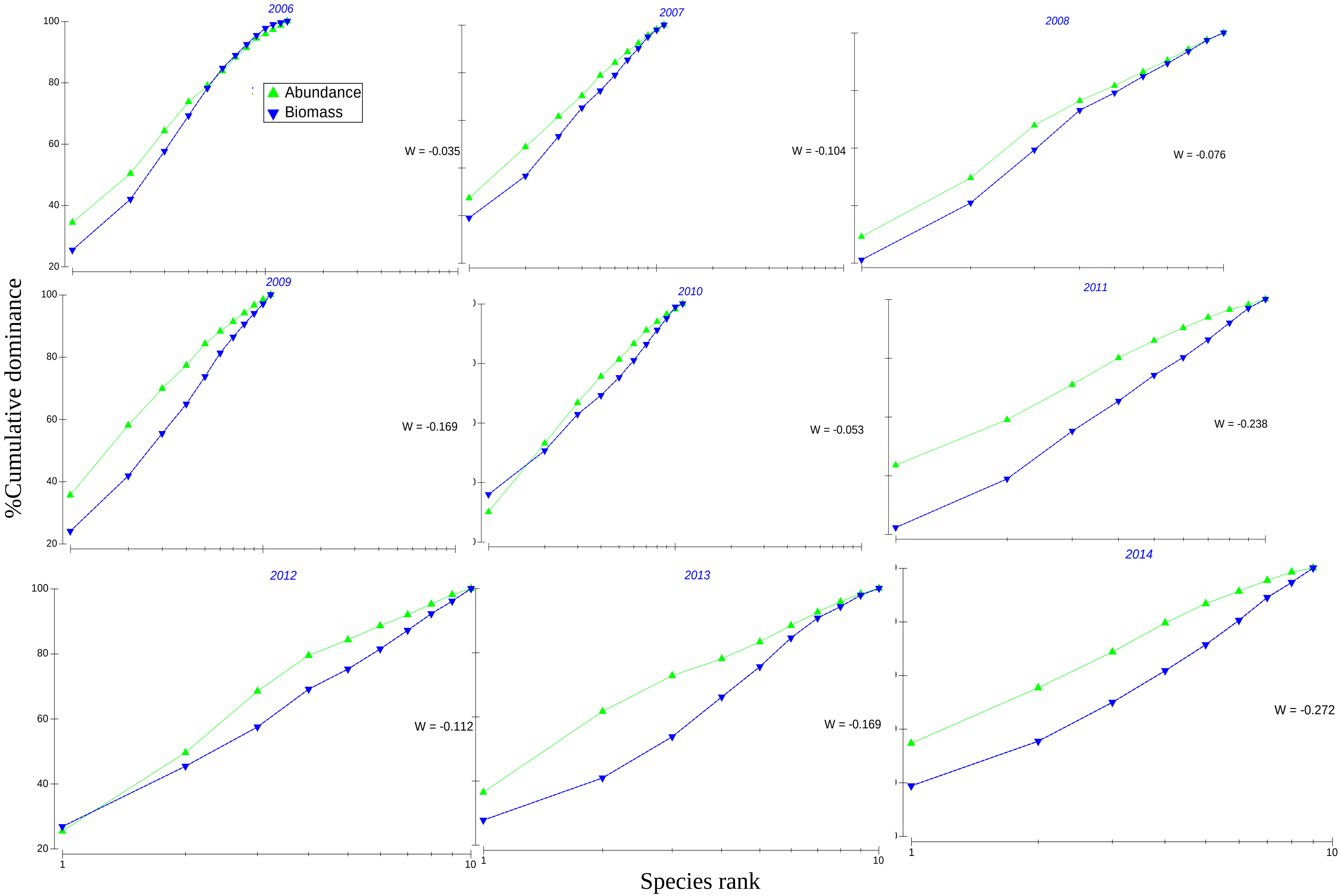


Figure 13 The relationship between stress/disturbance and the percentage relative abundance (A) and percentage relative biomass (B) of the most numerically dominant species *L. niloticus* (L. n), *M. kannume* (M. k) and *S. afrofischeri* (S. a). C and D are a depiction of the same relationship for *B. altianalis* (C) and haplochromines (D). Sample size (n) for all species = 11

The abundance and biomass of *L. niloticus* were both negatively affected by the disturbance unlike *S. afrofischeri* in which abundance was only to a small extend negatively affected by stress. However a relatively high positive increase in biomass per unit increase in stress was observed (Fig 13A and B). By biomass, the most negatively affected species was *B. altianalis* for which every unit increase in stress levels decreased the biomass by more than one unit measure (Fig 13D). The abundance of the species was however almost unaffected under disturbance conditions. In contrast, haplochromines numbers increased by almost one unit measure per equal increase in stress while their abundance almost remained the same (Fig 13C).



2006
2007
2008
2009
2010
2011
2012
2013
2014
Abundance
Biomass
W = -0.035
W = -0.104
W = -0.076
W = -0.169
W = -0.053
W = -0.238
W = -0.112
W = -0.169
W = -0.272
%Cumulative dominance
Species rank

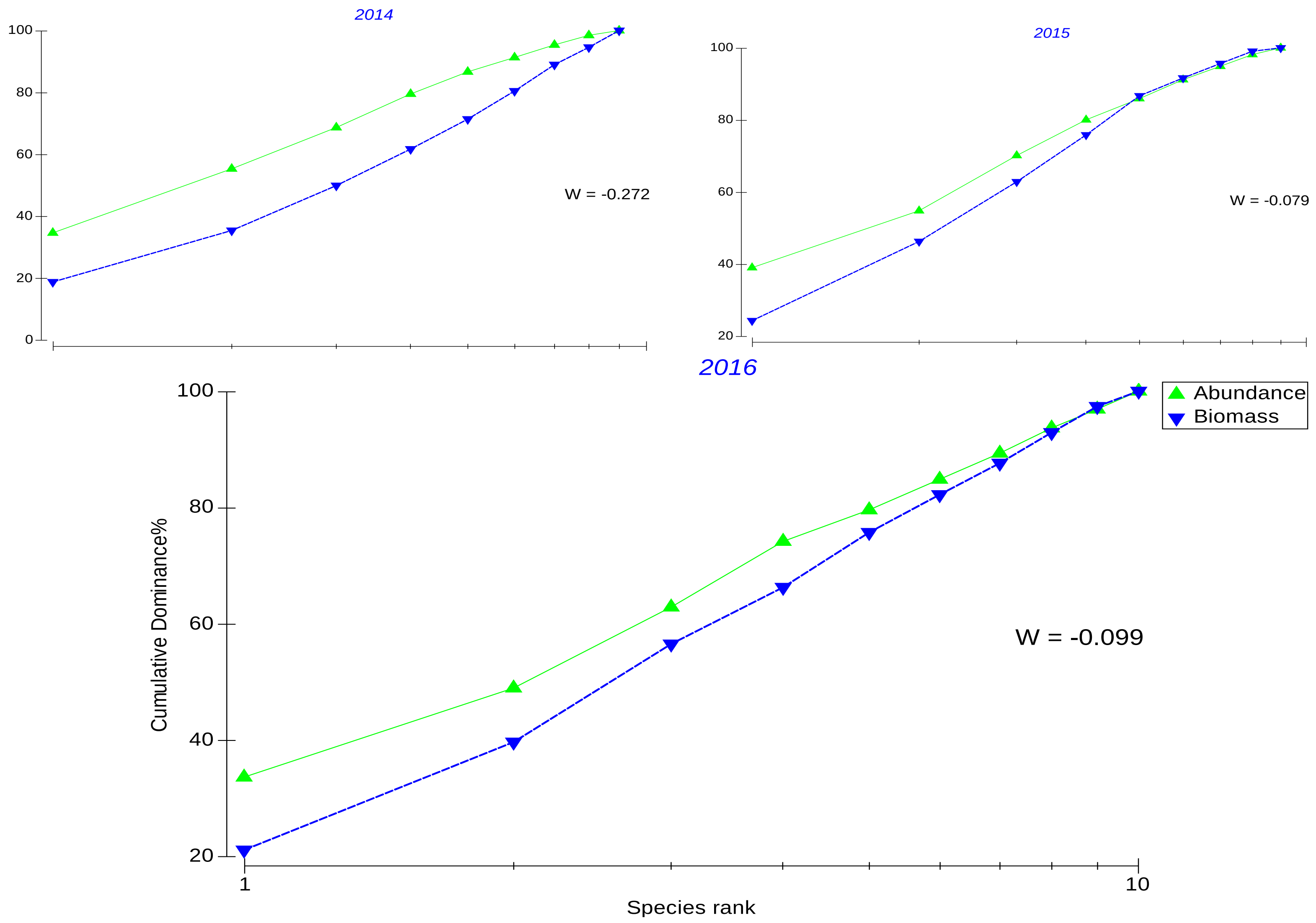


Figure 14: Abundance biomass comparison curves and the W-statistics for each year from 2006 to 2016. The magnitude the statistic is a representation of the stress levels/ disturbance the fish population in the study area have undergone since 2006.

Table 8 Correlation between stress/disturbance and fish population parameters length, biomass and abundance. Sample size (n) for all parameters was 11. Double (**) asterisk mean significant correlation at 0.01 confidence interval and single (*) asterisk means significance at 0.05 confidence interval. Species: L. n = *L. niloticus,* M. k = *M. kannume*, S. a= *S. afrofischeri*, Hap= *haplochromines and* B. a = *B. altianalis*

Parameter	Length			Biomass					Abundance (number)				
Species	L. n	M. k	S. a	L. n	M. k	S. a	Hap	B. a	L. n	M. k	S. a	Hap	B. a
Pearson Correlation	-.767**	.112	-.258	-.409	-.002	.638*	.077	-.425	-.267	-.465	-.100	.349	-.411
Sig. (2-tailed)	.006	.743	.443	.211	.995	.035	.428	.150	.771	.822	.292	.210	.193

CHAPTER FIVE

DISCUSSION

5.1 Contemporary fish species abundance, diversity and size structure

On average, a total of about 900 fish specimens had been annually recovered from all transects between 2006 and 2015 (NaFIRRI, 2015). In this study, a high number of fish were recovered as compared to earlier ecological baseline and impact monitoring studies due to a higher three months sampling effort in this study. The haplochromine group of cichlid fishes like in this study had dominated abundance in all earlier studies undertaken in the same area even before the construction of Bujagali dam (FIRRI, 2000). Before the construction Owen fall dam, Lake Victoria and the upper Victoria Nile shared similar waters and ichthyofauna (Beadle, 1974). Haplochromines almost entirely comprised the fish biomass in Lake Victoria prior to the introduction the Nile Perch (Witte *et al*., 1995). However the introduction of the highly vicious predator in the 1950s resulted in the disappearance of more than 80% of the haplochromine species flocks in the Lake (Witte *et al*., 1992; Kaufman, 1997). Of recent, it has been discovered that rocky habitants and hypoxic wetlands are important refugia grounds from the Nile perch predation (Balirwa *et al*., 2003). It is likely that the high abundance of rocky habitats in the upper Victoria Nile accounts for the high number of haplochromines in this system.

In relation to the abundance of Nile perch and rocky habitats, it was expected that haplochromine abundance could increase towards the downstream areas as noted by Musenero (2000). However similar to finding by NaFIRRI (2006), haplochromine abundance in this study was lower in the downstream transects. This disparity could most likely have been due to differences in fishing effort and season. For example notable differences in fish abundance were reported between April and August surveys undertaken in the same study area in 2000 (FIRRI 2000). This seasonal variation may account for the big differences in abundance of fish in 2006 when sampling was undertaken in rainy April and 2016 where sampling occurred between relatively dry December and February.

The high abundance of *L. niloticus* in the upstream transects of Kalange and Reservoir as compared to the downstream Buyala and Reservoir is similar to observations by Musenero (2000) and Atkins (2001). It is possibly the gradual downstream increase in rocky habitats in the upper Victoria Nile that makes the environment hostile for the Nile perch, a species that prefers living in deep well oxygenated waters (Barlow & Lisle, 1987). Similarly the abundance of *S. afrofischeri* and *S. victoriae* in this study was higher in the upstream transects than the downstream ones. Although this observation is in agreement with finding by NaFIRRI (2015), it contradicts reports by Atkins (2001) and NaFIRRI (2006) who observed the species as being more abundant in the downstream transects. It is probable that the ecological association between the *Synodontis* species and *M. kannume* accounts for this observation. According to Greenwood (1966), *Synodontis* and *M. kannume* are both predominantly insect feeders. It's highly likely that *M. kannume* outcompetes the *Synodontis* for the food resources thus declines in the latter in the high abundance of the former and vice versa (Fig. 2). Since *M. kannume* is one of the most targeted species in the commercial fishery (NaFIRRI, 2015), it is also likely that the increasing fishing pressure on the species has increased feeding opportunities for its resource competitor.

It can be expected that riverine species *M. kannume*, *S. afrofischeri*, *S. victoriae* and *B. docmak* and *B. altianalis* would be in higher abundancies in the downstream Buyala transect than the upstream Kalange transect. However on the contrary, all the species except *B. altianalis* have been recorded in both this and previous studies as being more abundant in currently upstream areas than the downstream (NaFIRRI, 2015). Various studies have shown that riverine fish species capable of completing their life cycles in both lotic and lentic environments (facultative riverine species) are less affected by impoundments compared to their obligate counterparts (Wolter, 2001; Kruk and Penczak, 2003). Whereas the biology and ecology of most African tropical fishes is yet to be fully understood, it is known that *M. kannume*, *B. docmak*, S. *victoriae* and *S. afrofischeri* can breed in both lakes and rivers (Lowe-McConnell, 1975; Mekkawy & Hassan, 2012). In contrast, *B. altianalis* has been reported to breed in only flowing waters (Tómasson *et al*., 1984; Rutaisire *et al*, 2013). For this reason, as the abundance of the other species remains still high in the upstream transects, that of *B. altianalis* has declined to almost insignificant levels.

Nonetheless, the high productively of the upstream Kalange transect in comparison to others can also be attributed to its location in the tail waters of the two upstream hydropower plants Kiira and Nalubaale. Tailwaters of hydropower plants have been reported to highly favour fisheries productively due the periodic release of waters with high abundance of food material which accumulates upstream when the dam is closed (Dreves *et al*., 2014). However the small size of specimens recovered from this transect as compared to others further downstream can be attributed to the higher commercial fishing activities (NaFIRRI, 2015). For this reason, the populations in the furthest downstream Kirindi transect where fishing is more or less non-existent (Atkins, 2001) grow to large sizes. However, the smaller size of *O. niloticus* at Kirindi transect can be attributed to the sample considered in the calculation of the mean size, all of which were small sized juveniles obtained through seining (Table 3).

The nonexistence of fishing activities at Kirindi also suggests that the decrease in the relative abundance and biomass of fishes observed at all transects in this study can possibly also be due to dam construction and operation.

5.1 Spatial-temporal changes in abundance and biomass, diversity and size structure

5.1.1 Abundance and biomass

The clustering of the different year groups at Buyala and Kalange are a depiction of the events that occurred from the onset of Bujagali dam construction to its closing and subsequent operation. Similar patterns of change during each phase of dam establishment were reported by Liu *et al.* (2013) and Warren (1999). The fish populations' shift to another phase after onset of construction in 2007 was possibly in response to poor quality water conditions during construction as reported by NaFIRRI (2007b). However the shit after dam closing and operation in 2013 was likely due to creation of semi lentic conditions upstream and flow alternation in the downstream areas. It was observed by Miranda *et al*. (2012) that upstream dam operation activities significantly affected fish populations in the downstream areas.

Higher dissimilarities among the year groups in the downstream transect as compared to its upstream counterpart reflects higher disturbance conditions downstream. This is in agreement to studies that have noted that the ecological impacts of dams magnify towards the downstream sections of the dammed river (Ligon *et al*., 1995; Richter *et al*., 2010). It was also noted in this study that the dissimilarity between the baseline conditions and early post dam closing era were higher as compared that between the baseline and the construction era. This observation suggests that alternation of the rivers flow regime had a significant effect on the fish population that the decline in water quality during construction.

Haplochromines, *L. niloticus*, *S. afrofischeri* and *M. kannume* were the species most responsible for the observed similarities/dissimilarities among the different year groups. This indicates they were the most dominant over the years as reported in all previous reports

(FIRRI, 2000; Atkins, 2001). The decline in the abundance of the species can be attributed to disturbances related to the construction and operation of the dam (Ligon *et al*., 1995).

The 0.3 R value obtained from ANOSIM comparison of abundance as biomass among the years is an indication of moderate ecological differences among them possibly due to the short time lag between the dam closing and the time of the study. A similar observation was made by Quinn and Kwak (2003) in Ozark river USA.

5.1.2 Diversity

The trends in diversity at Buyala with and without haplochromines were different, indicating haplochromines play a less role in the population dynamics of this transect as compared to upstream Kalange. However despite the central role they play, the contrasting diversity registered at Kalange in 2011 with and without haplochromines suggested the big influence of other dominant species such *L. niloticus*, *M. kannume* and *S. afrofischeri*. The general declining trends in the diversity at Buyala suggests a cumulative disturbance on downstream fish communities from dam construction and operation. It can be expected that being located in the tailwaters of Bujagali dam, the diversity in Buyala transect could increase over time (Dreves *et al*., 2014). However, observations on changes in diversity downstream of dams remain controversial. Whereas a number of studies have reported downstream declines in diversity following dam construction and operation, many more have also reported contrary (Quinn & Kwak, 2003; Agostinho *et al*., 2008; Gardner *et al*., 2011; Sa-oliveira *et al*., 2015). The real cause of these disparities still remains unclear, however some authors suggest that data variability and diversity of downstream conditions could be major contributing factors to these contradicting observations (Me´rona *et al*., 2005).

The upstream and downstream increase in diversity in 2012 was due to the increased abundance of large bodied species *L. niloticus*, *M. kannume* and *S. afrofischeri* at both transects (Fig 2). However the observed shifts in biomass observed following the commissioning of dam operation were most likely due to ecological changes that took place. It can be expected that initial upstream fish assemblages changes upon dam operation could be a decline in diversity as lentic intolerant species disappear as noted by Jackson *et al*. (1988). However in this study, there was a persistence of riverine species *M. kannume, B. docmak,* and *S. afrofischeri* in the upstream sections. This indicates their high adaptive capacity to lentic condition possibly due to their facultative nature (Lowe-McConnell, 1975; Kruk & Penczak, 2003).

5.1.3 Size structure

It has been observed in this study that the overall mean total lengths of the three numerically most dominant species in the period 2006-2016 were significantly lower in the downstream areas than the upstream areas. This observation conforms to other reports which have noted dam effects as generally magnifying towards the downstream areas (Ligon *et al*., 1995). Although *L. niloticus* is known to grow to a very larger size than *M. kannume,* such large specimens of the species have never been encountered in any previous study. It is still unclear whether this is due to the intensifying fishing pressure or the lotic conditions not favouring the optimum growth of the predator compared to its evolutionary lentic waters (Barlow & Lisle, 1987).

The decline in the mean size of *M. kannume* almost immediately following dam construction indicates the vulnerability of facultative riverine fish species to flow modification (Kruk & Penczak, 2003). However the steady increase in the size of the species after some time proves the resilient nature of facultative riverine fish species to hydrological alternations (Wolter, 2001; Penczak *et al*, 2002). *M. kannume* registered the smallest size in 2007 suggesting the species was very sensitive to the decreased water quality which occurred in that year (NaFIRRI, 2007b). In contrast *S. afrofischeri* was less affected by the poor water quality conditions of 2007 as shown by the increase in its mean size in that year (Fig 11). It is likely that the ability of the species to undertake aquatic surface respiration gives it more resilience to poor water quality conditions than *M. kannume* (Chapman *et al*., 1994). However, the immediate decline in the size of the species upon onset of dam operation in 2012 is an indicator of its sensitivity and vulnerability to physical flow alterations.

5.3 Spatial-temporal Trends and Degree of Dam Impact on Fish Populations

The negative W-statistics values in 2006 at both Kalange and Buyala are an indication of a fish population already under stress by the time the construction of Bujagali dam started. It is likely that this stress is a function of the upstream Nalubaale and Kiira dams which were constructed years before Bujagali. However it cannot be reliably determined in this study if the natural high abundance of small bodied fish species in the study area did not interfere with the W-statistic out-puts. Such natural ecosystem limitations have also been observed in similar studies that have successfully applied the ABC method in ecosystems quality assessments (Penczak and Kruk, 1999; Piperac *et al*., 2015). Nonetheless, the immediate increase in stress levels upon inception of dam construction in 2007 and its subsequent peaking at both transects upon commissioning of dam operation in 2012 (Table 7) indicates the dam as a source of the stress.

The higher stress values at the downstream Buyala transect is in agreement with Richter *et al*., (2010), who reported hydropower plant's impacts to magnify towards the downstream areas. However the lower stress at the upstream Kalange transect in comparison to Buyala in 2006 could be due to the location of the former in more or less the tailwaters of the two upstream hydropower plants Nalubaale and Kiira which haves fisheries productivity (Dreves *et al.*, 2014). The peaking of stress levels between 2011 and 2013 were due to the general decline in biomass at both transects which also peaked around the same years (Table 7). Reduction in the biomass of the large individuals during this period is supported by results of the size structure analysis which revealed the least mean total lengths of the large bodied species *M. kannume* and *L. niloticus* as being within the same period (Fig 8 and 10). It is likely that the continued decline in the biomass of one of the most dominant species *B. altianalis* may also have greatly contributed to the observed pattern *(*Fig 2*).*

The positive condition recorded at Buyala in 2010 was due to high abundance of large bodied species *L. niloticus* in that transect that year. Similarly, the observed decrease in stress levels in 2016 at Buyala was as a result of the high abundance of large bodied species *M. kannume* and the same reason was responsible for the decline in stress levels at Kalange in 2015 and 2016 (Fig. 2). Variations in *L. niloticus* and *M. kannume* at both transects could be associated to patterns in the commercial fishing activities in the study area. It is therefore highly likely that the observed disturbance patterns could be a combination of both dam and fishing pressure.

The positive relationship between stress and the small bodied species (haplochromines and *S. afrofischeri*) supports the basis on which the ABC model was built, such small fast growing species will always flourish under the stressful conditions (Warwick, 1986). However the absence of more less no relationship between stress and the large bodied *M. kannume* species is evidence of the high adaptive ability of this species under harsh conditions (Kruk and Penczak, 2003). The obligate riverine *B. altianalis* possess poor abilities to handle flow modification related stress hence the high negative relationship with stress.

CHAPTER SIX CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

Based on the objectives and results obtained in this study, it can be concluded that:

i) The species abundance upstream is dominated by the haplochromine cichlids, *L. niloticus*, *M.kannume* and *S. afrofischeri* compared to downstream due to habitat modification to a more lacustrine environment. The high abundance of *M. kannume* and *S. afrofischeri* reflects on the resilience and adaptive ability facultative riverine fish species to flow modification.

ii) a) The mean size of the most dominant species *L. niloticus*, *M. kannume* and *B. docmak* were higher at the most downstream Kirindi transect than the other three implying less stressful conditions in this transect.

b) There is no significant differences in the fish species diversity in Kalange, Reservoir, Buyala and Kirindi.

iii) There were significant inter-annual and spatial differences in the mean total length of the three most dominant species *L. niloticus, M. kannume* and *S. afrofischeri* over the years. This observation indicates the rapid adaptation of predatory and facultative riverine fish species to flow modification. No significant difference were also observed in the temporal variation in diversity of the upstream and downstream transect. However a general declining temporal trend possibly suggests a system which integrity is deteriorating. Significant temporal and spatial variations in species abundance and biomass were observed. These observations were directly to the events of dam construction and operation.

iv) No significant differences in the stress at the sampled sites in different years for the period 2006-2007, but an apparent increase in the stress levels was observed from the time of construction to 2016, suggesting an increasing disturbance on the fish populations.

6.2 Recommendations

Based on this study's findings, it is recommended that:

i) It has been observed that the most sensitive species to physical flow modification in this system are *B. altianalis* and *S. afrofischeri* while *M. kannume* is very sensitive to water quality changes. These species should be used as the principle keystone species for the purpose of regular monitoring of the impact of Bujagali dam on the upper Nile River.

ii) *M.kannume*, *S. afrofischeri* and *L. niloticus* size structure are all suitable for site comparison among years. However it is only the size structure of the *M.kannume* and *S. afrofischeri* which are suitable for long term monitoring since there are no distinct annual differentiation observed in the size structure of *L. niloticus*. The use of diversity indices in monitoring is recommended but their suitability as ecological impact indicators of damming in the upper Nile River may require more research.

iii) The ABC method is effective in quantifying the level of human impact on the fish populations of the Upper Victoria Nile. This method should be adopted for analysing the impact of human activities in the upper Nile River in addition to monitoring the impact of Bujagali dam on a larger spatial scale.

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APPENDICES

APPENDIX I:

Two-Way ANOVA output for *L.niloticus* with means, standard deviation, sample size (N) for each year in a transect. The overall values for all the parameters and with all transects combined each year are shown in the Total section. Transect (site) codes 1= Kalange, 2= Reservoir and 3= Buyala. Year (Yr1) codes 1=2006, 2=2007, 3=2008, 4=2009, 5=2010, 6= 2011, 7=2012, 8=2013, 9=2014, 10=2015 and 11= 2016.

Site	Yr1	Mean	Std. Deviation	N
1	1.0	19.368	2.9290	19
	2.0	17.905	3.7047	95
	3.0	19.821	5.7773	84
	4.0	16.821	4.2330	123
	5.0	18.786	4.6122	70
	6.0	18.143	7.6635	21
	7.0	16.895	4.1218	171
	8.0	17.375	6.7875	24
	9.0	16.857	8.7652	21
	10.0	21.158	10.8077	19
	11.0	20.240	5.4182	25
	Total	17.957	5.2400	672
2	7.0	21.910	7.2114	78
	Total	21.910	7.2114	78
3	1.0	18.029	5.7293	104
	2.0	16.956	3.9827	45
	3.0	17.865	4.3703	52
	4.0	16.127	3.5326	110
	5.0	18.185	4.9338	54
	6.0	17.200	3.5509	55
	7.0	14.750	4.5453	192
	8.0	16.350	3.4224	20
	9.0	13.314	3.9874	51
	10.0	12.556	5.3947	27
	11.0	17.937	3.1298	16
	Total	16.161	4.7376	726
Total	1.0	18.236	5.4050	123
	2.0	17.600	3.8080	140
	3.0	19.074	5.3522	136

4.0	16.494	3.9251	233
5.0	18.524	4.7446	124
6.0	17.461	4.9919	76
7.0	16.848	5.5765	441
8.0	16.909	5.4849	44
9.0	14.347	5.9556	72
10.0	16.109	9.0486	46
11.0	19.341	4.7519	41
Total	17.283	5.3100	1476

APPENDIX II:

Output of two-Way ANOVA multiple annual comparisons (Tukey HSD) for the annual mean total lengths of *L.niloticus*. Year (Yr1) codes 1=2006, 2=2007, 3=2008, 4=2009, 5=2010, 6= 2011, 7=2012, 8=2013, 9=2014, 10=2015 and 11= 2016.

(I) Yr1	(J) Yr1	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval Lower Bound	Upper Bound
1.0	2.0	.636	.6116	.994	-1.336	2.608
	3.0	-.838	.6158	.958	-2.823	1.148
	4.0	1.742	.5516	.061	-.036	3.520
	5.0	-.288	.6298	1.000	-2.319	1.742
	6.0	.775	.7221	.993	-1.553	3.103
	7.0	1.388	.5047	.179	-.239	3.015
	8.0	1.327	.8694	.911	-1.476	4.129
	9.0	3.889*	.7344	.000	1.521	6.256
	10.0	2.127	.8554	.313	-.630	4.885
	11.0	-1.106	.8925	.978	-3.983	1.772
2.0	1.0	-.636	.6116	.994	-2.608	1.336
	3.0	-1.474	.5959	.321	-3.395	.447
	4.0	1.106	.5292	.584	-.600	2.813
	5.0	-.924	.6103	.915	-2.892	1.043
	6.0	.139	.7052	1.000	-2.134	2.413
	7.0	.752	.4801	.896	-.796	2.300
	8.0	.691	.8554	.999	-2.067	3.448
	9.0	3.253*	.7177	.000	.939	5.567
	10.0	1.491	.8411	.796	-1.220	4.203
	11.0	-1.741	.8789	.662	-4.575	1.092

3.0	1.0	.838	.6158	.958	-1.148	2.823
	2.0	1.474	.5959	.321	-.447	3.395
	4.0	2.580*	.5341	.000	.858	4.302
	5.0	.549	.6145	.998	-1.432	2.530
	6.0	1.613	.7088	.451	-.672	3.898
	7.0	2.225*	.4854	.000	.660	3.790
	8.0	2.164	.8584	.293	-.603	4.932
	9.0	4.726*	.7213	.000	2.401	7.052
	10.0	2.965*	.8442	.020	.243	5.686
	11.0	-.268	.8818	1.000	-3.111	2.575
4.0	1.0	-1.742	.5516	.061	-3.520	.036
	2.0	-1.106	.5292	.584	-2.813	.600
	3.0	-2.580*	.5341	.000	-4.302	-.858
	5.0	-2.031*	.5501	.011	-3.804	-.257
	6.0	-.967	.6538	.927	-3.075	1.141
	7.0	-.355	.4008	.998	-1.647	.938
	8.0	-.416	.8135	1.000	-3.038	2.207
	9.0	2.146	.6673	.051	-.005	4.298
	10.0	.385	.7985	1.000	-2.189	2.959
	11.0	-2.848*	.8382	.029	-5.550	-.146
5.0	1.0	.288	.6298	1.000	-1.742	2.319
	2.0	.924	.6103	.915	-1.043	2.892
	3.0	-.549	.6145	.998	-2.530	1.432
	4.0	2.031*	.5501	.011	.257	3.804
	6.0	1.064	.7210	.928	-1.261	3.388
	7.0	1.676*	.5031	.036	.054	3.298
	8.0	1.615	.8685	.744	-1.185	4.415
	9.0	4.177*	.7333	.000	1.813	6.541
	10.0	2.415	.8544	.149	-.339	5.170
	11.0	-.817	.8916	.998	-3.692	2.057
6.0	1.0	-.775	.7221	.993	-3.103	1.553
	2.0	-.139	.7052	1.000	-2.413	2.134
	3.0	-1.613	.7088	.451	-3.898	.672
	4.0	.967	.6538	.927	-1.141	3.075
	5.0	-1.064	.7210	.928	-3.388	1.261
	7.0	.612	.6147	.996	-1.369	2.594
	8.0	.551	.9375	1.000	-2.471	3.574
	9.0	3.113*	.8139	.006	.489	5.737
	10.0	1.352	.9245	.932	-1.629	4.332
	11.0	-1.881	.9590	.676	-4.973	1.211

7.0	1.0	-1.388	.5047	.179	-3.015	.239
	2.0	-.752	.4801	.896	-2.300	.796
	3.0	-2.225*	.4854	.000	-3.790	-.660
	4.0	.355	.4008	.998	-.938	1.647
	5.0	-1.676*	.5031	.036	-3.298	-.054
	6.0	-.612	.6147	.996	-2.594	1.369
	8.0	-.061	.7825	1.000	-2.584	2.461
	9.0	2.501*	.6291	.004	.473	4.529
	10.0	.739	.7668	.997	-1.733	3.212
	11.0	-2.493	.8081	.075	-5.098	.112
8.0	1.0	-1.327	.8694	.911	-4.129	1.476
	2.0	-.691	.8554	.999	-3.448	2.067
	3.0	-2.164	.8584	.293	-4.932	.603
	4.0	.416	.8135	1.000	-2.207	3.038
	5.0	-1.615	.8685	.744	-4.415	1.185
	6.0	-.551	.9375	1.000	-3.574	2.471
	7.0	.061	.7825	1.000	-2.461	2.584
	9.0	2.562	.9470	.198	-.491	5.615
	10.0	.800	1.0436	1.000	-2.564	4.165
	11.0	-2.432	1.0743	.459	-5.896	1.031
9.0	1.0	-3.889*	.7344	.000	-6.256	-1.521
	2.0	-3.253*	.7177	.000	-5.567	-.939
	3.0	-4.726*	.7213	.000	-7.052	-2.401
	4.0	-2.146	.6673	.051	-4.298	.005
	5.0	-4.177*	.7333	.000	-6.541	-1.813
	6.0	-3.113*	.8139	.006	-5.737	-.489
	7.0	-2.501*	.6291	.004	-4.529	-.473
	8.0	-2.562	.9470	.198	-5.615	.491
	10.0	-1.761	.9342	.727	-4.773	1.250
	11.0	-4.994*	.9683	.000	-8.116	-1.873
10.0	1.0	-2.127	.8554	.313	-4.885	.630
	2.0	-1.491	.8411	.796	-4.203	1.220
	3.0	-2.965*	.8442	.020	-5.686	-.243
	4.0	-.385	.7985	1.000	-2.959	2.189
	5.0	-2.415	.8544	.149	-5.170	.339
	6.0	-1.352	.9245	.932	-4.332	1.629
	7.0	-.739	.7668	.997	-3.212	1.733
	8.0	-.800	1.0436	1.000	-4.165	2.564
	9.0	1.761	.9342	.727	-1.250	4.773
	11.0	-3.233	1.0630	.085	-6.660	.194

11.0	1.0	1.106	.8925	.978	-1.772	3.983
	2.0	1.741	.8789	.662	-1.092	4.575
	3.0	.268	.8818	1.000	-2.575	3.111
	4.0	2.848*	.8382	.029	.146	5.550
	5.0	.817	.8916	.998	-2.057	3.692
	6.0	1.881	.9590	.676	-1.211	4.973
	7.0	2.493	.8081	.075	-.112	5.098
	8.0	2.432	1.0743	.459	-1.031	5.896
	9.0	4.994*	.9683	.000	1.873	8.116
	10.0	3.233	1.0630	.085	-.194	6.660

APPENDIX III:

Two-Way ANOVA output for *M. kannume* means, standard deviation, sample size (N) for each year in a transect. The overall values for all the parameters and with all transects combined each year are shown in the Total section. Transect (site) codes 1= Kalange, 2= Reservoir and 3= Buyala. Year (Yr2) codes 1=2006, 2=2007, 3=2008, 4=2009, 5=2010, 6= 2011, 7=2012, 8=2013, 9=2014, 10=2015 and 11= 2016.

Dependent Variable: Mk

Site 2	Yr2	Mean	Std. Deviation	N
1.0	1.0	28.939	9.8360	33
	2.0	28.870	11.2464	23
	3.0	26.727	8.6098	55
	4.0	21.961	7.1523	23
	5.0	25.969	11.5911	32
	6.0	24.100	7.8129	20
	7.0	24.000	7.8435	26
	8.0	25.200	4.2374	10
	9.0	38.000	8.2865	4
	10.0	27.750	18.7598	8
	11.0	22.222	5.6937	18
	Total	26.020	9.6080	252
2.0	7.0	24.392	6.5179	51
	8.0	26.289	7.5763	38
	9.0	31.211	11.0016	38
	10.0	23.507	10.0444	67
	11.0	31.458	10.7581	24

	Total	26.417	9.6737	218
3.0	1.0	28.754	10.9362	69
	2.0	20.875	4.2866	24
	3.0	20.400	3.9738	45
	4.0	22.947	13.8342	19
	5.0	21.667	3.5751	30
	6.0	19.000	4.2336	27
	7.0	24.114	5.4398	44
	10.0	41.000	25.4558	2
	11.0	31.750	7.8876	8
	Total	23.877	8.6935	268
Total	1.0	28.814	10.5443	102
	2.0	24.787	9.2735	47
	3.0	23.880	7.5803	100
	4.0	22.407	10.5697	42
	5.0	23.887	8.8912	62
	6.0	21.170	6.4684	47
	7.0	24.207	6.4147	121
	8.0	26.063	6.9875	48
	9.0	31.857	10.8775	42
	10.0	24.403	11.7342	77
	11.0	28.180	9.7388	50
	Total	25.359	9.3623	738

APPENDIX IV:

Output of two-Way ANOVA multiple annual comparisons (Tukey HSD) for the annual mean total lengths of *L.niloticus*. Year (Yr1) codes 1=2006, 2=2007, 3=2008, 4=2009, 5=2010, 6= 2011, 7=2012, 8=2013, 9=2014, 10=2015 and 11= 2016.

(I) Yr2	(J) Yr2	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval Lower Bound	95% Confidence Interval Upper Bound
1.0	2.0	4.026	1.5510	.253	-.982	9.035
	3.0	4.934*	1.2381	.004	.936	8.932
	4.0	6.407*	1.6130	.004	1.198	11.615
	5.0	4.927*	1.4168	.023	.352	9.502
	6.0	7.644*	1.5510	.000	2.635	12.652
	7.0	4.607*	1.1826	.005	.788	8.426
	8.0	2.751	1.5399	.788	-2.221	7.724
	9.0	-3.043	1.6130	.726	-8.252	2.165
	10.0	4.411*	1.3282	.038	.122	8.700
	11.0	.634	1.5188	1.000	-4.271	5.538
2.0	1.0	-4.026	1.5510	.253	-9.035	.982
	3.0	.907	1.5559	1.000	-4.117	5.932
	4.0	2.380	1.8681	.973	-3.652	8.412
	5.0	.900	1.7015	1.000	-4.594	6.395
	6.0	3.617	1.8148	.654	-2.243	9.478
	7.0	.581	1.5121	1.000	-4.302	5.464
	8.0	-1.275	1.8054	1.000	-7.105	4.555
	9.0	-7.070*	1.8681	.008	-13.102	-1.038
	10.0	.385	1.6285	1.000	-4.874	5.643
	11.0	-3.393	1.7874	.719	-9.165	2.379
3.0	1.0	-4.934*	1.2381	.004	-8.932	-.936
	2.0	-.907	1.5559	1.000	-5.932	4.117
	4.0	1.473	1.6177	.998	-3.751	6.697
	5.0	-.007	1.4221	1.000	-4.599	4.585
	6.0	2.710	1.5559	.813	-2.315	7.734
	7.0	-.327	1.1890	1.000	-4.166	3.513
	8.0	-2.182	1.5448	.945	-7.171	2.806
	9.0	-7.977*	1.6177	.000	-13.201	-2.753
	10.0	-.523	1.3339	1.000	-4.830	3.785
	11.0	-4.300	1.5238	.152	-9.221	.621
4.0	1.0	-6.407*	1.6130	.004	-11.615	-1.198
	2.0	-2.380	1.8681	.973	-8.412	3.652

	3.0	-1.473	1.6177	.998	-6.697	3.751
	5.0	-1.480	1.7582	.999	-7.158	4.198
	6.0	1.237	1.8681	1.000	-4.795	7.269
	7.0	-1.799	1.5756	.988	-6.887	3.288
	8.0	-3.655	1.8589	.672	-9.658	2.347
	9.0	-9.450*	1.9198	.000	-15.650	-3.250
	10.0	-1.995	1.6876	.984	-7.445	3.454
	11.0	-5.773	1.8414	.066	-11.719	.174
5.0	1.0	-4.927*	1.4168	.023	-9.502	-.352
	2.0	-.900	1.7015	1.000	-6.395	4.594
	3.0	.007	1.4221	1.000	-4.585	4.599
	4.0	1.480	1.7582	.999	-4.198	7.158
	6.0	2.717	1.7015	.884	-2.778	8.211
	7.0	-.320	1.3741	1.000	-4.757	4.118
	8.0	-2.175	1.6914	.971	-7.637	3.287
	9.0	-7.970*	1.7582	.000	-13.648	-2.292
	10.0	-.516	1.5012	1.000	-5.363	4.332
	11.0	-4.293	1.6722	.268	-9.693	1.107
6.0	1.0	-7.644*	1.5510	.000	-12.652	-2.635
	2.0	-3.617	1.8148	.654	-9.478	2.243
	3.0	-2.710	1.5559	.813	-7.734	2.315
	4.0	-1.237	1.8681	1.000	-7.269	4.795
	5.0	-2.717	1.7015	.884	-8.211	2.778
	7.0	-3.036	1.5121	.643	-7.919	1.847
	8.0	-4.892	1.8054	.197	-10.722	.938
	9.0	-10.687*	1.8681	.000	-16.719	-4.655
	10.0	-3.232	1.6285	.660	-8.491	2.026
	11.0	-7.010*	1.7874	.005	-12.782	-1.238
7.0	1.0	-4.607*	1.1826	.005	-8.426	-.788
	2.0	-.581	1.5121	1.000	-5.464	4.302
	3.0	.327	1.1890	1.000	-3.513	4.166
	4.0	1.799	1.5756	.988	-3.288	6.887
	5.0	.320	1.3741	1.000	-4.118	4.757
	6.0	3.036	1.5121	.643	-1.847	7.919
	8.0	-1.856	1.5007	.978	-6.702	2.990
	9.0	-7.651*	1.5756	.000	-12.738	-2.563
	10.0	-.196	1.2825	1.000	-4.338	3.946
	11.0	-3.973	1.4791	.208	-8.750	.803
8.0	1.0	-2.751	1.5399	.788	-7.724	2.221

	2.0	1.275	1.8054	1.000	-4.555	7.105
	3.0	2.182	1.5448	.945	-2.806	7.171
	4.0	3.655	1.8589	.672	-2.347	9.658
	5.0	2.175	1.6914	.971	-3.287	7.637
	6.0	4.892	1.8054	.197	-.938	10.722
	7.0	1.856	1.5007	.978	-2.990	6.702
	9.0	-5.795	1.8589	.069	-11.797	.208
	10.0	1.660	1.6179	.995	-3.565	6.885
	11.0	-2.117	1.7778	.983	-7.858	3.623
9.0	1.0	3.043	1.6130	.726	-2.165	8.252
	2.0	7.070*	1.8681	.008	1.038	13.102
	3.0	7.977*	1.6177	.000	2.753	13.201
	4.0	9.450*	1.9198	.000	3.250	15.650
	5.0	7.970*	1.7582	.000	2.292	13.648
	6.0	10.687*	1.8681	.000	4.655	16.719
	7.0	7.651*	1.5756	.000	2.563	12.738
	8.0	5.795	1.8589	.069	-.208	11.797
	10.0	7.455*	1.6876	.001	2.005	12.904
	11.0	3.677	1.8414	.651	-2.269	9.624
10.0	1.0	-4.411*	1.3282	.038	-8.700	-.122
	2.0	-.385	1.6285	1.000	-5.643	4.874
	3.0	.523	1.3339	1.000	-3.785	4.830
	4.0	1.995	1.6876	.984	-3.454	7.445
	5.0	.516	1.5012	1.000	-4.332	5.363
	6.0	3.232	1.6285	.660	-2.026	8.491
	7.0	.196	1.2825	1.000	-3.946	4.338
	8.0	-1.660	1.6179	.995	-6.885	3.565
	9.0	-7.455*	1.6876	.001	-12.904	-2.005
	11.0	-3.777	1.5979	.392	-8.937	1.382
11.0	1.0	-.634	1.5188	1.000	-5.538	4.271
	2.0	3.393	1.7874	.719	-2.379	9.165
	3.0	4.300	1.5238	.152	-.621	9.221
	4.0	5.773	1.8414	.066	-.174	11.719
	5.0	4.293	1.6722	.268	-1.107	9.693
	6.0	7.010*	1.7874	.005	1.238	12.782
	7.0	3.973	1.4791	.208	-.803	8.750
	8.0	2.117	1.7778	.983	-3.623	7.858

9.0	-3.677	1.8414	.651	-9.624	2.269
10.0	3.777	1.5979	.392	-1.382	8.937

APPENDIX V:

Two-Way ANOVA output for *S. afrofischeri* with means, standard deviation, sample size (N) for each year in a transect. The overall values for all the parameters and with all transects combined each year are shown in the Total section. Transect (site) codes 1= Kalange, 2= Reservoir and 3= Buyala. Year (Yr3) codes 1=2006, 2=2007, 3=2008, 4=2009, 5=2010, 6= 2011, 7=2012, 8=2013, 9=2014, 10=2015 and 11= 2016.

Site 3	Yr 3	Mean	Std. Deviation	N
1	1	13.200	1.7889	5
	2	14.593	.8207	15
	3	13.500	.7071	2
	4	11.900	1.9048	46
	5	12.895	1.7605	19
	6	14.273	1.5551	11
	7	9.135	2.6129	111
	8	11.275	1.7127	222
	9	10.609	2.3426	46
	10	12.480	.8355	40
	11	13.292	6.2309	24
	Total	11.247	2.6793	541
2	8	12.733	1.1356	12
	10	13.333	2.0817	3
	11	11.545	2.2962	11
	Total	12.300	1.8687	26
3	2	14.500	1.3784	6
	3	10.350	3.0419	24
	4	11.000	1.3936	70
	5	12.833	1.6021	6
	6	12.286	.9694	14
	7	10.179	1.6853	56
	8	12.000	.0000	2
	9	11.333	1.3663	6
	10	11.944	.9376	18
	11	12.174	.9367	23
	Total	11.162	1.8785	225
Total	1	13.200	1.7889	5

	2	14.567	.9738	21
	3	10.592	3.0439	26
	4	11.357	1.6675	116
	5	12.880	1.6912	25
	6	13.160	1.5906	25
	7	9.485	2.3894	167
	8	11.355	1.7104	236
	9	10.692	2.2539	52
	10	12.364	.9778	61
	11	12.517	4.1728	58
	Total	11.257	2.4594	792

APPENDIX VI:

Output of two-Way ANOVA multiple annual comparisons (Tukey HSD) for the annual mean total lengths of *S.afrofischeri*. Year (Yr3) codes 1=2006, 2=2007, 3=2008, 4=2009, 5=2010, 6= 2011, 7=2012, 8=2013, 9=2014, 10=2015 and 11= 2016.

(I) Yr 3	(J) Yr 3	Mean Difference (I-J)	Std. Error	Sig.[d]	95% Confidence Interval for Difference[d]	
					Lower Bound	Upper Bound
1	2	-1.347[a,b]	1.083	.214	-3.473	.780
	3	1.275[a,b]	1.234	.302	-1.148	3.698
	4	1.750[a,b]	.974	.073	-.163	3.663
	5	.336[a,b]	1.076	.755	-1.776	2.448
	6	-.079[a,b]	1.045	.940	-2.132	1.973
	7	3.543[a,b,*]	.969	.000	1.641	5.446
	8	1.197[a]	1.098	.276	-.958	3.353
	9	2.229[a,b,*]	1.060	.036	.149	4.309
	10	.614[a]	1.057	.561	-1.461	2.689
	11	.863[a]	.999	.388	-1.098	2.824
2	1	1.347[a,b]	1.083	.214	-.780	3.473
	3	2.622[a,b,*]	.938	.005	.780	4.463
	4	3.097[a,b,*]	.553	.000	2.011	4.182
	5	1.683[a,b,*]	.717	.019	.275	3.090
	6	1.267[a,b]	.670	.059	-.049	2.583
	7	4.890[a,b,*]	.544	.000	3.823	5.957
	8	2.544[a,*]	.749	.001	1.073	4.015
	9	3.576[a,b,*]	.692	.000	2.217	4.934
	10	1.961[a,*]	.688	.005	.609	3.312

	11	2.210[a,*]	.595	.000	1.042	3.377
3	1	-1.275[a,b]	1.234	.302	-3.698	1.148
	2	-2.622[a,b,*]	.938	.005	-4.463	-.780
	4	.475[a,b]	.810	.558	-1.115	2.065
	5	-.939[a,b]	.930	.313	-2.764	.886
	6	-1.354[a,b]	.894	.130	-3.110	.401
	7	2.268[a,b,*]	.804	.005	.691	3.846
	8	-.078[a]	.955	.935	-1.952	1.797
	9	.954[a,b]	.911	.295	-.834	2.742
	10	-.661[a]	.908	.467	-2.443	1.121
	11	-.412[a]	.839	.624	-2.059	1.235
4	1	-1.750[a,b]	.974	.073	-3.663	.163
	2	-3.097[a,b,*]	.553	.000	-4.182	-2.011
	3	-.475[a,b]	.810	.558	-2.065	1.115
	5	-1.414[a,b,*]	.539	.009	-2.471	-.357
	6	-1.829[a,b,*]	.475	.000	-2.761	-.897
	7	1.793[a,b,*]	.267	.000	1.268	2.318
	8	-.553[a]	.581	.342	-1.693	.588
	9	.479[a,b]	.505	.343	-.512	1.470
	10	-1.136[a,*]	.500	.023	-2.117	-.155
	11	-.887[a,*]	.360	.014	-1.594	-.180
5	1	-.336[a,b]	1.076	.755	-2.448	1.776
	2	-1.683[a,b,*]	.717	.019	-3.090	-.275
	3	.939[a,b]	.930	.313	-.886	2.764
	4	1.414[a,b,*]	.539	.009	.357	2.471
	6	-.415[a,b]	.658	.528	-1.708	.877
	7	3.207[a,b,*]	.529	.000	2.169	4.245
	8	.861[a]	.739	.244	-.589	2.312
	9	1.893[a,b,*]	.680	.006	.557	3.229
	10	.278[a]	.677	.681	-1.050	1.607
	11	.527[a]	.581	.365	-.614	1.668
6	1	.079[a,b]	1.045	.940	-1.973	2.132
	2	-1.267[a,b]	.670	.059	-2.583	.049
	3	1.354[a,b]	.894	.130	-.401	3.110
	4	1.829[a,b,*]	.475	.000	.897	2.761
	5	.415[a,b]	.658	.528	-.877	1.708
	7	3.622[a,b,*]	.464	.000	2.712	4.532
	8	1.277[a]	.694	.066	-.085	2.638
	9	2.308[a,b,*]	.631	.000	1.069	3.547
	10	.693[a]	.627	.269	-.538	1.924
	11	.942[a]	.523	.072	-.084	1.968

7	1	-3.543[a,b,*]	.969	.000	-5.446	-1.641
	2	-4.890[a,b,*]	.544	.000	-5.957	-3.823
	3	-2.268[a,b,*]	.804	.005	-3.846	-.691
	4	-1.793[a,b,*]	.267	.000	-2.318	-1.268
	5	-3.207[a,b,*]	.529	.000	-4.245	-2.169
	6	-3.622[a,b,*]	.464	.000	-4.532	-2.712
	8	-2.346[a,*]	.572	.000	-3.469	-1.223
	9	-1.314[a,b,*]	.494	.008	-2.285	-.343
	10	-2.929[a,*]	.489	.000	-3.890	-1.968
	11	-2.680[a,*]	.346	.000	-3.358	-2.002
8	1	-1.197[b]	1.098	.276	-3.353	.958
	2	-2.544[b,*]	.749	.001	-4.015	-1.073
	3	.078[b]	.955	.935	-1.797	1.952
	4	.553[b]	.581	.342	-.588	1.693
	5	-.861[b]	.739	.244	-2.312	.589
	6	-1.277[b]	.694	.066	-2.638	.085
	7	2.346[b,*]	.572	.000	1.223	3.469
	9	1.032[b]	.715	.149	-.371	2.435
	10	-.583	.711	.412	-1.979	.813
	11	-.334	.621	.590	-1.553	.885
9	1	-2.229[a,b,*]	1.060	.036	-4.309	-.149
	2	-3.576[a,b,*]	.692	.000	-4.934	-2.217
	3	-.954[a,b]	.911	.295	-2.742	.834
	4	-.479[a,b]	.505	.343	-1.470	.512
	5	-1.893[a,b,*]	.680	.006	-3.229	-.557
	6	-2.308[a,b,*]	.631	.000	-3.547	-1.069
	7	1.314[a,b,*]	.494	.008	.343	2.285
	8	-1.032[a]	.715	.149	-2.435	.371
	10	-1.615[a,*]	.650	.013	-2.892	-.338
	11	-1.366[a,*]	.550	.013	-2.446	-.286
10	1	-.614[b]	1.057	.561	-2.689	1.461
	2	-1.961[b,*]	.688	.005	-3.312	-.609
	3	.661[b]	.908	.467	-1.121	2.443
	4	1.136[b,*]	.500	.023	.155	2.117
	5	-.278[b]	.677	.681	-1.607	1.050
	6	-.693[b]	.627	.269	-1.924	.538
	7	2.929[b,*]	.489	.000	1.968	3.890
	8	.583	.711	.412	-.813	1.979
	9	1.615[b,*]	.650	.013	.338	2.892
	11	.249	.546	.648	-.822	1.320
11	1	-.863[b]	.999	.388	-2.824	1.098

2	-2.210[b,*]	.595	.000	-3.377	-1.042
3	.412[b]	.839	.624	-1.235	2.059
4	.887[b,*]	.360	.014	.180	1.594
5	-.527[b]	.581	.365	-1.668	.614
6	-.942[b]	.523	.072	-1.968	.084
7	2.680[b,*]	.346	.000	2.002	3.358
8	.334	.621	.590	-.885	1.553
9	1.366[b,*]	.550	.013	.286	2.446
10	-.249	.546	.648	-1.320	.822

Based on estimated marginal means

*. The mean difference is significant at the .05 level.

a. An estimate of the modified population marginal mean (I).

b. An estimate of the modified population marginal mean (J).

d. Adjustment for multiple comparisons: Least Significant Difference (equivalent to no adjustments).