HYDRO-BIOLOGY AND FISH PRODUCTION OF THE BLACK VOLTA NEAR THE BUI DAM DURING THE PRE- AND POST-IMPOUNDMENT PERIODS

THIS DISSERTATION IS SUBMITTED TO THE UNIVERSITY OF GHANA, LEGON IN PARTIAL FULFILMENT OF THE REQUIREMENT FOR THE AWARD OF PhD FISHERIES SCIENCE DEGREE

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JUNE 2013

DECLARATION

This dissertation is the result of research work undertaken by Elliot Haruna Alhassan in the Department of Marine and Fisheries Sciences, University of Ghana, Legon, under the supervision of Prof. P. K. Ofori-Danson and Dr. F. K. E. Nunoo both of the Department of Marine and Fisheries Sciences, University of Ghana, Legon and Prof. James Samman of the Central University College, Accra.

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DEDICATION

This work is dedicated to my lovely wife, Godslove and son, Ellis for their love and support.

ACKNOWLEDGEMENTS

I wish to express my greatest appreciation to Prof. P. K. Ofori-Danson and Dr. F. K. E. Nunoo both of the Department of Marine and Fisheries Sciences, University of Ghana, Legon and Prof. James Samman of the Central University College, Accra, whose continuous useful advice as supervisors have been a major influence through all stages of the study.

I would also like to express my appreciation to Mr. James Akomeah, Technician of the Department of Marine and Fisheries Sciences, University of Ghana, Legon, Mr. Godwin Amegbe, Mrs. Ruth Amole and Ms. Millicent Ewurama Adu-Boakye, all Senior Technical Officers of CSIR-Water Research Institute, Mr. Stephen Mensah and Matthew Nkoom of the University for Development Studies, Tamale who assisted in the field and laboratory studies.

My sincerest thanks also go to Dr. K. Kwarfo-Apegyah of CSIR-Water Research Institute, Tamale, Mr. Lloyd Allotey and Mr. D. K. Atsu of the Department of Marine and Fisheries Sciences, University of Ghana, Legon for their suggestions during the thesis write-up. Also, the Head, all staff and students of the Department of Marine and Fisheries Sciences, University of Ghana, Legon for their encouragement.

I express my sincerest gratitude to the Ghana Education Trust Fund (GETFund) for supporting this research. My gratitude also goes to the fishers at Bui and Bamboi communities along the Black Volta whose fishing activities enabled me obtain relevant data on the fisheries.

Finally but not the least, I wish to express my sincere gratitude to my parents, Mr. and Mrs. E. A. Alhassan, my siblings; Emelia, Elvis, Erwin and Eva, my lovely wife, Godslove and little son, Ellis for their encouragement and prayers through all stages of the programme.

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ABSTRACT

The hydro-biology of the Black Volta near the Bui dam were studied in relation to fish production as measured by catch per unit effort (CPUE) during the pre- and post-impoundment periods between February 2011 and December 2012. The primary objective was to assess the ecological impacts of the dam on the hydro-biological factors and fish production and provide data for monitoring the hydro-biology and fisheries of the Bui reservoir. Two sampling stations at Bui and Bamboi were selected to represent the upstream and downstream stations respectively.

Physico-chemical parameters such as electrical conductivity, total dissolved solids, colour, dissolved oxygen and sulphates differed significantly ($p < 0.05$) during the pre-impoundment (March – May 2011), immediate post-impoundment (June – December 2011) and late post-impoundment (January – December 2012) periods. Hence, indicating the impact of the impoundment on these parameters.

The Canonical Correspondence Analysis was used to trace temporal phytoplankton and zooplankton community changes, and to examine the relationships between species composition and physico-chemical variables. These variables were subjected to analysis, pair-wise, to identify correlations. The relative abundance of some phytoplankton species such as *Anabaena sp*, *Planktothrix sp* and *Scenedesmus sp* were directly correlated to nitrates.

The phytoplankton groups, namely Bacillariophyceae, Chlorophyceae, Euglenophyceae and total phytoplankton differed significantly ($p < 0.05$) between the pre- and post-impoundment periods. The phytoplankton and zooplankton groups were also significantly ($p < 0.05$) higher in the upstream station than the downstream station indicating the impact of the impoundment on the downstream ecology.

The change from riverine to lacustrine conditions during the formation of the reservoir, led to the immediate reduction in the numbers of a variety of fish families, including Centropomidae, Clarotidae and Distichodontidae which were very sensitive to oxygen depletion. The basic trend in this study was towards the development of a community of fish species which had vegetarian food habits such as *Sarotherodon galilaeus*, *Labeo coubie* and *Labeo senegalensis*.

The mean estimated CPUE for the 2 years (2011 and 2012) was lower (6.23 kg/canoe/day) in the post-wet season than in the dry season (10.86 kg/canoe/day) with a mean of 7.95 kg/canoe/day. Hence, the dry season was the most important season for fish catches, while the post-wet season was the lean season in the study area. The CPUE also differed significantly ($p < 0.05$) between the pre- and post-impoundment periods indicating that the dam had impact on fish production as measured by CPUE. These findings suggest that the impoundment altered not only the river hydro-biology but also its fish production. A multi-linear regression analysis of both water level and chlorophyll *a* content on the CPUE indicated that both partial coefficients were significant ($p < 0.05$) and the best, most prudent model to predict fish production as measured by CPUE was derived as:

CPUE = - (0.456 x water level) + (0.062 x chlorophyll *a*) + 3.363

The coefficient of determination, R^2 of 0.906 of the model explained about 91 % of CPUE variability. This study has provided a checklist of organisms present in the water for subsequent exploitation, conservation and sustainable management of the resources of the Bui reservoir. The study also revealed that the impoundment altered the hydro-biology and fisheries characteristics of the downstream station. Hence, river management strategies should be implemented by fisheries managers to lessen the impact of the dam on the downstream ecology.

CHAPTER ONE

1.0 INTRODUCTION

1.1 Background

The creation of dams is known to be implicated in negative human impacts on river ecosystems as they modify physical environment and regulate flow (Allan, 1996). Dams also have significant consequences for fragmentation of habitats, blocking migration routes of organisms (e.g. fishes) and causing loss of biodiversity (Dudgeon, 2000). According to Junk *et al*. (1989), dams interrupt stream flow and generate hydrological changes along the integrated continuum of river ecosystems that ultimately reflect in the fisheries and hydro-biology of rivers. The most obvious effects of placing dams on rivers result from the formation of new lentic environments upstream from the dam, and tailwater environment downstream of the dam (FAO, 2001). Cumulative effects of dams in catchment basins and tributary streams can significantly block nutrient flow, affecting plankton and fish production in river channels (Hess *et al*., 1982). Tolmazin (1979) for instance, related reduced fish yields in the Sea of Azov to impoundments on the Danube, Dnieper and Dniester rivers in Europe. Despite these negative impacts, the number of damming projects worldwide keeps increasing purposely for the generation of hydroelectric power, irrigation and flood control (World Commission on Dams, WCD, 2000).

The Government of Ghana is contributing to the global increase in dams by creating another dam at the Bui gorge on the Black Volta in addition to the existing Akosombo and Kpong dams for the generation of hydroelectric power. The Bui Hydroelectric Project (BHP) seeks to develop a 400 MW hydroelectric generation plant at Bui (Ampratwum-Mensah, 2011). The development of the BHP, involves the construction of a dam in the Bui gorge of the Black Volta. The dam will create a reservoir covering an area of about 444 km^2 at full supply level (FSL) of 183 meters above sea level (masl) and holding about 12,600 million m^3 of water which will be used for the generation of electricity. Most of the reservoir area will be contained within the nearby Bui National Park. The dam created by the Bui barrage is expected to impact through inundation, the densities of fishes, plankton and macrobenthic fauna. Construction of a dam and consequent impoundment brings a sudden transformation of a lotic environment to a lentic zone. It is anticipated that there would likely be a shift of fish and plankton communities from lotic to lentic components.

Man-made lakes during their early stages of existence are studied because the new community undergoes series of ecological changes immediately after the construction before it gradually approaches a relatively stable state (Visser, 1970; Wetzel, 1983). Thus it is predicted that after the completion of the Bui dam, there would be typical lacustrine, transitional and riverine zones, with their corresponding fish assemblages. This process will trigger a series of changes in the riverine community, which are akin to secondary community succession. Like the aftermath of creation of the Akosombo dam on the Volta River, a number of organisms would perish, few will migrate to more hospitable environs and more hardy ones would adapt themselves to the changed habitats (FAO, 2001).

Communities inhabiting the dam can be classified based on their position in the energy level: producers; consumers and decomposers (Visser, 1970). Primary production by algae in the water sets the level of food available for secondary producers such as zooplankton and fish. When nutrient availability is high, overall production varies with water temperature (Hewett and Johnson, 1987). It is likely that rates of primary production will be altered by climate change and may result in significant consequences for aquatic ecosystems and the human communities that use them. On the one hand, decreases in primary production will reduce food availability at the bottom of the food web, ultimately causing reductions in productivity of fish at the top of the food web while on the other, excessive increases in primary production will produce eutrophic (highly productive) conditions, exemplified by degraded water quality and noxious blue green algal blooms.

The impoundment of a river determines a series of modifications in its physical, chemical and biological characteristics observed from the filling stage until reaching a relative stability. This period is very variable and can last up to twenty years in some tropical reservoirs (Balon and Coche, 1974). Differences in the water temperature, transparency and nutrient level produces changes in the phytoplankton, zooplankton and fish composition (Tundisi, 1993), which can be seen up to 350 km downstream from the Yacyreta dam in Argentina (Petts, 1984). The present study has, therefore its specific value, for assessing the hydro-biology and fish production of the Black Volta in the Bui dam area during the pre- and post-impoundment periods (February 2011 - December 2012). This will therefore generate information on the floral and faunal characteristics that will be altered as a result of river impoundment.

1.2 Problem Statement and Justification

Dams are an inextricable element of our society and are built for a multitude of reasons like irrigation, power generation, drinking water supply and flood control at increasing cost (Collier *et al.,* 1998). The construction of dams also block or delay upstream fish migration and thus contribute to the decline and even the extinction of species that depend on longitudinal movements along the stream continuum during certain phases of their life cycle. Mortality resulting from downstream passage through hydraulic turbines or over spillways can be significant. Reservoirs resulting from construction of dams can however, lead to productive fisheries. This is particularly true for locations where river fisheries contribute little to overall national fish yields (FAO, 2001).

Although there are several reports of beneficial effects of river impoundment and greater economic benefits of having hydro-power schemes compared to cutting and selling of timber from catchments (Pandit, 2000), it is equally well-known that this leads to physico-chemical and hydro-biological changes in the aquatic ecosystem.

The artificial blocking of rivers is known to disrupt upstream and downstream riverine communities and result in the changes in species composition, by obstructing the natural processes of organismal dispersal. Among the aquatic communities, phytoplanktons are of great ecological significance, because they comprise major portion of primary producers in an aquatic ecosystem (Zalewski *et al*., 1999). They are also highly susceptible to altered physical and chemical properties of water and any variations in their community structure get reflected in the corresponding changes in higher components of the food chain (Badoni *et al.*, 1997). For the study of fisheries, the lack of data concerning production at all trophic levels lead to the demand for specific studies of the production problems in tropical waters (Karlman, 1982).

The need for comprehensive pre- and post-impoundment studies of African man-made lakes has been clearly recognized (Adeniji, 1981). Despite these studies, relatively few have been carried out and those that have been carried out tend to be descriptive accounts of the river system without making predictions about future conditions in the lake (Balon, 1978; Bond *et al.*, 1978; Egborge, 1979). Therefore, one-size-fits-all prescriptions cannot substitute for local knowledge in developing prescriptions for dam structure and operation to protect local biodiversity (Power *et al.*, 1996).

In Ghana, most dams such as Vea, Akosombo, Weija, Barekese, Kpong, and Tono have been constructed either for irrigation, hydroelectric power supply, urban and peri-urban water supply, flood control or fisheries by damming the White Volta and Rivers Volta, Densu, Owabi and Tono, respectively. However, a great majority of them have been inadequately monitored and/or studied especially during the pre-impoundment period and formative years of these reservoirs. The Bui reservoir created by damming the Black Volta on the 8th of June, 2011 is subjected to considerable climatic and temperature fluctuations. Effective utilization of large inland water impoundments is based on adequate knowledge of their pre- and post-impoundment status. A pre- and post-impoundment survey of the fauna and flora of any water body is necessary, because it provides a checklist of organisms present in the water for subsequent exploitation, conservation and sustainable management of the resources (Visser, 1970). This therefore necessitated a comprehensive assessment of the water quality, hydro-biology and fisheries in the Bui dam area of the Black Volta.

The present study focuses on the physico-chemical characteristics, hydro-biology and fish production during the pre-impoundment and formative years of the Bui reservoir. This is aimed at generating data for the newly created Bui reservoir after closure of the Bui dam in June 2011, for monitoring and evaluating future changes in the aquatic ecosystem. The study also highlights modifications that occurred after impoundment from lotic to lentic environment that impinge on the hydro-biology and fish production to enhance appropriate management measures to preserve the ecological health and productivity of the ecosystem.

1.3 Hypothesis

The following hypotheses were tested.

Null Hypothesis

The hydro-biological factors such as chlorophyll *a*, phytoplankton, zooplankton and macrobenthic invertebrate densities and fish production of the Black Volta will not change following the impoundment by the Bui dam.

Alternative Hypothesis

The hydro-biological factors such as chlorophyll *a*, phytoplankton, zooplankton and macrobenthic invertebrate densities and fish production of the Black Volta will change following the impoundment by the Bui dam.

1.4 Objectives of the Study

1.4.1 Primary Objective

The primary objective of this study was to assess the ecological impacts of the dam on the hydro-biological factors and fish production during the pre- and post-impoundment periods to provide information on the plankton and fisheries characteristics that may be altered by impoundment which could serve as early warning system for changes in the ecosystem after impoundment.

1.4.2 Specific Objectives

The specific objectives were to:

i. establish the inter-relationships between the physico-chemical factors and plankton abundance of the Black Volta near the Bui dam;

ii. provide information on the influence of selected hydro-biological factors (e.g. chlorophyll *a*, phytoplankton, zooplankton and macrobenthic invertebrates) on the patterns of fish production as well as the relationship between primary productivity as measured by chlorophyll *a* concentration, flood regime as measured by water level and Catch per Unit Effort (CPUE) of the Black Volta near the Bui dam;

iii. develop a multi-linear regression predictive model for monitoring CPUE levels of the Black Volta near the Bui dam;

iv. estimate the existing potential fish yield of the newly created Bui reservoir using the morpho-edaphic index; and,

v. assess the impact of the Bui dam impoundment on phytoplankton, zooplankton and macrobenthic invertebrate densities and changes in fish catches for the effective planning and management of the Black Volta near the Bui dam for socio-economic development.

CHAPTER TWO

2.0 LITERATURE REVIEW

2.1 Impacts of Impoundments on Riverine Ecosystems

Building a dam across a river, and impounding water behind it, may cause profound changes in the limnological regime and biological productivity of the water body (Egborge, 1979; Reynolds, 1997; Ogbeibu and Oribhabor, 2002). The ecological impacts of impounding a river have been dramatic and extensive. Dams can affect the geomorphology of streams that have a large sediment load, as the reservoir traps sediments and release clear water. The resulting downstream geomorphic effects of clear water releases from dams include channel instability and alteration of habitat (Collier *et al*., 1998). Dams can also affect fish by blocking upstream and downstream passage. Elimination or reduction of spawning grounds, or delayed access to the spawning areas has been the most significant effects of physical barriers (FAO, 2001). The blockage of upstream fish movements by dams may have serious impacts on species whose life history includes migrations for various purposes. For fish trying to move upstream, a dam can pose an impassable barrier, and fish moving downstream are at high risk of being entrained in the turbine intake and injured or killed during downstream passage (FAO, 2001).

Flow variability controls physico-chemical and hydro-biological phenomena in a river. Impoundments, particularly those of storage-release nature, reduce the natural variability of flow, although hydro-power impoundments may increase diel variability. Some hydro-power dams have underground power stations, resulting in the desiccation of a section of the river downstream of the dam. Resident fishes experiencing flow alterations may be affected for great distances downstream. Flow modifications affect water quality, water depth and velocity, substrate composition, food production and transport, stimuli for migration and spawning, survival of eggs, and eventually fish species composition (Petts, 1984).

The physico-chemical and hydro-biological attributes of the downstream ecosystem are dictated by whether the release of water is drawn from hypolimnion, epilimnion, or from multi-levels (Cassidy, 1989). Depth of withdrawals affects water temperature, dissolved gases, nutrients, turbidity, biotic assemblage and diversity. Hypolimnetic releases are relatively cold, oxygen depleted and nutrient-rich. Epilemnetic releases on the other hand are typically less disruptive as temperature and water quality characteristics are more suitable to the downstream biota (Crisp, 1977). Reducing water flow therefore changes the biota and landscape downstream (Simons, 1979).

Reductions in sediment load caused by impoundments prompt the river downstream to recapture its load by eroding the downstream channel and banks. Release patterns also affect downstream biota in several ways (Welcomme, 1985). Large flow variations may adversely affect downstream productivity by impacting spawning and disrupting benthic populations. Lower nutrient concentrations in releases can result in lower primary production in the tail water. Conversely, nutrient-rich releases stimulate increases and lavish the development of algae and macrophytes. The benthic community may shift towards grazers and collectors and experience loss of diversity as organisms depending on thermal cues for spawning, hatching and emergence will dwindle (FAO, 2001).

The main effect of impoundment on fish is a shift in species composition and abundance, with extreme proliferation of some species and reduction, or even elimination of others (Agostinho *et al.*, 1999). According to FAO (2001), the level of impact on the biological diversity is generally influenced by the characteristics of the local biota (e.g. reproductive strategies and migratory patterns), characteristics of the reservoir (e.g. morphology and hydrology), design and operational characteristics of the dam, and characteristics and uses of the watershed (e.g. agriculture, mining and urbanization). Wetzel (1990) noted that the response of fish assemblages to impoundment is a chaotic succession of reactions marked by a reduction in the established interdependence among species, and a lower biotic stability as well as natural succession processes.

Impoundments also reduces the cyclic nature of the riverine environment by restraining natural hydrologic cycles, and may introduce non-cyclic perturbations related to operations of the dam, exacerbating the instability induced by the foreign environment (FAO, 2001). The biotic community then responds by reducing species diversity and becoming gradually simpler, a response evident during the first few years after impoundment. Paller and Gladden (1992) observed that these responses are aggravated by catalysts such as unsuitable water temperature, low dissolved oxygen, low habitat diversity, inadequate or few spawning sites, absence of shelter for prey, and exclusion through inter-specific interactions.

Changes in habitat caused by impoundments often limit the lotic fish fauna to the upper, unimpounded reaches of streams. This is because the reservoir acts as a barrier for dispersal, preventing upstream or downstream passage, these populations often remained isolated. These small and fragmented populations may survive for many years in a river basin, but much of the original genetic variation may be lost (Wilson, 1988). Lack of passage also restricts the ability of fish to decolonize suitable habitat following catastrophic events. Thus, impoundments have fragmented the home ranges of certain species, causing local extinctions.

2.2 Hydro-biology of Rivers

The contribution made by phytoplankton to primary production within rivers is generally regarded to be low when compared to other types of aquatic ecosystems. However, phytoplankton is present in rivers and contributes to the nutrient balance and to the trophic requirements of some of the fish species. The major factors determining the presence and abundance of phytoplankton in rivers are temperature, velocity of the current, availability of nutrients and light (Welcomme, 1985).

In tropical rivers, temperature plays a much diminished role in phytoplankton abundance and the greatest densities of phytoplankton coincide with low water temperature (Welcomme, 1985). Also, phytoplankton are sensitive to velocity and turbulence of flow in rivers as the rapid currents and mechanical stresses of rapids and waterfalls inhibit the development of new plankton and rapidly suppress any existing organisms discharged from any associated lentic waters. In the Nile River at the Gebel Aulia dam in Sudan, the dam slowed the Nile current and produces a rapid increase in phytoplankton concentration. When the dam was opened, the flow was faster and the plankton concentration dropped and hence demonstrated a strong correlation between phytoplankton and current velocity in the river (Prowse and Talling, 1958). According to Rzoska and Talling (1966), phytoplankton was more abundant in the backwaters of the Nile Sudd in south Sudan than in the main channels. With specific composition of phytoplankton, blue-green algae such as *Anabaena sp.* and *Lyngbya sp.* dominated in the standing waters whereas in the river, the sparse flora consisted mainly of diatoms especially *Melosira sp.* (Rzoska, 1974).

Nutrient availability also plays an important role in the determination of the abundance of phytoplankton in rivers. In clear white waters of neutral pH, diatoms and green algae were more abundant and in eutrophicated waters or those with high pH, blue-green algae were the most common (Welcomme, 1985). In the Nile River in north-eastern Africa, there was a negative correlation between phytoplankton abundance and nitrate concentration (Talling, 1957).

The phytoplankton of the Dniester River in Ukraine underwent qualitative and quantitative changes after the closure of the Dubassery reservoir with increases in Cyanophyceae downstream of the dam. The phytoplankton biomass varied between 0.66 and 0.85 gm^{-3} before the dam was built but rose to about 4 gm^{-3} in the river after its construction (Welcomme, 1985). In the Mississipi River in North America, phytoplankton abundance as represented by chlorophyll *a* concentrations was less in the main channels of the river than in the backwaters or in the river lake where the current was slow (Baker and Kromerbaker, 1979). In the Missouri river in North America however, low planktonic densities (0.067 cells/ml) were associated with high current, turbidity and a lack of subsidiary floodplain water bodies to feed into the main stream (Berner, 1951). Phytoplankton abundance in rivers is also associated with seasonal differences in flow. Densities usually reach a peak in the dry season and diminish in the floods unless otherwise influenced by temperature (Welcomme, 1985). In the Kafue River in Zambia, phytoplankton densities were found to be less during the floods and dense blooms occurring in the river at Nampongwe between August and November when the floods had receded (Carey, 1971).

Zooplankton abundance in rivers has been attributed to differences in flow, turbidity, dissolved oxygen concentration, conductivity, degree of vegetation and seasons (Welcomme, 1985). Under normal flow regimes, only low densities of zooplankton are present in the main channel of rivers. Distribution of zooplankton varies from place to place and year to year due to the dynamic nature of aquatic systems (FAO, 2006). Zooplankton species succession and spatial distribution result from differences in ecological tolerance to abiotic and biotic factors (Marneffe *et al*., 1998), yet, bio-indicator approaches, using the responses of organisms to evaluate trophic state, have often been neglected in favour of chemical and physical techniques. Despite the considerable potential of zooplankton as effective indicators of environmental change and their fundamental importance in the transfer of energy and nutrient cycling in aquatic ecosystems, the zooplanktonic communities have not been widely used as ecosystem indicators (Stemberger and Lazorchak, 1994).

Macrobenthic invertebrates are those organisms that live on or inside the deposit at the bottom of a water body (Barnes and Hughes, 1988; Idowu and Ugwumba, 2005). They are ubiquitous and diverse group of long lived species that react strongly and often predictably to human influences in aquatic ecosystem. In addition they are sedentary, therefore body burdens reflects local conditions, allowing detection of a variety of perturbations in a range of aquatic habitats (Rosenberg and Resh, 1993). Macrobenthic invertebrates are an important and integral part of any aquatic ecosystem as they form the basis of the trophic level and any negative effects caused by pollution in the community structure can in turn affect trophic relationships. According to Carlisle *et al*. (2007), macro invertebrate populations in streams and rivers can assist in the assessment of the overall health of the stream. These organisms play a vital role in the circulation and recirculation of nutrients in aquatic ecosystems.

2.3 Overview of the Physico-chemical conditions of the Black Volta

The physico-chemical factors measured by Petr (1970) from April 1965 to April 1967 in the Black Volta were temperature ranging from 29.8 ºC – 31.7 ºC; pH ranging from 7.7 – 8.6; and dissolved oxygen ranging from 6.4 mgl^{-1} – 8.1 mgl^{-1}. Welcomme (1972) recorded conductivity values of 41 - 124 µS/cm and pH values of 6.5 – 7.3 in the Black Volta.

The Snowy Mountains Engineering Corporation feasibility study conducted in the 1970s made an assessment of water quality in the Black Volta based on spot measurements taken in May 1976 at Bui, and reached the conclusion that the river was a suitable source that can be treated for drinking and other purposes and that slightly acidic to slightly alkaline conditions exist in the Black Volta with sampled values ranging from 6.6 to 7.2 (Environmental Resources Management, ERM, 2007). The alkalinity values also suggest that the waters of the Black Volta were buffered and thus not easily susceptible to changes in pH.

Water quality parameters in the Black Volta including surface water temperature, pH, turbidity, nutrient levels, and dissolved oxygen also fulfilled the biological requirements of aquatic flora and fauna (Gordon *et al*., 2003). The waters were generally turbid, with high levels of total suspended solids; however, such levels were not so high as to adversely affect the presence or abundance of aquatic flora and fauna.

2.4 Overview of Macro-invertebrates of the Black Volta

During April 1965 to April 1967, Petr (1970) identified the following taxa of invertebrates in the Black Volta: Coleoptera; Diptera; Ephemeroptera; Lepidoptera; Odonata; Oligochaeta; Plecoptera; and Tricoptera. During the hydro-biological monitoring of the Black Volta in relation to Onchocerciasis Control Programme (OCP) larviciding of rivers for control of the blackfly, *Similium damnosum* complex, a vector of river blindness or Onchocerciasis, Samman and Amakye (1986) recorded Baetidae, Caenidae, Chironomidae and Hydropsychidae as the predominant groups of macro-invertebrates.

Surveys in the Black Volta in the Bui dam area in 2001 focused on nymphs or larvae (pre-adult stages) belonging to the insect orders Ephemeroptera, Diptera, and Odonata (Gordon *et al*., 2003). The most prevalent macro-invertebrates observed during the survey belonged to the order Ephemeroptera (mayflies), followed by Diptera (Chironomid midges), and Odonata (dragonflies and damselflies). Gordon *et al*. (2003) did not however, record any macro-invertebrates from the order Trichoptera (caddisflies) neither did they identify organisms beyond the order level (to family or species).

2.5 Overview of Fish Community Structure of the Black Volta

The key fish species identified during October 1989 to September 1990 sampling period in the Black Volta at Bamboi in relation to the OCP were *Petrocephalus bovei*, *Hydrocynus forskalii*, *Brycinus nurse*, *Brycinus leuciscus*, *Labeo senegalensis*, *Schilbe mystus*, *Synodontis ocellifer*, *Synodontis gambiensis*, and *Eutropius niloticus* (Samman and Abban, 1991). However, during the October, 1993 to September, 1994 sampling period, no catches were made for *Petrocephalus bovei* in the river (Dankwa *et al*., 1995).

A total number of 96 fish species were identified in the Ghana portion of the Black Volta (Vanden Bossche and Bernacsek, 1990). Another fish survey conducted in the Black Volta within the Bui National Park identified 46 species of fish from 17 families (Bennett and Basuglo, 1998). A Mormyrid fish, probably *Gnathonemus petersi*, was reported by local fisherman to have disappeared from the river in the mid-1990s (Bennett and Basuglo, 1998). Surveys in 2001 and 2002 at the Bui dam site, the possible reservoir inundation area, recorded a total of 49 species of fish belonging to 26 genera and 14 families (Gordon *et al*., 2003). The fish species composition in the samples taken during the wet and dry seasons differed markedly, possibly due to habitat preferences associated with water levels in the river. Small-sized fish (species caught in the 12.5-15 mm mesh sizes) dominated the catch at all stations in both seasons (Gordon *et al*., 2003). None of the species encountered during the surveys were of conservation concern: all were known to occur elsewhere in the Black Volta system, including Lake Volta (Gordon *et al*., 2003).

2.6 The Bui dam Project

The idea to develop the Bui dam on the Black Volta at the Bui gorge was conceived in the colonial days of the 1920s when one Albert Kitson, a British-Australian geologist and naturalist on his assignment with the Geological Survey Department of Ghana, visited the site (Ampratwum-Mensah, 2011). The development of a hydropower scheme on the Black Volta at the Bui gorge had been the subject of many studies; namely, detailed studies by J. S. Zhuk Hydroproject of the USSR in 1966, a feasibility study by Snowy Mountains Eng. Corp (SMEC) of Austrailia in 1976 and another feasibility study by Coyne et Bellier of France in 1995. The proposed 400 MW Bui hydropower scheme was considered to be the most technically and economically attractive hydropower site in Ghana after the Akosombo and Kpong hydropower plants. The feasibility study of 1995 was subsequently updated by Coyne et Bellier in October, 2006 to enable commencement of the project.

The Bui Power Authority Act (Act 740) was enacted by the parliament of Ghana and assented to by the President of the Republic of Ghana in July, 2007 to establish an Authority known as the Bui Power Authority (BPA) which was to plan, execute and manage the Bui Hydroelectric Project (BHP). The BHP which is currently being implemented was designed primarily for hydropower generation (Ampratwum-Mensah, 2011). It however also includes the development of an irrigation scheme for agriculture development and presents an opportunity for enhanced ecotourism and fisheries. The morphometric characteristics of the Bui reservoir when completed are: full supply level (FSL) of 183.0 m; reservoir area at FSL of 444 km^2; storage volume at FSL of 12.57×10^9 m^3; minimum operating level of 168.0 m; and active storage of 7.72×10^9 m^3.

Prior to the construction of the Bui dam, an Environmental and Social Impact Assessment (ESIA) was carried out (ERM, 2007).

The key findings of the ESIA were outlined under the following headings:

i. impacts on land and land use;
ii. impacts on the water environment;
iii. impacts on ecology and biodiversity;
iv. social, health and economic impacts;
v. impacts on archaeology and cultural heritage; and
vi. other issues.

According to the ESIA, with full and proper implementation of the measures identified in the Environmental and Social Management Plan and Resettlement Planning Framework, the construction and operation of the Bui dam project is likely not to cause unacceptable impacts on the environment or the communities of the surrounding area. Environmental assessment has multi-fold benefits in addressing impacts that will or likely to happen to affected communities and natural resources. According to Ogbeide *et al*. (2003), if ESIA reports are prepared based on correct principle and practice, they provide ample opportunities to make the development project environment-friendly and sustainable.

2.7 Anticipated Impacts of the Bui dam on Biodiversity

The biodiversity of the Black Volta includes the living aquatic resources such as fishes (both shell and fin fishes) and some isolated populations of primates and other large mammals such as the hippopotamus (*Hippopotamus amphibious*) population and plant vegetation that depend on the river at Bui. The major anticipated environmental impact that is expected to affect the flora and fauna in the basin would be the effect of lacusterization on the Black Volta, as a result of the construction of the Bui dam in June 2011.

Construction of the Bui dam and associated structures, and the creation of the reservoir, will cause both loss and alteration of habitats, with resulting impacts on ecology and biodiversity. Filling and operation of the Bui dam will ultimately create 440 km^2 of new lacustrine habitat with a maximum depth of 29 m, replacing approximately 40 km of riverine habitat along the Black Volta (Ampratwum-Mensah, 2011). Prior to filing, the main part of the reach will retain its current riverine characteristics, with the exception of the temporary diversion channel. During this period the biological communities in the reservoir will begin to be exposed to lacustrine characteristics. The initial change will include declines in mayflies and certain fish species that prefer running water and coarse substrate. Species that prefer shallow habitat are likely to colonize the periphery of the reservoir, and others that require running water may disappear or persist as relict populations in the headwaters of the reservoir (ERM, 2007).

Sediment retention and subsequent deposition within the reservoir will cause most of the coarse substrate, rocky outcrop and other elements of the riverbed to progressively disappear under layers of silt transported from upstream. This will alter the fish and macro-invertebrate composition with a reduction in the abundance of riverine fishes such as Mormyridae species and an increase in benthic and algal feeders, such as species of Bagridae and Cichlidae (Gordon *et al*., 2003).

It is also expected that the Bui dam will block upstream movements of potadromous fish (species of fish that requires movement through freshwater system to complete life cycle) such as *Alestes sp.* and *Labeo sp.*, disrupting spawning activities and ultimately leading to a possible decrease in gene flow and genetic variation between isolated populations in the river (ERM, 2007). The entire life cycle of this potadromous fish species occurs within fresh waters of a river system (Northcote, 1998). In addition, the blockage and altered downstream flows after completion of the Bui dam could affect the fish communities in Lake Volta. During May and September, certain potadromous fish species (e.g. of the genera *Hydrocynus, Labeo, Chrysichthys, Bagrus* and *Synodontis*) migrate from Lake Volta upstream to spawn in the Black Volta tributaries (Samman *et al*., 1992). If suitable spawning sites for these species do not occur between Lake Volta and the Bui dam site, then these species could be lost from this part of the Black Volta system.

Inundation of the reservoir will result in drowning of some terrestrial fauna unable to escape from flooded forest, grassland, and savannah woodland habitats. The reservoir will fill slowly which should avoid large scale direct loss, as most animals will be able to move to higher ground as the water level rises. Mortality due to drowning is likely to be most prevalent in ground dwelling and feeding mammals with limited mobility. Certain primates, such as black and white colobus monkeys (*Colobus guereza*), live in groups and spend over 50 % of their time on tall trees (approximately 30 m high) to avoid aerial and ground predation (Von Hippel, 1998). As these primates mainly reside in the upper canopies of trees, they are at risk of being stranded in trees by the rising water (Onderdonk and Chapman, 2000).

Displacement of territorial and gregarious animals from the reservoir area as it starts to fill, for example ungulates such as kob (*Kobus kob*) and waterbuck (*Kobus ellipsiprymnus*), could cause short term disorganization of the social structure of herds. This could alter reproductive behaviour, and cause aggression and increased stress, leading to injury, disease or mortality in some young, old or otherwise vulnerable individuals (Estes, 1991). The magnitude of the effect will be high initially, but will decrease over time as the animals re-orient to the new habitat conditions and herds stabilize.

According to Ofori-Amanfo (2005), inundation of the reservoir will eliminate existing dry season *Hippopotamus amphibious* pool habitats within the Bui National Park (BNP), displacing the hippos that use these pools for resting. During the transition period when the reservoir is filling and shoreline vegetation communities are not yet established, hippopotamus which are endemic in the Black Volta, will move from these traditional pools and feeding areas in search of suitable resting and foraging areas. They will therefore be vulnerable to hunting during this period. Once the reservoir is full and vegetation communities are established along the shoreline, hippopotamus will benefit from the increased area of littoral habitat provided by the reservoir (Lewison and Carter, 2004).

It is also anticipated that the construction of the Bui dam will impact on the vegetation communities. According to Hill and Curran (2003), vegetation loss and disturbance will alter the area, shape, and continuity of remaining vegetation patches within the landscape, altering the floral and faunal species composition and possibly rendering the fragmented ecosystems unable to support the species assemblages found in undisturbed ecosystems.

The creation of the reservoir will raise the water table in the area surrounding it, and in the area immediately adjacent to the river channel downstream of the dam. Increased soil moisture throughout the year will benefit vegetation, particularly during the dry season, resulting in more dense and possibly more diverse vegetation communities than currently exist around the reservoir margin and along the riverbank for a short distance downstream (Hill and Curran, 2003; ERM, 2007). The ultimate plant species composition will also depend on species-specific ability to tolerate anoxic conditions. The higher water table could saturate the root zones of plants growing in the most affected areas, depleting the oxygen available for aerobic respiration and causing plants to switch to anaerobic (oxygen-free) metabolism. This could cause mortality in plants not adapted to flood conditions, such as savannah shrubs and other upland or drought-tolerant species that are intolerant to root water-logging (Drew, 1997).

Flood-tolerant plants that are currently prevalent in the floodplain at Bui are *Panicum parvifolium*, *Hyparrhenia cyanescens*, and *Acacia sp*. These species will dominate the affected zone in the periphery of the reservoir and floodplain habitats downstream of the dam. These plants have a number of adaptations to help them cope with anoxic conditions, such as increasing oxygen supply to the roots and adjusting root growth to the level of the water table (Le Maitre *et* *al*., 1999). The extent to which plant species composition will be altered by water logging is uncertain, but will only be likely in localized, low-lying areas (Hill and Curran, 2003).

2.8 Impacts of Impoundments on people's Livelihoods

The impoundments after damming the rivers adversely impact the finances of fishing communities that depend on fishing for their livelihood. Because dams tend to be constructed to enhance socio-economic development activities, they tend to attract people and industry. Subsequently, river ecosystems containing dams must contend with secondary environmental pressures such as increases in pollution as well as increase exploitation and extraction of their resources (FAO, 2001).

The impact of dams on people's livelihood, health, social systems and culture are not easily quantified and often not considered in the analysis of the benefits of dams (Ogbeide *et al*., 2003). The direct benefits dams provide to people are typically reduced to monetary figures for economic analysis and are not often recorded in human terms. The numerous dams built round the world have played important role in helping communities and economies harness water resources for several uses. An estimated 30 - 40 % of irrigated land worldwide now relies on dams and that dams generate 19 % of world electricity (WCD, 2000).

The construction of dams has led to displacements of about 40 – 80 million people worldwide (WCD, 2000). In China alone, dams have displaced 10.2 million people between 1950 and 1990 (Asian Development Bank, 1999). Among projects involving displacements funded by the World Bank, dams account for 63 % (World Bank, 1996). Inestimable and almost irreversible environmental degradations and colossal loss of lives and property due either to the outright failure of dams, improper monitoring and management of flood regimes in the dams, remain the most devastating cause of flood disasters in Nigeria (Ogbeide *et al*., 2003).

Environmental change and social disruption resulting from the construction and operation of dams and the associated infrastructure developments such as irrigation schemes can have significant adverse health outcomes for local populations and downstream communities. Among the resettled, access to drinking water, health services and ability to cope with new social and physical environment determines health conditions. Numerous vector-borne diseases are associated with reservoir development in tropical areas. According to Ogbeibu (2002), the adverse effects on the health of local people living around dams due to environmental changes such as increased breeding of mosquitoes and other insect vectors from dams, spread of water-borne diseases and resettlement of people in river basins are rampart.

Dams have significant effects on cultural heritage through loss of local cultural resources, temples, shrines and sacred elements of the landscape, artifacts and buildings. Dams can also cause loss of or damage of cultural heritage through land reclamation and irrigation project. During the construction of the Inanda dam in South Africa, remains of human bodies buried under the reservoir site were exhumed and all buried in one hole which profoundly disturbed local communities (Gwala, 2000). According to Kinahan (2000), the risk of submerging ancestral graves was the main reason the Himba people in Namibia opposed the construction of the Epupa dam.

In terms of the social impacts of dams, WCD (2000) found that the negative effects were frequently neither adequately assessed nor accounted for. According to the Commission, the range of these impacts is substantial, including on the lives, livelihoods and health of the affected communities dependent on the riverine environment:

i. some 40 - 80 million people have been physically displaced by dams worldwide;
ii. millions of people living downstream from dams, particularly those reliant on natural floodplain function and fisheries have also suffered serious harm to their livelihoods and the future productivity of their resources has been put at risk;
iii. many of the displaced were not recognized (or enumerated) as such, and therefore were not resettled or compensated;
iv. where compensation was provided it was often inadequate, and where the physically displaced were enumerated, many were not included in resettlement programmes;
v. those who were resettled rarely had their livelihoods restored, as resettlement programmes have focused on physical relocation rather than the economic and social development of the displaced;
vi. the larger the magnitude of displacement, the less likely it is that even the livelihoods of affected communities can be restored; and
vii. impacts on downstream livelihoods were, in many cases, not adequately assessed or addressed in the planning and design of dams.

CHAPTER THREE

3.0 MATERIALS AND METHODS

3.1 Study Area

The study was conducted on the Bui dam section of the Black Volta (Figure 1). The study area stretched from the Bui reservoir (upstream) to Bamboi (downstream) within latitudes 8º 09´ - 8º 16´ N and longitudes 2º 01´ - 2º 15´ W and a distance of about 37.5 km. This formed part of the Black Volta basin primarily located in north-western Ghana approximately 150 km upstream of Lake Volta. The basin covers portions of the Upper, Northern and Brong Ahafo Regions of Ghana. The basin has a total catchment area of 142,056 km^2 including areas outside Ghana. The portion of the Black Volta in Ghana is estimated to be 650 km in length with a catchment area of 35,105 km^2 (Vanden Bossche and Bernacsek, 1990).

3.1.1 Topography and Relief of the Black Volta

There is considerable variation in local relief of the Black Volta basin. The northern areas ranged between 300 – 600 meters above sea level (masl). The Basin is gently undulating from the north to the south. Most part of the Black Volta falls under the Savanna zone which is undulating with gentle slopes that promotes overland flow. The low relief is also a cause for the poor surface drainage with a consequent flooding which characterize the desertification-prone areas during the wet season (Agorsah, 2004).

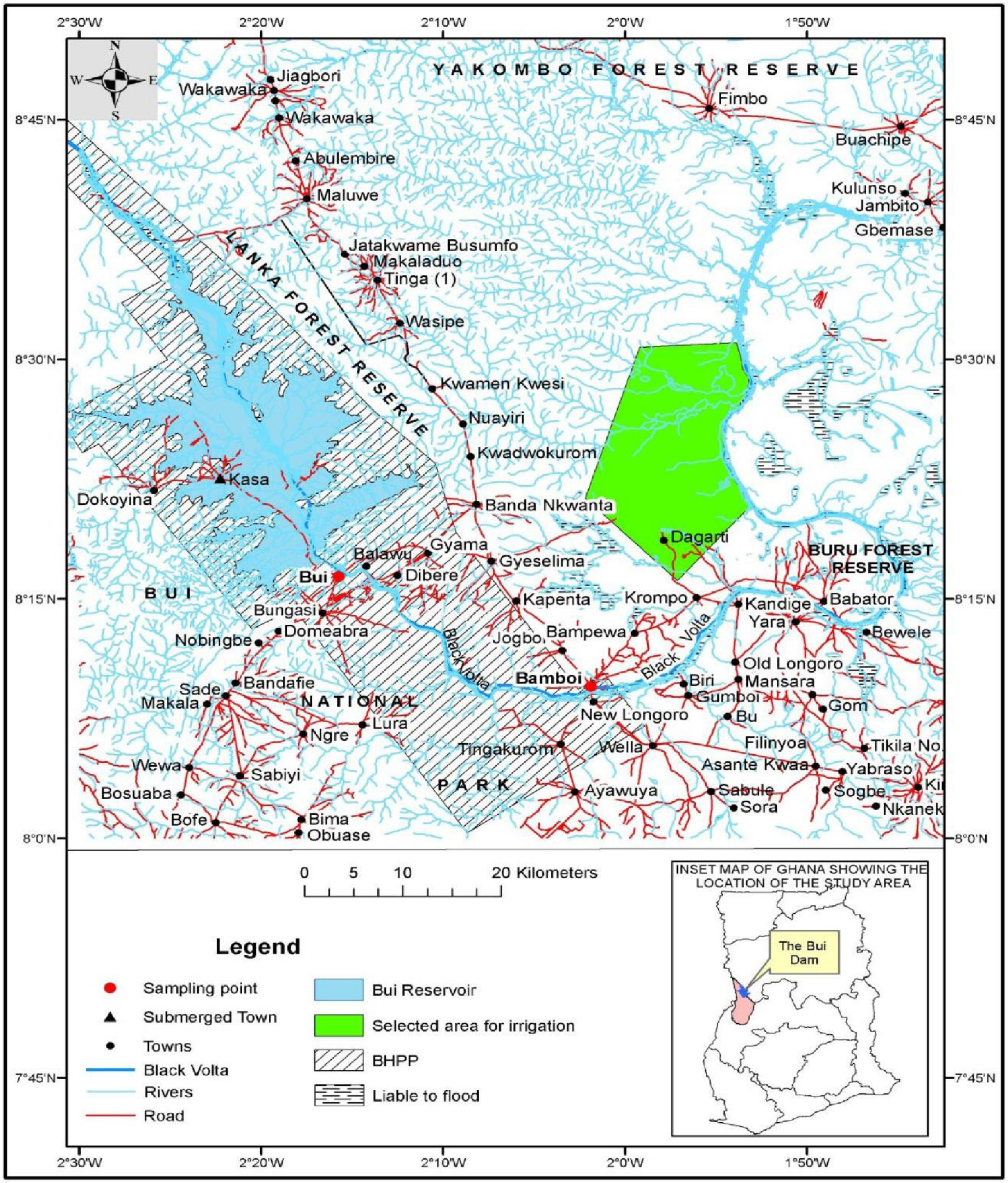


Figure 1: Map of Study area showing sampling stations (after Ofori-Danson *et al*., 2012)

3.1.2 Climate of the Black Volta

Two regimes of wet and dry seasons, occasioned by the northward and southward movement of the International-Tropical Convergence Zone (ITCZ) constitute the major climatic markers. This has created a remarkably clear east-west divide of climatic and vegetation zones that in turn have had significant influence on human activities, mainly agriculture. According to Abban *et al*. (2000), four hydrological seasons can be discerned within the study area: dry season (January – March); pre-wet season (April – June); wet season (July – September); and post-wet season (October – December).

In January and February when Ghana lies entirely on the path of the harmattan air masses, the weather becomes dry and warm during the day and cool at night. In the northern portion of the Black Volta, the daily high temperature reaches about 29 ^{o}C and the night low temperature is about 21 ^{o}C (Agorsah, 2004). The climatic condition however changes in June and July when the weather is characterized by thunderstorms and heavy precipitation usually lasting fifteen to thirty minutes.

3.1.3 Communities and Demographic characteristics of the Black Volta

The major communities along the Black Volta in Ghana are: Hamele, Lawra, Nadowli, Babile and Wa, all in the Upper West Region, Sawla, Bole, Bamboi and Damango in the Northern Region, and Bui, Banda, Bungasi, Nsawkwa, Wenchi and Tainso in Brong-Ahafo Region (Agorsah, 2004). Some major ethnic groups along the Basin are Lobi, Dagarti, Wala, Moh, Gonja, Senufa, Vagala, Sisala, Banda, Bono and Gyaman. The inhabitants are predominantly peasant farmers, fishermen, fish processors and others engage in petty trading.

3.1.4 Agriculture and Land Use of the Black Volta

About 70 % of the population was engaged in agriculture in 1995 which was projected to decline to about 59 % in 2020 (Nabila, 1997). Agricultural practices in the Black Volta vary from the transitional zone to the interior savanna. Agriculture is dominated by smallholder farms characterized by low input technologies and low yields.

The major annual crops are classified as cereals, root crops, pulses and nuts and vegetables. The major cereals are maize, millet and sorghum. Maize is grown throughout the country with about 63 % of food farmers involved in its cultivation. However, millet and sorghum are mainly grown in the savannah zone. Cassava is grown almost everywhere in the basin; however, in small scale. The cultivation of yam is concentrated mainly in the transition and Guinea Savannah zones. Groundnuts and bambara beans are largely cultivated in the Upper West and Northern Regions (Asamoah and Kotartsi, 1997).

3.2 Sampling Design

In order to provide an all-year round picture of the hydro-biology and fish production of the study area, a three-level stratified random sampling approach was adopted. The first stratum which was defined by the four designated hydrological seasons in the study area was referred to as follows: dry season (January to March); pre-wet season (April to June); wet season (July to September); and post-wet season (October to December) (Abban *et al*., 2000). The second stratum which was defined by the three impoundment periods in the study area was referred to as follows: pre-impoundment (March to May 2011); immediate post-impoundment (June to December 2011); and late post-impoundment (January to December 2012). The third stratum, on the other hand which was defined to improve sampling for accuracy was as follows: above the dam site or reservoir area with sampling station at Bui (old town currently submerged); and below the dam site area with sampling station at Bamboi.

3.3 Field Data Collection

3.3.1 Measurement of changes in Water levels

The water level of the river was measured monthly by taking readings in metres from permanent calibrated poles mounted in the river at Bamboi Bridge. The poles were established by the Ghana Highway Authority.

3.3.2 Measurement of Physico-chemical Parameters

Monthly readings of temperature, pH, dissolved oxygen, conductivity, colour and TDS were taken in the field using the Water Quality Checker or probe (Fresenius *et al.* 1998). The probe was immersed in the water at each of the two sampling stations and the mode for each of the above parameters pressed and the figure registered on the screen recorded. Three readings, about 30.0 cm below the water surface were taken at each sampling station for each of the above parameters and the average found and recorded. Water samples were also taken from the same depth at each station with a 2.0-litre Hydro-Bios Kiel TP water sampler to the laboratory for nutrient content analysis.

3.3.3 Measurement of Hydro-biological factors

The hydro-biological factors measured during the study period were chlorophyll *a* concentration, phytoplankton, zooplankton and macrobenthic invertebrates' densities.

3.3.3.1 Chlorophyll a concentration as a measure of Primary Productivity

Triplicate surface and bottom water samples were collected monthly between 0600 and 0700 GMT using a 2.0 litre Hydro-Bios Kiel TP water sampler. The water samples were kept in 1 litre plastic sampling bottles at each of the two sampling stations. The samples were then taken to the CSIR-Water Research Institute Water Quality Laboratory in Tamale for analysis of chlorophyll *a* content as a function of primary productivity. The data obtained were used in a simple linear regression model for prediction of fish production of the Black Volta in the study area.

3.3.3.2 Collection of Phytoplankton and Zooplankton Samples

The phytoplankton samples were collected monthly for 22 months (March, 2011 – December, 2012) between 0600 and 0700 GMT. The samples were obtained by towing a 0.5 m diameter phytoplankton net (35 μm mesh size and 0.25 m^2 mouth surface area) from a non-motorised canoe through a distance of about 100 m against the current from downstream to upstream. The phytoplankton samples were preserved with Lugol solution in 50 ml sampling bottles.

The zooplankton samples were also collected monthly using a zooplankton net of 55 µm mesh size and 0.25 m^2 mouth surface area for 22 months (March, 2011 – December, 2012) between 0600 and 0700 GMT from a non-motorised canoe (Plate A) through a distance of 100 m. Collected zooplankton samples were preserved in 4 % formaldehyde solution in 50 ml sampling bottles.

3.3.3.3 Collection of Benthic Samples

Benthic samples were collected using an Ekman grab monthly for 22 months (March, 2011 – December, 2012). At each sampling station, the grab was lowered into the bottom to take 3 replicates of sediments. The benthic samples were back-washed through a sieve of 1 mm x 1 mm mesh size to separate the benthos from mud. The washed sediment with macro benthos were poured into a labeled plastic container and preserved with 4 % formaldehyde solution buffered with Borax (Sodium borate).

Plate A: Plankton net used throughout the study for collection of zooplankton samples at Bamboi using local canoe (picture taken, November 2012)

3.3.4 Fish catch Assessment

3.3.4.1 Commercial gill net catches

Monthly fish sampling surveys were undertaken during the four designated hydrological seasons from February, 2011 to December, 2012 at the Bui and Bamboi sampling stations. During the survey, a sampling unit was considered as a fisher utilizing a canoe normally with gill net. The gears used by the fishermen consisted of a mixed battery of multi-filament gill nets of mesh sizes of between 20 and 80 mm. At each sampling station, the catch from a sampling unit were then weighed in grams and the number of fish counted for individual species caught. The fish caught were identified individually using identification keys by Dankwa *et al*. (1999) and Paugy *et al*. (2003).

3.3.4.2 Experimental gill net catches

Experimental fishing was also undertaken to ascertain whether experimental catches could be a good indicator of trends in commercial fish catches (Vanderpuye, 1984). The gears used consisted of a mixed battery of multi-filament gill nets of mesh sizes 15 mm, 17.5 mm, 20 mm, 22.5 mm, 25 mm, 30 mm and 40 mm (lateral stretched). All nets were set in the evening (1700 – 1800 GMT) by a hired local fisher using a canoe. The nets were retrieved the following morning between 0600 and 0800 GMT. The fish caught were identified individually using taxonomic identification keys by Dankwa *et al*. (1999) and Paugy *et al*. (2003). The fish was later weighed in grams and the number of fish counted for individual species caught.

3.4 Laboratory based data

3.4.1 Nutrient Content Analysis

The nutrients analysed were phosphates, nitrates and sulphates using a Hach DR2010 direct-reading spectrophotometer and pre-package reagents within 24 hours after sampling.

For phosphates analysis, about 25 ml of the water taken to the laboratory was measured into a reaction bottle and Phos Ver 3 reagent added and swirled. The sample was then allowed to stand for 2 minutes. Blue colour indicated the presence of phosphates in the sample. The concentration of phosphate was recorded in mgl^{-1} on a spectrophotometer at 890 nm to two decimal places following the methods described in APHA (1998).

For nitrates analysis, another 25 ml of water was measured into a reaction bottle and Nitra Ver 5 reagent added and shaken for 1 minute. The sample was then allowed to stand for 5 minutes. Brown colour indicated the presence of nitrates in the sample. The concentration of nitrates was recorded in mgl^{-1} to two decimal places on the spectrophotometer at 400 nm following the methods described in APHA (1998).

For sulphates analysis, about 10 ml of water sample was measured into a 25 ml erlenmeyer flask. Exactly 0.5 ml conditioning reagent was added and mixed by stirring. A spoonful of barium chloride crystals was then added while still stirring and timing immediately for 60 seconds at a constant speed. After stirring, the absorbance rate was measured at 420 nm on the spectrophotometer within 5 minutes. The concentration of sulphate was read directly from the calibration curve and the results expressed in mgl^{-1} to two decimal places.

3.4.2 Determination of Chlorophyll *a* concentration and Phytoplankton densities as a measure of Primary Productivity

For analysis of chlorophyll *a* concentration, 500 ml sub-sample was taken from the water sample. About 3-5 drops of aqueous solution of 50 % magnesium carbonate was added to avoid the degradation of chlorophyll. The sample was then centrifuged at 2000 rpm for 15 minutes.

The filtrate was transferred into dark bottle and capped tightly. It was then placed in a refrigerator for 14 hours to allow complete extraction of chlorophyll. The content of bottle was again centrifuged at 3000 rpm for about 15 minutes. The supernatant was transferred to a volumetric flask of 10 ml and the volume of the content was raised to 10 ml by adding 90 % acetone. The optical density (OD) of the extract was recorded in mgl^{-1} on a spectrophotometer at 630 nm, 663 nm and 750 nm following the methods described in APHA (1998).

For enumeration of algal taxa, the description by Lund *et al.* (1958) was adopted using a Carl Zeiss inverted microscope. The species identification was done following Needham and Needham (1962). The water samples were well shaken and aliquots of 15 ml were transferred into counting chambers for microscopic study. Sedimentation was carried out in counting chambers with a settling time of 4 hours for every 1 cm of water column of the sample described by Wetzel and Likens (1990). The densities of phytoplankton were expressed as number/m^3 from the average count of three aliquots of 5 ml each. Pictures of key phytoplankton species were later taken using a Celestron LCD digital microscope model # 44340.

3.4.3 Determination of Zooplankton and Macrobenthic invertebrates as measure of Secondary Productivity

For enumeration of zooplankton, a 5 ml sub-sample was taken from the water sample. Zooplankton counts were carried out in Sedgwick-Rafter counting chambers under an inverted microscope (Nikon Eclipse TE-200) as recommended by Downing and Rigler (1984). The zooplankton densities were expressed as number/m^3 from the average count of three aliquots of 5 ml each. Zooplankton species identification was guided by descriptions from Edmondson (1969), Jeje and Fernando (1986), and Fernando (2002). Pictures of key zooplankton species were later taken using a Celestron LCD digital microscope model # 44340.

For macrobenthic invertebrate population, the washed and preserved sediment with benthic invertebrates were poured into a white enamel tray and sorted. The sorting was made effective by adding moderate volume of water into the container to improve visibility. Large benthic organisms were picked using forceps, while the smaller ones were pipetted out. The organisms were sorted into their different groups and preserved in 4 % formaldehyde solution. The preserved invertebrates were later identified to their lowest taxonomic group following Days (1967), Pennak (1978), Hart (1994), and Merritt and Cummis (1996) under light and stereo dissecting microscope and counted to determine the number.

3.5 Data Analysis

3.5.1 Normality Test

Parametric statistical tests require that, the data are normally distributed and therefore need to always be checked if this assumption is violated. The normality test was done using the Statistical Package for Social Sciences, SPSS v. 16 (Martin and Acuna, 2002). In this software, two tests for normality are run. For dataset smaller than 2000 elements, the Shapiro-Wilk test is used; otherwise, the Kolmogorov-Smirnov test is used. In this study however, since the data sets were less than 2000 elements, the Shapiro-Wilk test was used to test for normality.

3.5.2 Predictive model for Fish Production

In order to develop a simple linear regression model for predicting fish production of the Black Volta near the Bui dam, a regression analysis was done using the Statistica software v. 8.0 (StatSoft.Inc., 2007) to analyse the relationship between water level, chlorophyll *a* concentration, phytoplankton and zooplankton densities and CPUE. Regression analysis provides the most comprehensive sensitivity measure and is commonly utilised to build response surfaces that approximate complex models (Hamby, 1994). Regression methods are often used to replace a highly complex model with a simplified 'response surface' (Cox, 1977). The response surface is simply a regression equation that approximates model output using only the most sensitive model input parameters. Regression coefficients provide a means of applying sensitive rankings to input parameters and have been used for such in several investigations (Iman and Conover, 1980; Iman *et al*., 1981; Helton *et al*., 1993). The use of the regression technique allows the sensitivity ranking to be determined based on the relative magnitude of the regression coefficient (Hamby, 1994). This value is indicative of the amount of influence the parameter has on the whole model.

3.5.3 Analysis of Variance

The plankton and fish abundance of the four hydrological seasons were analysed using one way analysis of variance (ANOVA) of SPSS v. 16 on log (x+1) transformed data. Fixed effect ANOVAs were performed using dates as replicates. Where there were significant differences ($p < 0.05$) among means, the Duncan multiple range (DMR) test was used to compare the treatment means.

3.5.4 Canonical Correspondence Analysis

Plankton – physico-chemical relationships were evaluated by Canonical Correspondence Analysis (CCA) using CANOCO software version 4.5A (Smilauer, 2003). Before using CCA, variables that covaried with other variables (Pearson correlation $r > 0.8$, $p < 0.05$) were removed. Rare species (< 2 % per season) were not included in the CCA. In addition, data were subjected to log (x + 1) transformation before the CCA analysis to prevent extreme values (outliers) from unduly influencing the ordination. Species – physico-chemical correlation coefficients provided a measure of how well variation in community composition could be explained by individual physico-chemical factors. A Monte Carlo permutation test with 499 permutations (Jckel, 1986) was used to assess the significance of the canonical axes extracted.

3.5.5 Fish species Diversity Indices

The data on richness, evenness and diversity of fish species during each hydrological season were calculated with the data from fish abundance by number using the PRIMER software version 6.1.6 (Clarke and Gorley, 2006).

Specific diversity was estimated using the Shannon-Wiener index (H´) (Shannon and Weaver, 1963) from the density data (bits $ind.^{-1}$) and was expressed as:

$H' = -\Sigma pi \log Pi$, where Pi is the proportion of individuals in the i^{th} species (Dahlberg and Odum, 1970).

The Margalef's species richness (D) was computed and expressed as:

$D = (S-1)/\log_e N$, where S is the number of species in the sample and N the total number of individuals.

The species equitability or evenness (J) (Pielou, 1969) was determined by the equation:

$J = H'/\log_2 S$, where H´ is diversity index and S is number of species in a population.

3.5.6 Estimation of Potential fish yield of Bui reservoir

Estimates of the potential fish yield were obtained using the physico-chemical characteristics of the reservoir and the relationship $Y = 23.281\ MEI^{0.447}$ (Marshal, 1984). Where Y is the potential fish yield in $kgha^{-1}yr^{-1}$, MEI is morpho-edaphic index, which is given in µS/cm and is estimated by dividing the mean conductivity by the mean depth (Ryder *et al.*, 1974).

3.5.7 Estimation of total fish catches of commercial gill nets

The estimated total quantity of fish produced by local fishers utilizing gill nets was done as follows:

Total fish catch (kg or metric tonnes) = (mean canoe-day per month x mean CPUE) x (fifty percent (50 %) of canoes assumed active or half the fishing effort) (after Ofori-Danson *et al.*, 2012).

CHAPTER FOUR

4.0 RESULTS

4.1 Physico-chemical Parameters

4.1.1 Electrical Conductivity

Tables 1 and 2 show the seasonality of mean physico-chemical factors in 2011 and 2012 with emphasis on which seasons were significantly different from the other. Table 3 shows the mean physico-chemical parameters during the pre- and post-impoundment periods of the Black Volta with emphasis on the impact of the dam during these periods. Table 4 on the other hand shows the mean station physico-chemical factors in 2011 and 2012 with emphasis on the impact of the dam on the downstream ecology.

In 2011, mean electrical conductivity (EC) values consistently decreased from 141.83 µS/cm in the dry season to 75.07 µS/cm in the post-wet season. There were significant differences ($p < 0.05$) between the dry and pre-wet seasons and the wet and post-wet seasons. The mean seasonal physico-chemical characteristic for the 2012 sampling year indicated that, EC values decreased from 122.53 µS/cm in the wet season to 82.3 µS/cm in post-wet season with significant difference ($p < 0.05$) between pre-wet season and post-wet season. The EC values also decreased from 142.13 µS/cm during the pre-impoundment period to 83.4 µS/cm during the immediate post-impoundment period with a significant difference ($p < 0.05$) between the pre- and post-impoundment periods.

Table 1: Seasonality of mean physico-chemical factors in 2011 (mean ± standard error)

Parameter	Dry season	Pre-wet season	wet season	Post-wet season
Conductivity (µS/cm)	$141.83^{a} \pm 0.65$	$129.97^{a} \pm 12.94$	$84.83^{b} \pm 7.92$	$75.07^{b} \pm 2.06$
Total Dissolved Solids (mgl^{-1})	$85.03^{a} \pm 0.31$	$78.09^{a} \pm 7.98$	$50.80^{b} \pm 4.71$	$40.50^{b} \pm 3.50$
Temperature (°C)	$29.50^{a} \pm 0.12$	$27.90^{a} \pm 1.41$	$27.37^{a} \pm 1.53$	$29.63^{a} \pm 0.13$
Colour (Hz)	$28.33^{b} \pm 1.67$	$165.00^{ab} \pm 10.54$	$291.67^{a} \pm 20.83$	$191.67^{ab} \pm 12.02$
Nitrates (mgl^{-1})	$0.25^{b} \pm 0.35$	$2.32^{a} \pm 0.96$	$2.96^{a} \pm 0.12$	$1.83^{ab} \pm 0.39$
Sulphates (mgl^{-1})	$3.85^{c} \pm 1.16$	$14.34^{bc} \pm 9.04$	$36.28^{a} \pm 1.78$	$21.73^{ab} \pm 0.43$
Phosphates (mgl^{-1})	$0.04^{a} \pm 0.18$	$0.02^{a} \pm 0.02$	$0.01^{a} \pm 0.01$	$0.29^{a} \pm 0.18$
Dissolved Oxygen (mgl^{-1})	$2.47^{a} \pm 0.58$	$2.51^{a} \pm 0.47$	$2.73^{a} \pm 0.12$	$1.77^{b} \pm 0.15$
Water level (m)	$0.85^{a} \pm 0.10$	$1.39^{a} \pm 0.36$	$5.73^{a} \pm 1.69$	$3.97^{a} \pm 2.27$

Figures on the same row with different superscript letters are significantly different ($p < 0.05$) from one another.

Hence there were seasonal variations with respect to some physico-chemical variables in 2011.

Table 2: Seasonality of mean physico-chemical factors in 2012 (mean ± standard error)

Parameter	**Dry season**	**Pre-wet season**	**wet season**	**Post-wet season**
Conductivity (µS/cm)	$104.07^{ab} \pm 2.95$	$98.63^{ab} \pm 4.81$	$122.53^{a} \pm 15.31$	$82.30^{b} \pm 2.03$
Total Dissolved Solids (mgl^{-1})	$62.50^{b} \pm 1.85$	$59.57^{b} \pm 2.66$	$76.67^{a} \pm 7.05$	$65.60^{ab} \pm 2.27$
Temperature (°C)	$30.20^{a} \pm 0.62$	$29.57^{a} \pm 1.84$	$27.20^{a} \pm 0.15$	$28.80^{a} \pm 0.15$
Colour (Hz)	$29.17^{a} \pm 16.73$	$32.50^{a} \pm 1.51$	$139.43^{a} \pm 61.68$	$67.53^{a} \pm 1.57$
Nitrates (mgl^{-1})	$1.12^{a} \pm 0.19$	$2.13^{a} \pm 0.91$	$2.25^{a} \pm 0.14$	$1.53^{a} \pm 0.38$
Sulphates (mgl^{-1})	$3.85^{c} \pm 1.89$	$7.67^{bc} \pm 4.66$	$21.13^{a} \pm 4.73$	$16.30^{ab} \pm 2.28$
Phosphates (mgl^{-1})	-	-	$0.11^{a} \pm 0.00$	$0.11^{a} \pm 0.02$
Dissolved Oxygen (mgl^{-1})	$3.30^{b} \pm 1.10$	$3.97^{ab} \pm 0.89$	$6.00^{a} \pm 0.06$	$5.87^{a} \pm 0.03$
Water level (m)	$0.89^{a} \pm 0.02$	$1.70^{a} \pm 0.70$	$1.34^{a} \pm 0.34$	$1.53^{a} \pm 0.49$

Figures on the same row with different superscript letters are significantly different ($p < 0.05$) from one another.

Hence there were seasonal variations with respect to some physico-chemical variables during 2012.

Table 3: Mean physico-chemical factors during the pre- and post-impoundment periods (mean ± standard error)

Parameter	**Pre-impoundment (Mar - May 2011)**	**Immediate post-impoundment (June - Dec 2011)**	**Late post-impoundment (Jan - Dec 2012)**	***P* value**
Conductivity (µS/cm)	$142.13^{a} \pm 0.93$	$83.40^{b} \pm 4.99$	$101.88^{b} \pm 5.57$	**0.00***
TDS (mg/l)	$85.47^{a} \pm 0.35$	$48.08^{b} \pm 3.83$	$66.07^{c} \pm 2.59$	**0.00***
Temperature (°C)	$29.27^{a} \pm 0.20$	$28.01^{a} \pm 0.87$	$28.94^{a} \pm 0.54$	**0.52**
Colour (Hz)	$50.00^{a} \pm 13.23$	$260.71^{b} \pm 28.33$	$67.16^{a} \pm 19.36$	**0.00***
Dissolve Oxygen (mg/l)	$2.56^{a} \pm 0.07$	$2.27^{a} \pm 0.19$	$4.78^{b} \pm 0.46$	**0.00***
Nitrates (mg/l)	$1.07^{a} \pm 0.61$	$2.62^{b} \pm 0.35$	$1.76^{ab} \pm 0.26$	**0.05**
Phosphates (mg/l)	$0.02^{a} \pm 0.02$	$0.13^{a} \pm 0.09$	$0.05^{a} \pm 0.02$	**0.39**
Sulphates (mg/l)	$5.00^{a} \pm 2.23$	$29.42^{b} \pm 2.86$	$13.7^{a} \pm 2.27$	**0.00***
Water level (m)	$0.98^{a} \pm 0.10$	$4.45^{b} \pm 1.19$	$1.36^{a} \pm 0.22$	**0.01***

**** on the P value indicates significant differences (p < 0.05); figures on the same row with different superscript letters are also significantly different (p < 0.05) from one another. Hence there was impact of the dam with respect to some physico-chemical variables between the pre- and post-impoundment periods.***

Table 4: Mean station physico-chemical factors in 2011 and 2012 (mean ± standard error)

Parameter	**2011**		**2012**	
	Bamboi	**Bui**	**Bamboi**	**Bui**
Conductivity (µS/cm)	101.00[a] ± 9.59	101.05[a] ± 12.88	101.88[a] ± 5.57	101.04[a] ±7.82
Total Dissolved Solids (mg/l)	59.29[a] ± 6.28	58.99[a] ± 8.14	66.08[a] ± 2.59	59.11[a] ± 5.00
Temperature (°C)	28.40[a] ± 0.63	28.68[a] ± 0.79	28.94[a] ± 0.54	28.54[a] ± 0.49
Colour (Hz)	197.50[a]± 37.71	114.80[a] ± 24.12	67.16[a] ± 19.36	156.10[a]±23.76
Nitrates (mg/l)	2.16[a] ± 0.37	1.86[a] ± 0.40	1.76[a] ± 0.26	2.01[a] ± 0.27
Sulphates (mg/l)	22.09[a] ± 4.25	19.82[a] ± 4.03	12.24[a] ± 2.58	20.96[a] ± 2.86
Phosphates (mg/l)	0.10[a] ± 0.06	0.21[a] ± 0.19	0.06[a] ± 0.02	0.16[a] ± 0.99
Dissolved Oxygen (mg/l)	2.35[a] ± 0.14	2.39[a] ± 0.11	4.78[a] ± 0.46	2.37[a] ± 0.09

Figures on the same row with same superscript letters within the same year are homogenous ($p < 0.05$). Hence there was no impact of the dam with respect to all the measured physico-chemical variables on the downstream station during the study period.

4.1.2 Total Dissolved Solids

The total dissolved solids (TDS) decreased from 85.03 mgl^{-1} during the dry season to 40.5 mgl^{-1} in the post-wet season in 2011. There were significant differences ($p < 0.05$) between the dry and pre-wet seasons and the wet and post-wet seasons. In 2012, TDS increased from 59.57 mgl^{-1} in the pre-wet season to 76.67 mgl^{-1} in the wet season with significant differences ($p < 0.05$) between the pre-wet and wet seasons. TDS also decreased from 85.47 mgl^{-1} during the pre-impoundment period to 48.08 mgl^{-1} during the immediate post-impoundment period. There were significant differences ($p < 0.05$) between the pre-impoundment, immediate post-impoundment and late post-impoundment periods.

4.1.3 pH

The pH values increased from a minimum of 7.16 in the dry season to 7.78 in the post-wet season in 2011. There were however, no significant differences ($p > 0.05$) among all the four hydrological seasons. Seasonal pH for 2012 decreased from 7.73 in the dry season to 7.29 in the wet season with no significant differences among the four seasons. The pH values also increased from 7.47 during the immediate post-impoundment period to 7.57 in the late post-impoundment period. There were no significant differences ($p > 0.05$) among the three periods.

4.1.4 Temperature

The temperature rose from 27.37 ºC in the wet season to 29.63 ºC in the post-wet season in 2011 with no significant differences ($p > 0.05$) among the four seasons. In 2012 however, temperature decreased from 30.2 ºC in the dry season to 27.2 ºC in the wet season with no significant differences ($p > 0.05$) among the four seasons. Temperature also decreased from 29.27 ºC in the pre-impoundment phase to 28.01 ºC during the immediate post-impoundment period with no significant differences ($p > 0.05$) among the three periods.

4.1.5 Dissolved Oxygen

Dissolved oxygen (DO) values recorded decreased from 2.73 mgl^{-1} in the wet season to 1.77 mgl^{-1} in the post-wet season with a significant difference ($p < 0.05$) between the post-wet season and the other three seasons in 2011. DO levels in 2012 however, increased from 3.3 mgl^{-1} in the dry season to 6.0 mgl^{-1} in the wet season with no significant difference ($p > 0.05$) between the wet and post-wet seasons. DO values also increased from 2.27 mgl^{-1} during the immediate post-impoundment period to 4.78 mgl^{-1} during the late post-impoundment period. There was significant difference ($p < 0.05$) between the late post-impoundment period and the other two periods.

4.1.6 Colour

The colour values in the 2011 sampling year increased from 28.33 Hz in the dry season to 291.67 Hz in the wet season. There was a significant difference ($p < 0.05$) between the dry and wet seasons. The colour values of the water in 2012 also increased from 29.17 Hz in the dry season to 139.43 Hz in the wet season with no significant differences ($p > 0.05$) among the four seasons. The colour values increased from 50 Hz during the pre-impoundment period to 260.71 Hz during the immediate post-impoundment period. There were significant differences ($p < 0.05$) between the immediate post-impoundment and the other 2 periods.

4.1.7 Nitrates

Mean seasonal nitrate levels in 2011 increased from 0.25 mgl^{-1} in the dry season to 2.96 mgl^{-1} in the wet season with significant differences ($p < 0.05$) between the dry season and the other three seasons. In the 2012 sampling year, nitrate levels also increased from 1.12 mgl^{-1} in the dry season to 2.25 mgl^{-1} in the wet season. There was however no significant differences ($p > 0.05$) among the four seasons. Nitrate levels also increased from 1.07 mgl^{-1} during the pre-impoundment period to 2.62 mgl^{-1} during the immediate post-impoundment period. There was significant difference ($p < 0.05$) between the pre- and immediate post-impoundment periods.

4.1.8 Phosphates

Phosphate levels in 2011 increased from 0.01 mgl^{-1} in the wet season to 0.29 mgl^{-1} in the post-wet season with no significant differences ($p > 0.05$) among the four hydrological seasons. In 2012 however, there was no phosphate levels recorded during the dry and pre-wet seasons. A value of 0.11 mgl^{-1} was however, recorded for both the wet and post-wet seasons. A mean of 0.056 mgl^{-1} was recorded in 2012. The phosphate levels also increased from 0.02 mgl^{-1} during the pre-impoundment period to 0.13 mgl^{-1} during the immediate post-impoundment period. There was however no significant differences ($p > 0.05$) among the three periods.

4.1.9 Sulphates

Sulphate levels in 2011 increased from 3.85 mgl^{-1} in the dry season to 36.28 mgl^{-1} in the wet season with a significant difference ($p < 0.05$) between the dry and wet seasons. Sulphate levels in 2012 also increased from 3.85 mgl^{-1} in the dry season to 21.13 mgl^{-1} in the wet season with a significant difference ($p < 0.05$) between the wet and dry seasons. Sulphate levels also increased from 5.0 mgl^{-1} during the pre-impoundment period to 29.42 mgl^{-1} during the immediate post-impoundment period. There were significant differences ($p < 0.05$) between the immediate post-impoundment period and the two other periods.

4.1.10 Water Level

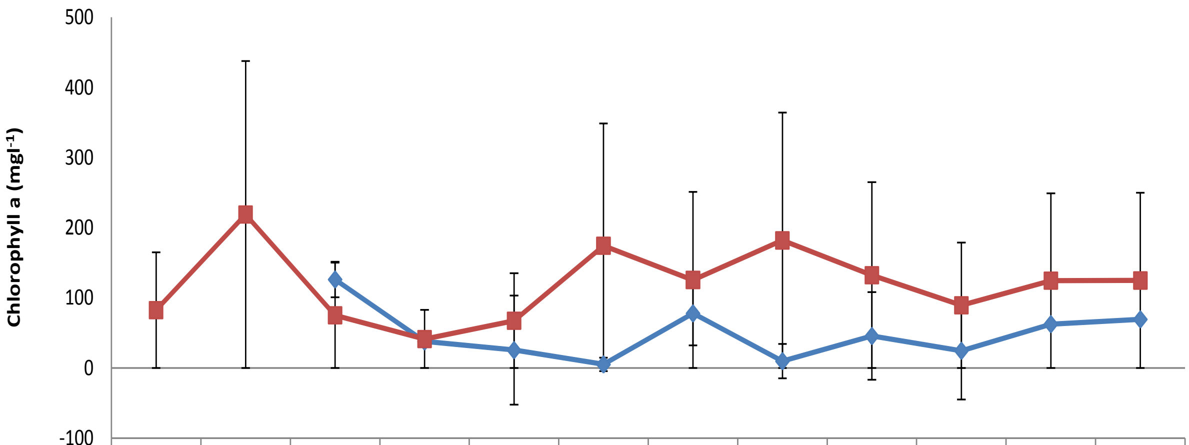
During the 2011 sampling year, water level increased from 0.85 m in the dry season to 5.73 m in the wet season with no significant differences ($p > 0.05$) among the four seasons. In 2012 however, the water level increased from 0.89 m in the dry season to 1.7 m in the pre-wet season. There were no significant differences ($p > 0.05$) among the four seasons. The water levels also increased from 0.98 m during the pre-impoundment period to 4.45 m in the immediate post-impoundment period with a significant difference ($p < 0.05$) between the immediate post-impoundment period and the other two periods.

4.2 Hydro-biological Factors

Tables 5 and 6 show the seasonality of hydro-biological factors in 2011 and 2012 with emphasis on which seasons were significantly different from the other. Table 7 shows the hydro-biological factors before and after the impoundment of the Black Volta with emphasis on the impact of the dam during these periods, while Table 8 on the other hand shows the mean station hydro-biological factors in 2011 and 2012 with emphasis on the impact of the dam on the downstream ecology.

4.2.1 Chlorophyll *a* Concentration

Figure 2 shows the mean monthly chlorophyll *a* concentration in 2011 and 2012. In 2011, chlorophyll *a* concentration decreased from 126.1 mgl^{-1} in March to 5.12 mgl^{-1} in June. In 2012 however, chlorophyll *a* concentration decreased from 218.9 mgl^{-1} in February to 41.41 mgl^{-1} in April. In 2011, chlorophyll *a* content decreased from 126.1 mgl^{-1} in the dry season to 22.7 mgl^{-1} in the pre-wet season with significant difference ($p < 0.05$) between the dry season and the three other hydrological seasons. Chlorophyll *a* levels also increased from 94.4 mgl^{-1} in the pre-wet season to 146.7 mgl^{-1} in the wet season with no significant differences ($p > 0.05$) among the four seasons during 2012. There was also no significant difference ($p > 0.05$) in chlorophyll *a* concentration between the upstream (Bui) and downstream (Bamboi) stations in 2011 and 2012. The chlorophyll *a* levels increased from 42.08 mgl^{-1} during the immediate post-impoundment period to 119.92 mgl^{-1} during the late post-impoundment period. There was significant difference ($p < 0.05$) between the immediate post-impoundment and late post-impoundment period. Hence, the alternative hypothesis that the hydro-biological factors and fisheries production will change following the impoundment of the Black Volta by the Bui dam is accepted and the null rejected.



	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2011			126.1	37.7	25.42	5.12	77.8	9.73	45.6	24.53	62.5	69.31
2012	82.5	218.9	75.2	41.41	67.48	174.4	125.6	182	132.4	89.4	124.6	124.9

Figure 2: Mean monthly chlorophyll *a* concentration in 2011 and 2012 (standard error bars)

Table 5: Seasonality of mean hydro-biological factors in 2011 (mean ± standard error)

FACTOR	Dry season	Pre-wet season	Wet season	Post-wet season
PRIMARY PRODUCTION				
Chlorophyll *a* (mgl^{-1})	126.1^{a} ± 0.1	22.7^{b} ± 0.9	44.4^{b} ± 1.9	52.1^{b} ± 13.9
PHYTOPLANKTON				
Bacillariophyceae (no. m^{-3})	171.7^{ab} ± 6.1	357.7^{a} ± 15.0	8.0^{b} ± 0.6	1.7^{b} ± 0.2
Chlorophyceae (no. m^{-3})	1119.3^{ab} ± 24.9	1983.0^{a} ± 45.7	8.0^{c} ± 0.5	244.0^{bc} ± 20.2
Cyanophyceae (no. m^{-3})	194.0^{a} ± 2.5	1907.3^{b} ± 10.9	274.3^{a} ± 2.9	641.0^{a} ± 29.0
Euglenophyceae (no. m^{-3})	-	53.3 ± 2.7	-	-
Total phytoplankton (no. m^{-3})	1414.3^{ab} ± 30.5	3906.0^{a} ± 19.6	290.3^{b} ± 12.5	886.7^{ab} ± 48.8
ZOOPLANKTON				
Cladocera (no. m^{-3})	32.3^{a} ± 2.3	684.3^{b} ± 6.6	2085.3^{b} ± 20.2	-
Copepoda (no. m^{-3})	68.3^{a} ± 2.3	733.3^{b} ± 5.9	1043.7^{b} ± 10.4	24.7^{a} ± 2.2
Total zooplankton (no. m^{-3})	100.7^{a} ± 2.5	1417.7^{b} ± 12.4	3134.0^{b} ± 30.6	24.7^{a} ± 2.2

Figures on the same row with different superscript letters are significantly different ($p < 0.05$) from one another.

Hence there were seasonal variations with respect to some hydro-biological factors during 2011.

Table 6: Seasonality of mean hydro-biological factors in 2012 (mean ± standard error)

FACTOR	Dry season	Pre-wet season	Wet season	Post-wet season
PRIMARY PRODUCTION				
Chlorophyll *a* (mgl^{-1})	$125.5^{a} \pm 4.7$	$94.4^{a} \pm 4.1$	$146.7^{a} \pm 17.8$	$112.9^{a} \pm 11.8$
PHYTOPLANKTON				
Bacillariophyceae (no. m^{-3})	$6.0^{a} \pm 0.4$	$27.3^{a} \pm 0.9$	$4.7^{a} \pm 0.7$	$6.3^{a} \pm 1.5$
Chlorophyceae (no. m^{-3})	$194.7^{ab} \pm 8.2$	$246.0^{a} \pm 4.6$	$62.0^{b} \pm 3.5$	$56.7^{b} \pm 19.0$
Cyanophyceae (no. m^{-3})	$696.0^{a} \pm 15.3$	$533.0^{ab} \pm 21.2$	$117.0^{b} \pm 7.4$	$94.7^{b} \pm 12.7$
Total phytoplankton (no. m^{-3})	$897.7^{a} \pm 18.2$	$806.3^{a} \pm 26.3$	$183.7^{b} \pm 11.3$	$157.7^{b} \pm 32.6$
ZOOPLANKTON				
Cladocera (no. m^{-3})	$60.7^{ab} \pm 2.4$	$110.0^{a} \pm 3.8$	$145.3^{a} \pm 2.8$	$11.0^{b} \pm 2.5$
Copepoda (no. m^{-3})	$83.0^{ab} \pm 3.0$	$89.3^{ab} \pm 1.9$	$157.3^{a} \pm 7.2$	$16.0^{b} \pm 6.1$
Total zooplankton (no. m^{-3})	$143.7^{ab} \pm 5.4$	$199.3^{ab} \pm 5.8$	$302.7^{a} \pm 9.9$	$27.0^{b} \pm 7.1$

Figures on the same row with different superscript letters are significantly different ($p < 0.05$) from one another.

Hence there were seasonal variations with respect to some hydro-biological factors during 2012.

Table 7: Mean hydro-biological factors during the pre- and post-impoundment periods (mean ± standard error)

Factor	**Pre-impoundment (Mar - May 2011)**	**Immediate post-impoundment (June - Dec 2011)**	**Late post-impoundment (Jan - Dec 2012)**	***P* value**
PRIMARY PRODUCTION				
Chlorophyll *a* (mgl^{-1})	$63.1^{ab} \pm 3.2$	$42.1^{a} \pm 1.1$	$119.9^{b} \pm 15.1$	**0.01***
PHYTOPLANKTON				
Bacillariophyceae (no. m^{-3})	$268.0^{a} \pm 12.6$	$78.4^{b} \pm 7.4$	$11.1^{b} \pm 3.7$	**0.02***
Chlorophyceae (no. m^{-3})	$108.2^{a} \pm 46.8$	$518.4^{b} \pm 40.2$	$139.8^{a} \pm 33.1$	**0.09**
Cyanophyceae (no. m^{-3})	$654.3^{a} \pm 39.3$	$962.4^{a} \pm 52.3$	$360.2^{a} \pm 97.7$	**0.35**
Euglenophyceae (no. m^{-3})	53.3 ± 2.7	-	-	**0.00***
Total phytoplankton (no/m^{3})	$2057.7^{b} \pm 91.2$	$1559.3^{a} \pm 99.6$	$511.1^{a} \pm 26.1$	**0.22**
ZOOPLANKTON				
Cladocera (no. m^{-3})	$29.7^{a} \pm 5.8$	$1179.4^{b} \pm 86.9$	$423.9^{a} \pm 28.6$	**0.20**
Copepoda (no. m^{-3})	$130.3^{a} \pm 26.2$	$731.4^{a} \pm 47.9$	$86.4^{a} \pm 12.8$	**0.18**

Table 7 (continued)

Total zooplankton (no. m^{-3})	$160.0^{a} \pm 2.8$	$1910.9^{b} \pm 13.4$	$168.2^{a} \pm 4.5$	**0.19**
MACRO-INVERTEBRATES				
Chironomidae (no.)	$15.0^{a} \pm 0.3$	$15.5^{a} \pm 0.9$	$19.0^{a} \pm 1.5$	**0.87**
Ephemeroptera (no.)	$7.0^{a} \pm 0.4$	$11.0^{a} \pm 0.6$	$8.0^{a} \pm 1.8$	**0.75**
Odonata (no.)	$1.5^{a} \pm 0.1$	$2.5^{a} \pm 0.2$	$2.5^{a} \pm 1.0$	**0.82**
Tricoptera (no.)	$8.5^{a} \pm 0.6$	$3.0^{a} \pm 0.3$	$7.8^{a} \pm 1.3$	**0.63**
Total Benthos (no.)	$32.0^{a} \pm 1.5$	$32.0^{a} \pm 1.7$	$37.0^{a} \pm 2.6$	**0.78**

***** on the P values indicates significant differences ($p < 0.05$); figures on the same row with different superscript letters are also significantly different ($p < 0.05$) from one another. Hence there was impact of the dam with respect to some hydro-biological factors between the pre- and post-impoundment periods.***

Table 8: Mean station hydro-biological factors in 2011 and 2012 (mean ± standard error)

	2011		2012	
FACTOR	**Bamboi**	**Bui**	**Bamboi**	**Bui**
Primary productivity				
Chlorophyll a (mgl^{-1})	$48.4^{a} \pm 1.6$	$55.9^{a} \pm 1.7$	$119.9^{a} \pm 15.1$	$52.2^{a} \pm 8.1$
Phytoplankton				
Bacillariophyceae ($no.m^{-3}$)	$133.9^{a} \pm 6.8$	$268.1^{a} \pm 1.3$	$11.1^{a} \pm 0.7$	$201.0^{b} \pm 13.9$
Chlorophyceae (no. m^{-3})	$3603.6^{a} \pm 31.7$	$5465.7^{a} \pm 44.4$	$139.8^{a} \pm 13.1$	$4536.2^{b} \pm 26.6$
Cyanophyceae (no. m^{-3})	$2647.1^{a} \pm 17.1$	$7930.2^{b} \pm 45.5$	$360.2^{a} \pm 17.7$	$5288.6^{b} \pm 24.4$
Euglenophyceae (no. m^{-3})	$8.0^{a} \pm 0.1$	$8.0^{a} \pm 0.1$	-	-
Total phytoplankton (no. m^{-3})	$6395.3^{a} \pm 40.9$	$14862.0^{b} \pm 89.8$	$511.1^{a} \pm 12.6$	$10629.0^{b} \pm 50.7$
Zooplankton				
Cladocera (no. m^{-3})	$723.7^{a} \pm 16.1$	$6569.0^{b} \pm 49.3$	$81.7^{a} \pm 9.0$	$3146.4^{b} \pm 24.8$
Copepoda (no. m^{-3})	$397.7^{a} \pm 3.1$	$2345.4^{b} \pm 18.5$	$86.1^{a} \pm 12.8$	$1371.6^{b} \pm 9.4$
Total zooplankton (no. m^{-3})	$1121.4^{a} \pm 19.2$	$7915.4^{b} \pm 67.8$	$168.2^{a} \pm 20.5$	$4518.4^{b} \pm 34.2$

Table 8 (continued)

Macro-invertebrates

Chironomidae (no.)	$21.0^{a} \pm 2.2$	$40.0^{a} \pm 6.5$	$37.0^{a} \pm 6.8$	$39.0^{a} \pm 4.2$
Ephemeroptera (no.)	$17.0^{a} \pm 1.6$	$19.0^{a} \pm 2.1$	$15.0^{a} \pm 1.3$	$17.0^{a} \pm 1.8$
Odonata (no.)	$3.0^{a} \pm 0.9$	$5.0^{a} \pm 0.4$	$4.0^{a} \pm 0.5$	$6.0^{a} \pm 0.5$
Tricoptera (no.)	$19.0^{a} \pm 1.6$	$4.0^{a} \pm 0.2$	$25.0^{a} \pm 2.5$	$6.0^{a} \pm 0.9$
Total Benthos (no.)	$60.0^{a} \pm 3.2$	$68.0^{a} \pm 1.8$	$81.0^{a} \pm 4.7$	$68.0^{a} \pm 2.3$

Figures on the same row with different superscript letters within the same year are significantly different ($p < 0.05$) from one another. Hence there were impacts of the dam with respect to some hydro-biological factors on the downstream station.

4.2.2 Phytoplankton Composition, Density and Diversity

Table 9 shows a checklist of phytoplankton species during the pre- and post-impoundment periods. Figures 3 and 4 show the species composition and abundance of phytoplankton in 2011 and 2012. Figure 5 shows the composition of phytoplankton groups during 2011 and 2012. Figure 6 on the other hand, show the monthly distribution of total phytoplankton abundance during 2011 and 2012, while Figure 7 shows the seasonal variations in density and diversity of phytoplankton during 2011 and 2012.

In 2011, thirty-five species of phytoplankton belonging to four classes were identified. The four classes were Bacillariophyceae (7.6 %), Chlorophyceae (43 %), Cyanophyceae 48.6 %) and Euglenophyceae (0.8 %). In the 2012 sampling year, however, eighteen species belonging to three classes were identified. The three classes were Bacillariophyceae (2.2 %), Chlorophyceae (26.1 %) and Cyanophyceae (71.7 %).

Mean monthly total phytoplankton abundance decreased from June (7384 no. m^{-3}) to August (106 no. m^{-3}) in 2011. In 2012 however, mean total phytoplankton decreased February (1237 no. m^{-3}) to August (4 no. m^{-3}). The phytoplankton species diversity in 2011 and 2012 are presented in Appendix VIII.

Table 9: Checklist of phytoplankton species identified during the pre- and post-impoundment periods

TAXA	Pre-impoundment (Mar - May 2011)	Immediate post - impoundment (Jun - Dec 2011)	Late post-impoundment (Jan - Dec 2012)
BACILLARIOPHYCEAE			
Gyrosigma sp.	+	+	-
Navicula sp.	+	+	+
Surirella sp.	+	-	-
Synedra ulna	+	+	+
CHLOROPHYCEAE			
Ankistrodesmus sp.	+	+	+
Carteria sp.	+	-	-
Chlamydomonas sp.	+	-	+
Chlorella sp.	+	+	+
Chlorogonium sp.	+	-	-
Closterium sp.	+	-	-
Coelastrum sp.	+	+	-
Cosmarium sp.	+	-	-
Dictyosphaerium sp.	+	-	-
Micrasterias sp.	+	-	+
Pediastrum sp.	+	-	+
Scenedesmus sp.	+	+	+
Schroederia sp.	+	-	-
Staurastrum sp.	+	-	+
Stigeoclonium sp.	+	-	-
Ulothrix sp.	+	+	+
Volvox sp.	-	+	-
CYANOPHYCEAE			
Anabaena sp.	+	+	+

Chroococcus sp.	+	-	-

Table 9 (continued)

Coelosphaerium sp.	+	-	-
Lyngbya circumcreta	+	+	+
Merismopedia puntata	-	+	+
Microcystis aeruginosa	+	+	+
Microcystis wesenbergii	-	+	+
Oscillatoria sp.	+	+	+
Planktothrix sp.	+	+	+
Pseudanabaena sp.	+	+	+
Rivularia sp.	+	-	-
Spirulina sp.	+	-	-
EUGLENOPHYCEAE			
Euglena sp.	+	-	-
Phacus pyrum	+	-	-

+ means present; - indicates absent

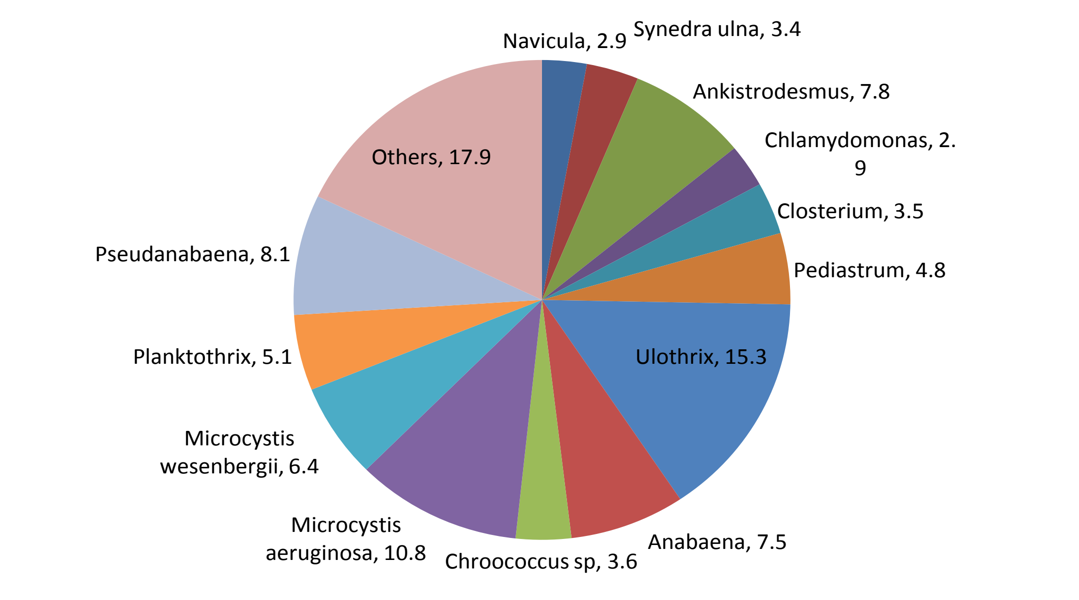


Figure 3: Phytoplankton species composition in 2011

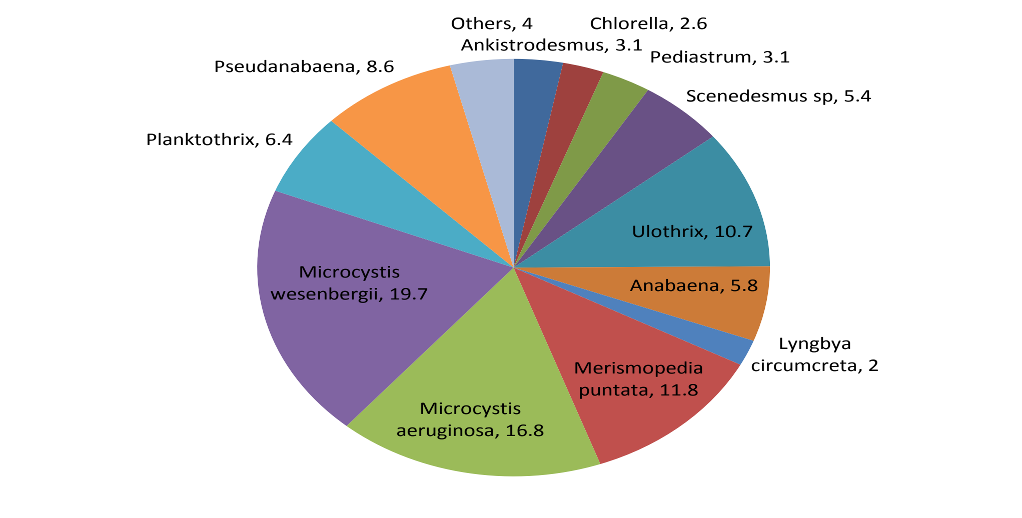
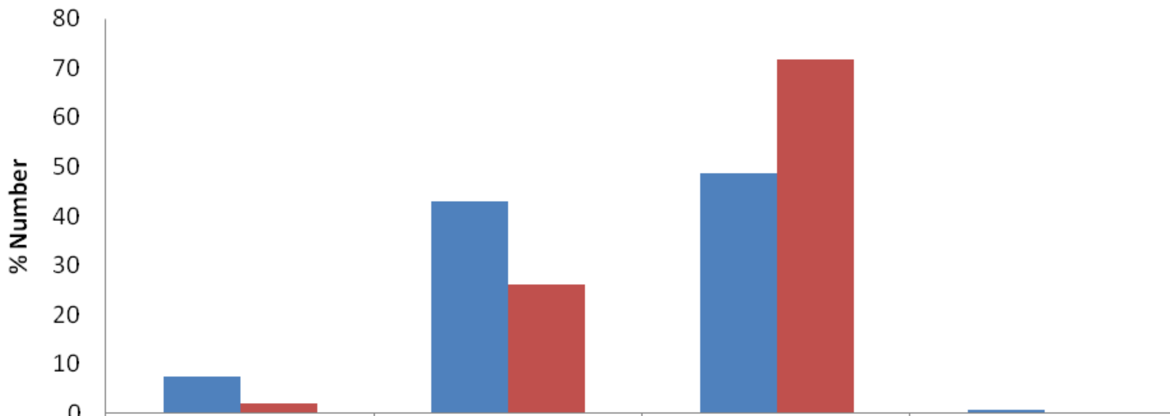
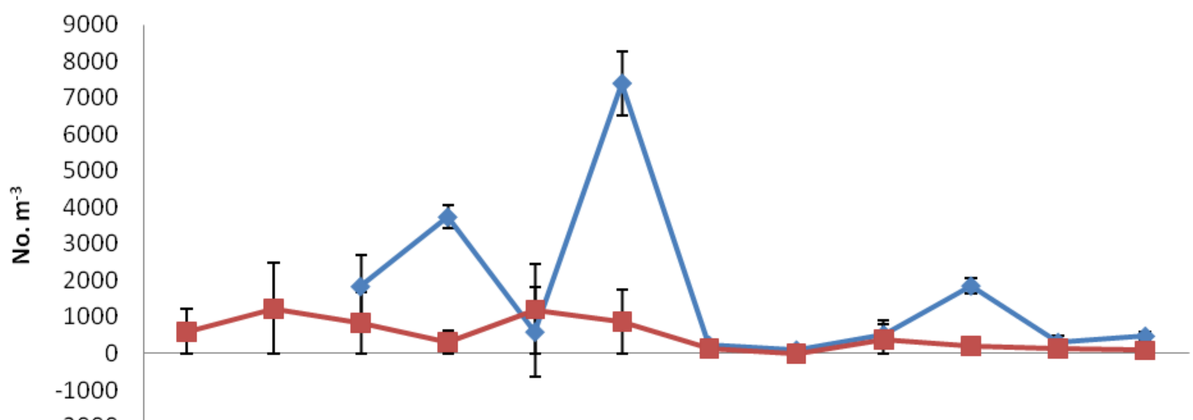


Figure 4: Phytoplankton species composition in 2012



	Bacillariophyceae	Chlorophyceae	Cyanophyceae	Euglenophyceae
2011	7.6	43	48.6	0.8
2012	2.2	26.1	71.7	0

Figure 5: Composition of phytoplankton groups in 2011 and 2012



	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2011			1839	3735	599	7384	237	106	528	1858	317	485
2012	614	1237	839	319	1219	881	154	4	393	214	158	101

Figure 6: Mean monthly changes in the quantitative distribution of total phytoplankton in 2011 and 2012 (standard error bars)

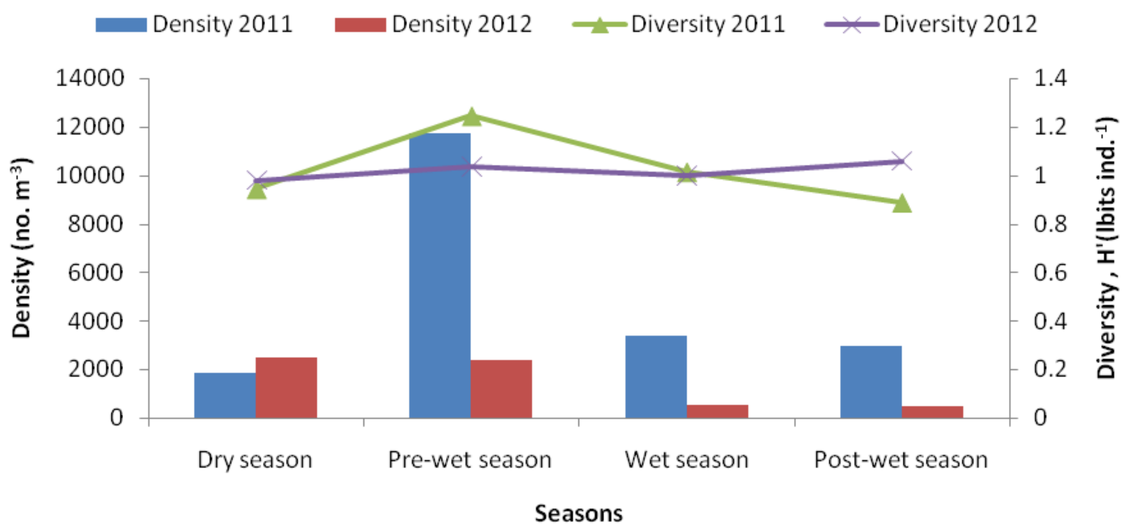


Figure 7: Seasonal variations in density and diversity of phytoplankton in 2011 and 2012

4.2.2.1 Class Bacillariophyceae

Four species of Bacillariophyceae were identified with *Synedra ulna* (Plate B) being the most dominant species in 2011 (Appendix II). In 2012, however, 2 species were identified with *Synedra ulna* being the dominant species as well (Appendix III). The mean seasonal variations in Bacillariophyceae during 2011 decreased from 171.7 no. m^{-3} in the dry season to 1.7 no. m^{-3} in the post-wet season with a significant difference ($p < 0.05$) between the pre-wet and wet seasons. Bacillariophyceae densities decreased from 27.3 no. m^{-3} in the pre-wet season to 4.67 no. m^{-3} in the wet season with significant differences ($p < 0.05$) between the pre-wet season and the three other hydrological seasons in 2012. There was no significant difference ($p > 0.05$) between the upstream and downstream stations in 2011. There was however, a significant difference ($p < 0.05$) between the upstream and downstream stations in 2012. Also, mean densities of Bacillariophyceae decreased from 268 no. m^{-3} during the pre-impoundment period to 11.08 no.

m^{-3} during the late post-impoundment period. There was a significant difference ($p < 0.05$) between the pre- and post-impoundment periods. Hence, the alternative hypothesis that hydro-biological factors will change following the impoundment of the Black Volta by the Bui dam was accepted and the null rejected.

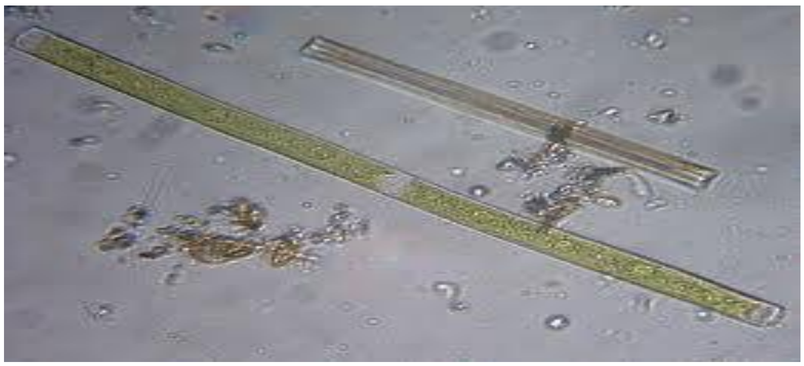


Plate B: *Synedra sp*. (Bacillariophyceae)

4.2.2.2 Class Chlorophyceae

In 2011, 17 species of Chlorophyceae were identified with *Ulothrix sp.* (Plate C) as the most dominant. In 2012 however, 8 species were recorded with *Ulothrix sp.* as the most dominant. The mean seasonal variations in Chlorophyceae densities during 2011 decreased from 1983 no. m^{-3} in the pre-wet season to 8 no. m^{-3} in the wet season with a significant difference ($p < 0.05$) between the pre-wet and wet seasons. Chlorophyceae densities decreased from 246 no. m^{-3} in the pre-wet season to 56.7 no. m^{-3} in the post-wet season in 2012. There were significant differences ($p < 0.05$) between the pre-wet, wet and post-wet seasons. There was no significant difference ($p > 0.05$) between the upstream and downstream stations in 2011. There was however, a significant difference ($p < 0.05$) between the upstream and downstream stations in 2012. Also, mean densities of Chlorophyceae increased from 108.2 no. m^{-3} during the pre-impoundment period to 518.43 no. m^{-3} in the immediate post-impoundment period. There was a significant difference ($p < 0.05$) between the pre-impoundment and late post-impoundment periods. Hence, the alternative hypothesis that hydro-biological factors will change following the impoundment of the Black Volta by the Bui dam was accepted and the null rejected.

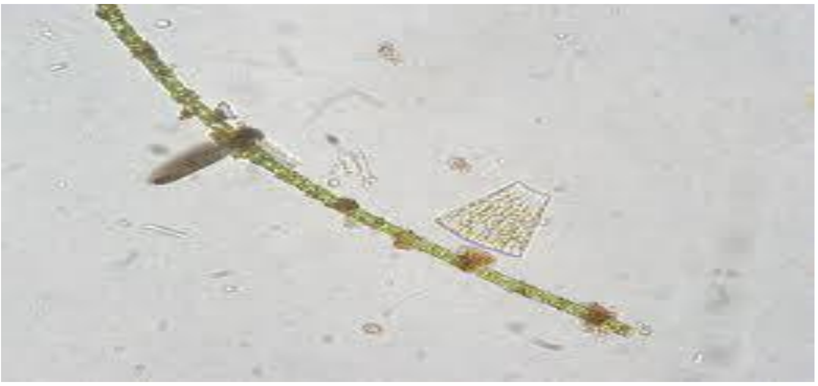


Plate C: *Ulothrix sp*. (Chlorophyceae)

4.2.2.3 Class Cyanophyceae

Twelve (12) species of Cyanophyceae were recorded and the dominant species was *Microsystis aeruginosa* (Plate D) in 2011. In 2012 however, 8 species were recorded and *Microsystis wesenbergii* predominated in the samples. The mean seasonal variations in 2011 increased from 194 no. m^{-3} in the dry season to 1907.3 no. m^{-3} in the pre-wet season with a significant difference ($p < 0.05$) between the pre-wet season and the other 3 hydrological seasons. During 2012, densities also increased from 94.7 no. m^{-3} during the post-wet season to 696 no. m^{-3} in the dry season. There were significant differences ($p < 0.05$) between the dry and wet seasons.

Cyanophyceae differed significantly ($p < 0.05$) between the upstream and downstream stations in both 2011 and 2012. Also, Cyanophyceae densities decreased from 962.43 no. m^{-3} in the immediate post-impoundment period to 360.17 no. m^{-3} during the late post-impoundment period with no significant difference among the periods. Hence, the null hypothesis that hydro-biological factors will not change following the impoundment of the Black Volta by the Bui dam was accepted and the alternative rejected.

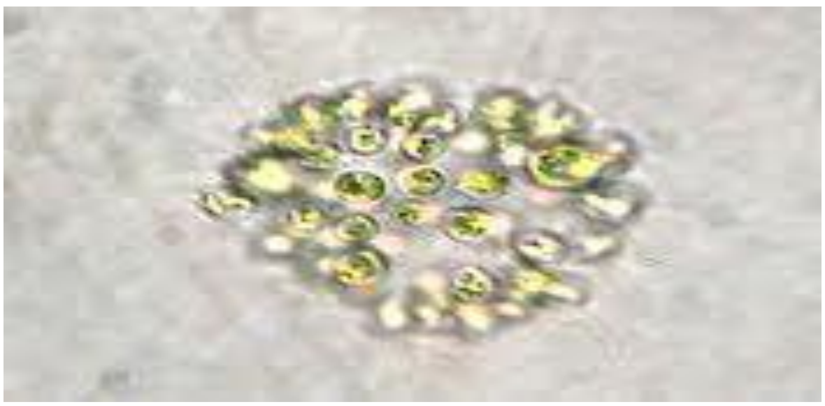


Plate D: *Microcystis sp.* (Cyanophyceae)

4.2.2.4 Class Euglenophyceae

Two species of the class Euglenophyceae: *Euglena sp.* and *Phacus pyrum* (Plate E) were recorded in the 2011 sampling year. During 2012, however, no species of Euglenophyceae was identified. The seasonal variations showed that species of the class Euglenophyceae were absent in the dry, wet and post-wet seasons, but present (53.3 no. m^{-3}) in the pre-wet season. There were however, significant differences ($p < 0.05$) between the pre-wet season and the other three hydrological seasons. Also, species of Euglenophyceae were only present during the pre-impoundment period of the study. Hence, the alternative hypothesis that hydro-biological factors will change following the impoundment of the Black Volta by the Bui dam was accepted and the null rejected.

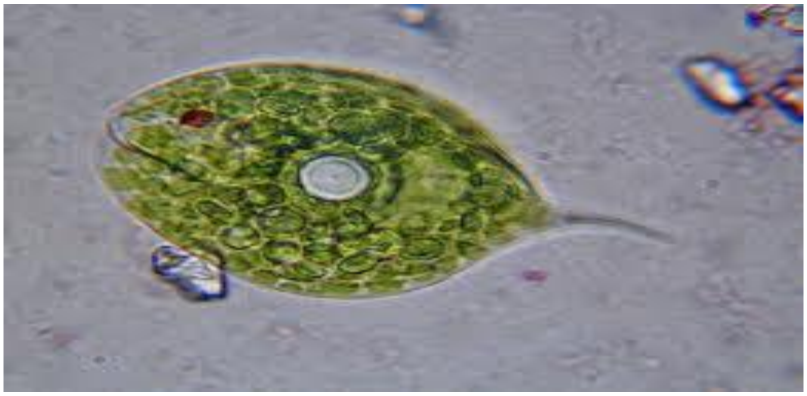


Plate E: *Phacus sp.* (Euglenophyceae)

4.2.3 Zooplankton Composition, Density and Diversity

Table 10 shows a checklist of zooplankton species identified during the pre- and post-impoundment periods. Figures 8 and 9 shows the species composition and abundance of zooplankton in 2011 and 2012, whiles Figure 10 shows the composition of zooplankton groups in 2011 and 2012. Figure 11 on the other hand, shows monthly distribution of total zooplankton abundance in 2011 and 2012, while Figure 12 shows seasonal variations in density and diversity of zooplankton in 2011 and 2012. In both the 2011 and 2012 sampling years, sixteen species of zooplankton belonging to two taxonomic groups: Subclass Copepoda and Order Cladcera were recorded (Appendices IV and V). Appendix IX on the other hand, show the diversity indices for zooplankton species in the 2011 and 2012 sampling years.

Table 10: Checklist of zooplankton species identified during the pre- and post-impoundment periods

Taxa	Pre-impoundment (Mar - May 2011)	Immediate post-impoundment (Jun - Dec 2011)	Late post-impoundment (Jan - Dec 2012)
CLADOCERA			
Alonella sp	-	+	+
Bosmina sp	+	-	+
Ceriodaphnia sp	+	+	+
Daphnia sp	+	-	+
Diaphanosoma sp	+	-	+
Leptodora sp	+	+	+
Macrothrix sp	+	-	+
Moina sp	+	-	+
Polyphemus sp	+	+	+
Sida sp	-	+	+
COPEPODA			
Canthocamptus sp	+	-	+
Cyclops sp	+	+	+
Cypridopsis sp	+	+	+
Diaptomus sp	+	+	+
Eubranchanpus sp	+	-	+
Limnocalanus sp	+	-	+

+ means present; - indicates absent

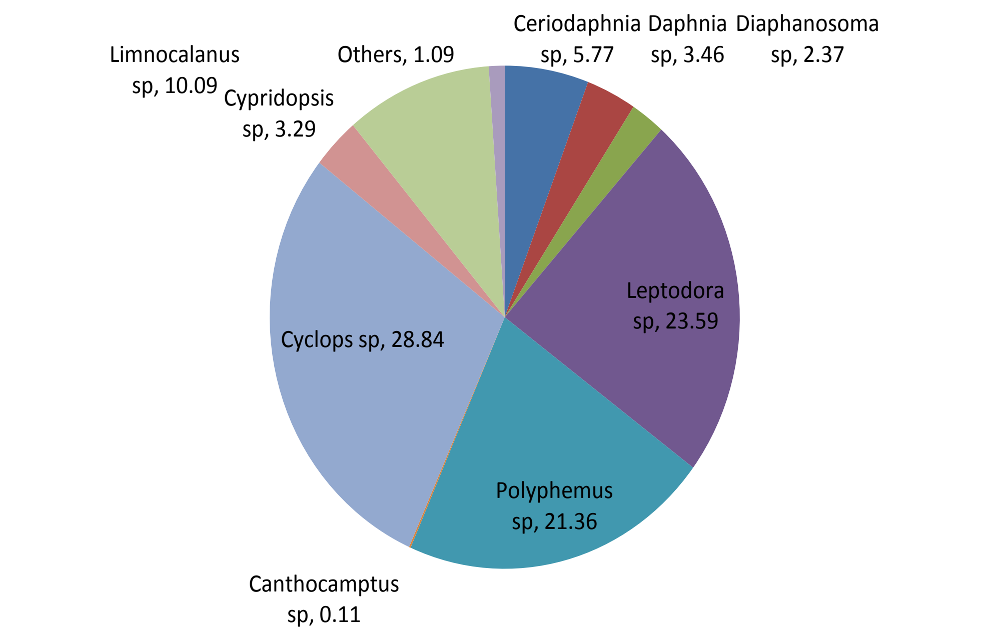


Figure 8: Zooplankton species composition in 2011

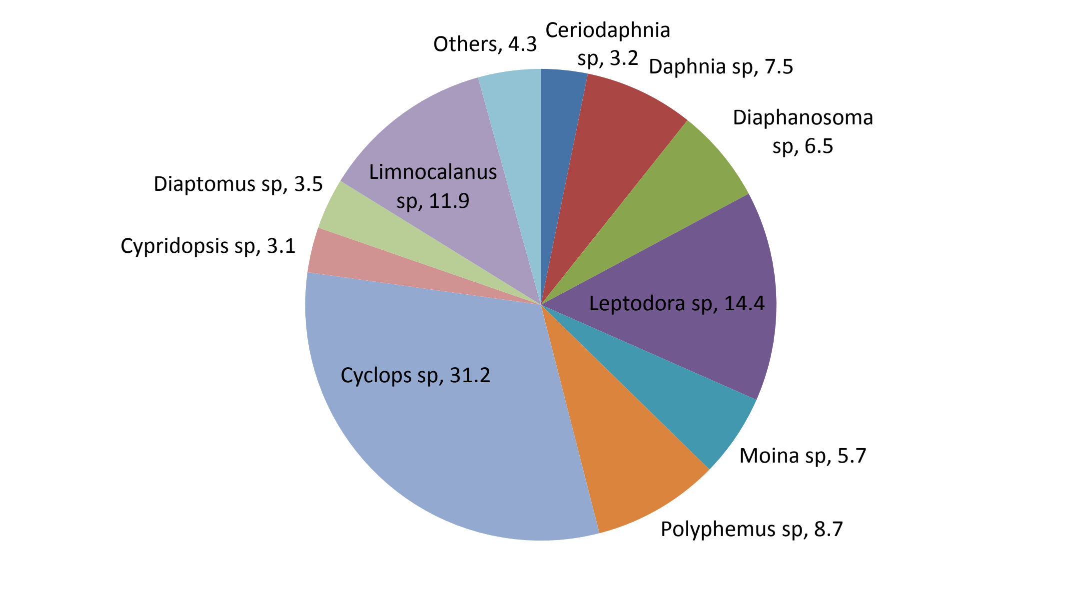
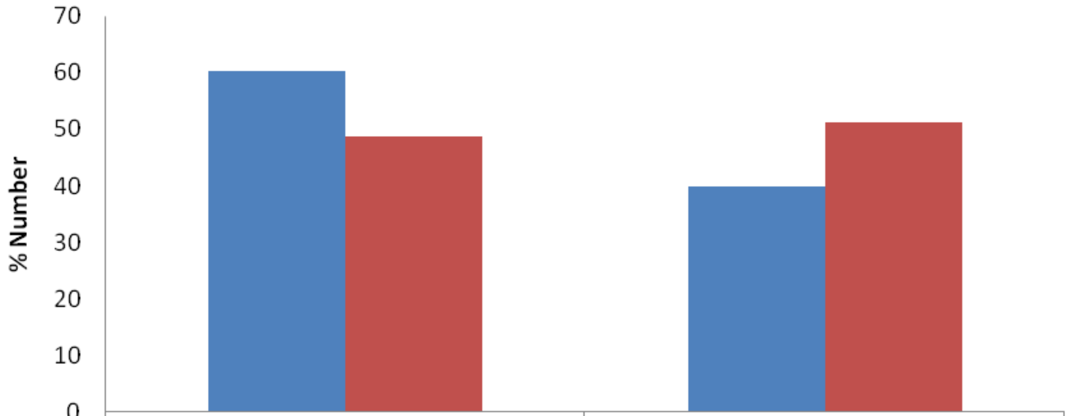
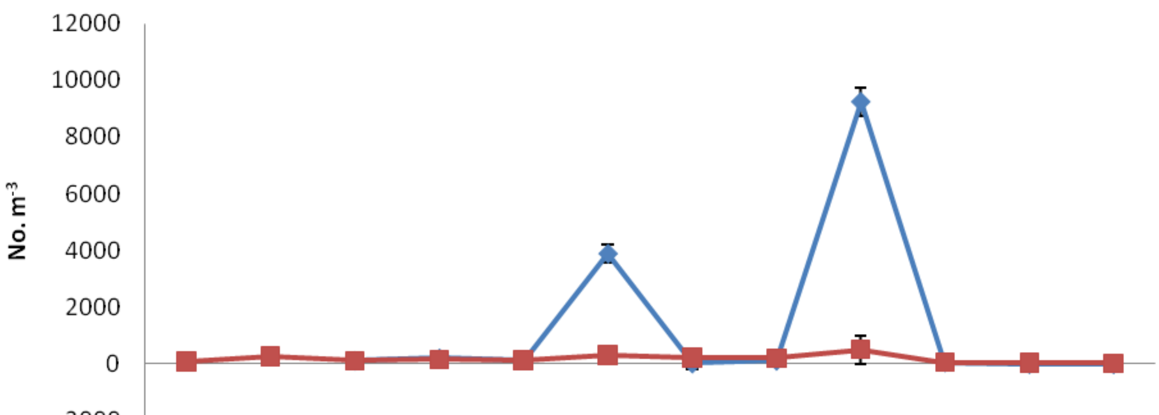


Figure 9: Zooplankton species composition in 2012



	Cladocera	Copepoda
2011	60.2	39.8
2012	48.8	51.2

Figure 10: Composition of zooplankton groups in 2011 and 2012



	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2011			127	215	138	3900	27	125	9250	68	5	1
2012	84	251	96	151	132	315	207	199	502	41	22	18

Figure 11: Monthly changes in the quantitative distribution of total zooplankton in 2011 and 2012 (standard error bars)

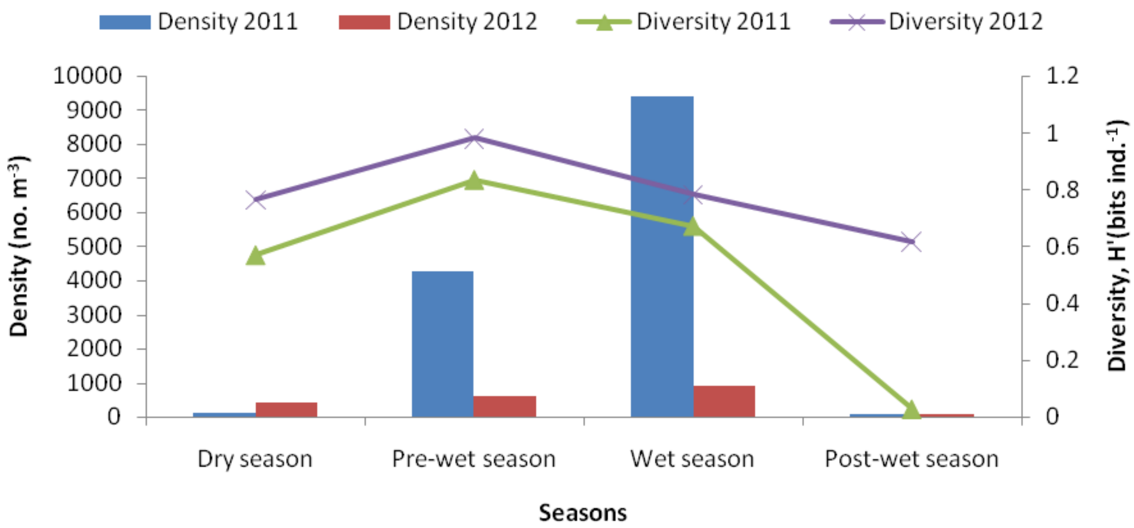


Figure 12: Seasonal variations in density and diversity of zooplankton in 2011 and 2012

4.2.3.1 Order Cladocera

Ten Cladoceran species were recorded in 2011 and 2012 (Appendix V). They were dominated by *Leptodora sp.* (Plate F). Density in 2011 decreased from 2085.3 no. m^{-3} in the wet season to 0.0 no. m^{-3} (absent) in the post-wet season. There were significant differences ($p < 0.05$) between the wet and post-wet seasons. Cladoceran densities decreased from 145.3 no.m^{-3} in the wet season to 11 no. m^{-3} in the post-wet season with significant difference ($p < 0.05$) between the wet season and post-wet season in 2012. Cladoceran densities differed significantly ($p < 0.05$) between the upstream and downstream stations in both 2011 and 2012. Cladoceran densities increased from 29.67 no. m^{-3} in the pre-impoundment period to 1179.4 no.m^{-3} during the immediate post-impoundment period. They differed significantly ($p < 0.05$) from each other during the pre- and immediate post-impoundment periods. Hence, the alternative hypothesis that hydro-biological factors will change following the impoundment of the Black Volta by the Bui dam was accepted and the null rejected.

Plate F: *Leptodora sp*. (Cladocera)

4.2.3.2 Subclass Copepoda

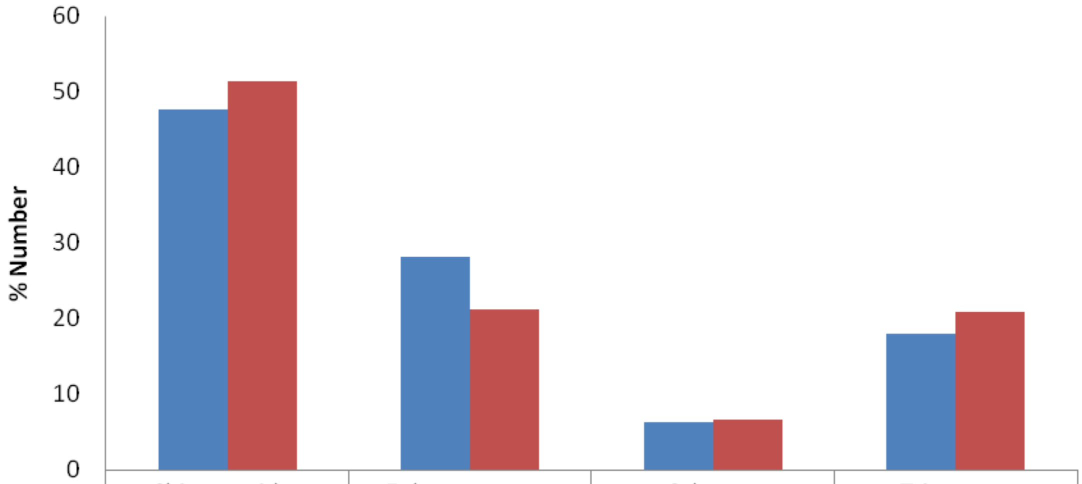
Six species of the subclass Copepoda were recoded and were dominated by *Cyclops sp.* (Plate G) in 2011 and 2012 (Appendix V). Densities during 2011 decreased from 1043.7 no. m^{-3} in the wet season to 24.7 no. m^{-3} during the post-wet season with a significant difference ($p < 0.05$) between the dry and wet seasons. Mean seasonal densities also decreased from 157.3 no. m^{-3} in the wet season to 16 no. m^{-3} in the post-wet season in 2012. There was a significant difference ($p < 0.05$) between the wet season and post-wet season. There was also a significant difference in Copepoda densities between the upstream and downstream stations in both 2011 and 2012. Copepoda densities decreased from 731.43 no. m^{-3} during the immediate post-impoundment period to 86.42 no. m^{-3} during the late post-impoundment period. They did not, however, differ significantly ($p > 0.05$) from each other during the pre- and post-impoundment periods. Hence, the null hypothesis that hydro-biological factors will not change following the impoundment of the Black Volta by the Bui dam was accepted and the alternative rejected.

Plate G: *Cyclops sp*. (Copepoda)

4.2.4 Macrobenthic invertebrate Composition, Abundance and Diversity

Figure 13 shows the macrobenthic invertebrate composition in 2011 and 2012. Figure 14 on the other hand, shows the seasonal variations in abundance and diversity of macrobenthic invertebrates in 2011 and 2012. The diversity indices of macrobenthic invertebrates are shown in Appendix V. Chironomidae (Plate H) dominated the benthic samples during both 2011 and 2012 sampling years. The seasonal distribution showed that the post-wet season recorded the highest invertebrate numbers, while the wet season recorded the lowest numbers during the 2 years sampling period. Chironomidae increased from 15 during the pre-impoundment period to 19 during the late post-impoundment period. Ephemeroptera numbers, however, increased from 7 during the pre-impoundment period to 11 in the post-impoundment period. There was however no significant differences ($p > 0.05$) in the numbers of the macro-invertebrates between the pre- and post-impoundment periods. Hence, the null hypothesis that hydro-biological factors will not change following the impoundment of the Black Volta by the Bui dam was accepted and the alternative rejected.

Plate H: Chironomidae (Benthos)



	Chironomidae	Ephemeroptera	Odonata	Tricoptera
2011	47.7	28.1	6.3	17.9
2012	51.4	21.2	6.6	20.8

Figure 13: Macrobenthic invertebrate composition in 2011 and 2012

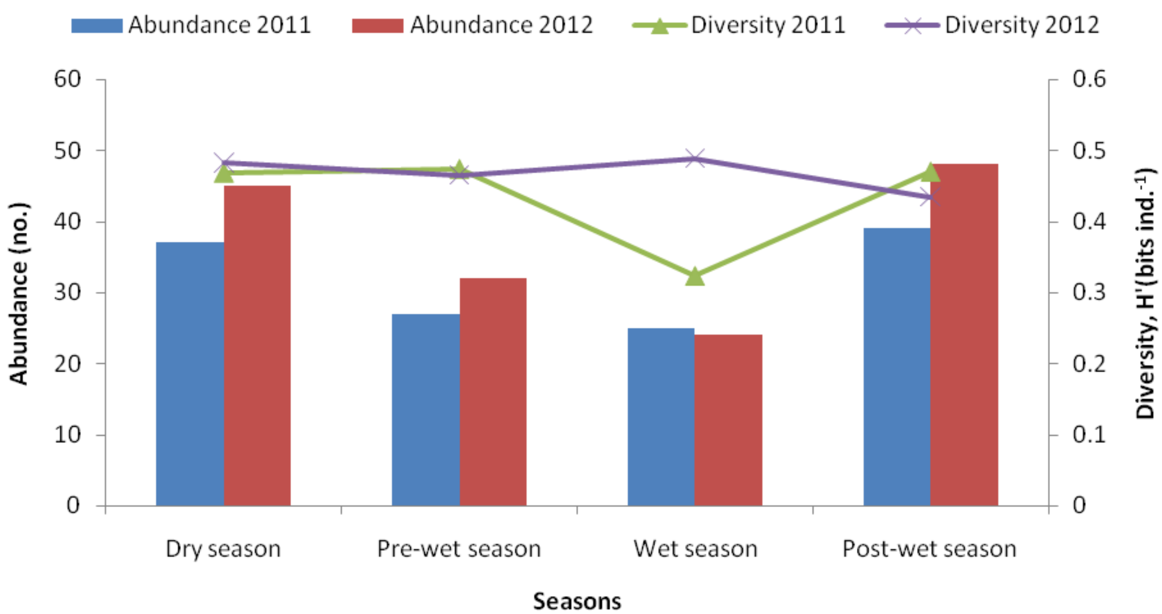


Figure 14: Seasonal variations in abundance and diversity of macrobenthic invertebrates in 2011 and 2012

4.3 Relationship between Hydro-biological and Physico-chemical factors

Tables 11 and 12 show the correlation among physico-chemical factors during 2011 and 2012. The CCA ordination showed a high correlation between conductivity and TDS (r = 0.997), nitrates and sulphates (r = 0.819), phosphates and temperature (r = 0.718) but the correlation between DO and conductivity (r = 0.461), temperature and pH (r = 0.299), pH and TDS (r = -0.299), temperature and DO (r = -0.788) were moderate or low in 2011. Thus, making, conductivity, TDS, temperature, nitrates and phosphates the driving physico-chemical factors in 2011.

The CCA ordination showed a positive correlation between conductivity and TDS (r = 0.913), phosphates and TDS (r = 0.953), sulphates and TDS (r = 0.944), DO and phosphates (r = 0.981), DO and sulphates (r = 0.981), phosphates and sulphates (r = 0.977) and pH and temperature (r = 0.993). There was, however, a negative correlation between pH and conductivity (r = -0.883), temperature and conductivity (r = -0.849) and temperature and DO (r = -0.975) in 2012. Thus, making, conductivity, TDS, DO, pH, temperature, phosphates and sulphates the driving physico-chemical factors in 2012.

Table 11: Correlation among physico-chemical factors used in CCA during 2011

	Cond.	**TDS**	**pH**	**Temp.**	**Nitrates**	**Sulphate**	**Phos.**	**DO**
Cond.	1							
TDS	**0.997**	1						
pH	-0.2488	-0.2991	1					
Temp.	-0.0353	-0.1049	0.2994	1				
Nitrates	-0.5002	-0.4519	0.1629	-0.795	1			
Sulphate	-0.807	-0.7596	-0.1098	-0.5405	**0.8199**	1		
Phos.	-0.634	-0.6919	0.6713	**0.7181**	-0.1488	0.0624	1	
DO	0.4612	0.5282	-0.7431	-0.7879	0.2671	0.1361	-0.9771	1

Relationships that are highly positively correlated are in bold

Table 12: Correlation among physico-chemical factors used in CCA during 2012

	Cond.	**TDS**	**pH**	**Temp.**	**Nitrates**	**Sulphate**	**Phos.**	**DO**
Cond.	1							
TDS	**0.9129**	1						
pH	-0.8833	-0.9843	1					
Temp.	-0.8499	-0.9572	**0.9932**	1				
Colour	**0.8926**	**0.989**	-0.9996	-0.9895				
Nitrates	0.4889	0.497	-0.6347	-0.7153	1			
Sulphate	0.7908	**0.9438**	-0.984	-0.9933	0.6968	1		
Phos.	0.7538	**0.9534**	-0.9659	-0.9569	0.5328	**0.9767**	1	
DO	0.7237	**0.9162**	-0.9622	-0.9755	0.6823	**0.9943**	**0.9805**	1

Relationships that are highly positively correlated are in bold

4.3.1 Phytoplankton – Physico-chemical Relationship

The relationship between phytoplankton species abundance and measured physico-chemical factors in 2011 and 2012 are shown in Figures 15 and 16. In 2011, 8 % of variations in species abundance were accounted for by the physico-chemical factors measured. The strongest explanatory factors were nitrates, phosphates and temperature. The length of the environmental arrows in the CCA ordination plots indicates their relative importance to each axis. Environmental arrows represent a gradient, where the mean value is located at the origin, and the arrow points in the direction of its increase. It was observed from the ordination plot that *Pseudanabaena sp.* was more sensitive to nitrates. *Lyngbya circumcreta,* on the other hand, seemed to prefer phosphates or more likely low DO while *Synedra ulna* was associated with moderate temperature. *Chlorella sp.* and *Scenedesmus sp.* were extremely positioned. TDS, conductivity, pH and sulphates had little influence on the variations on phytoplankton species abundance.

In 2012, 12 % of variations in the species abundance were accounted for by the physico-chemical factors measured. The most important physico-chemical variables were conductivity, TDS and nitrates. It was observed that *Pseudanabaena sp.* and *Microcystis wesenbergii* were associated with moderate conductivity but low levels of nitrates. *Planktothrix sp.*, *Scenedesmus sp.* and *Anabaena sp.* were more sensitive to high nitrate levels but low TDS. *Merismopedia puntata* and *Pediastrum sp.* preferred high TDS or more likely, a lower temperature and pH. There was little or no influence of pH, temperature, phosphates, sulphates and DO on the species abundance during 2012.

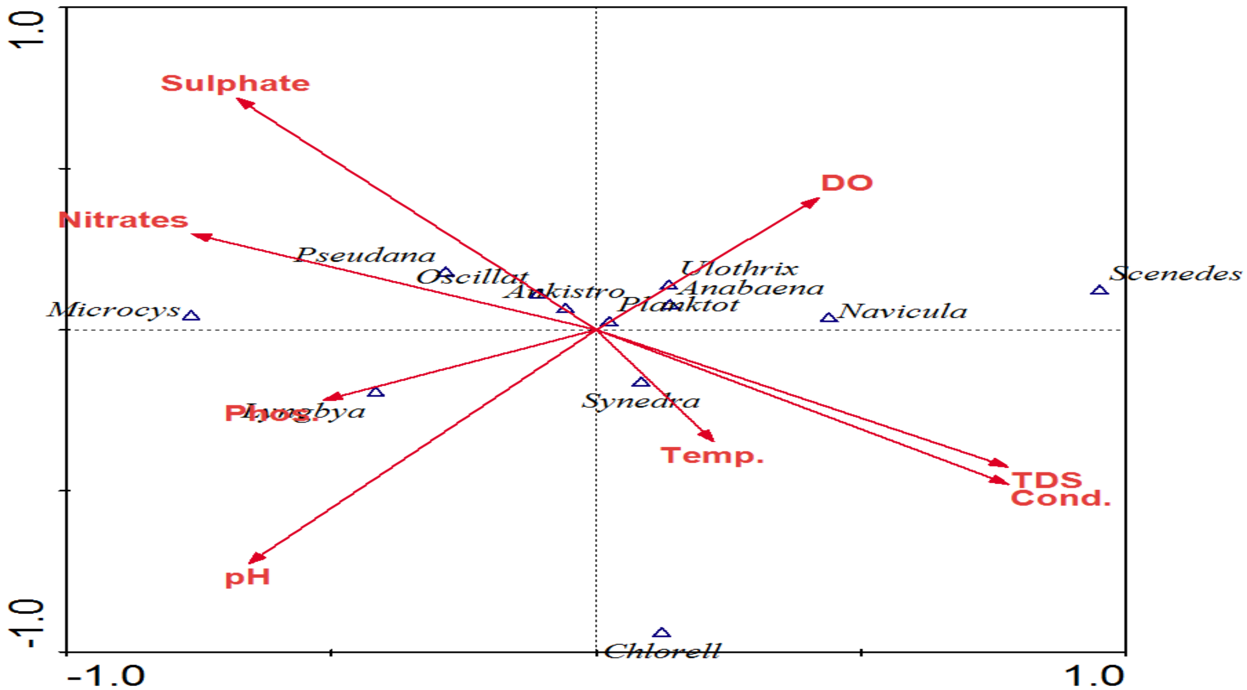


Figure 15: CCA ordination plot for phytoplankton – physico-chemical relationship in 2011

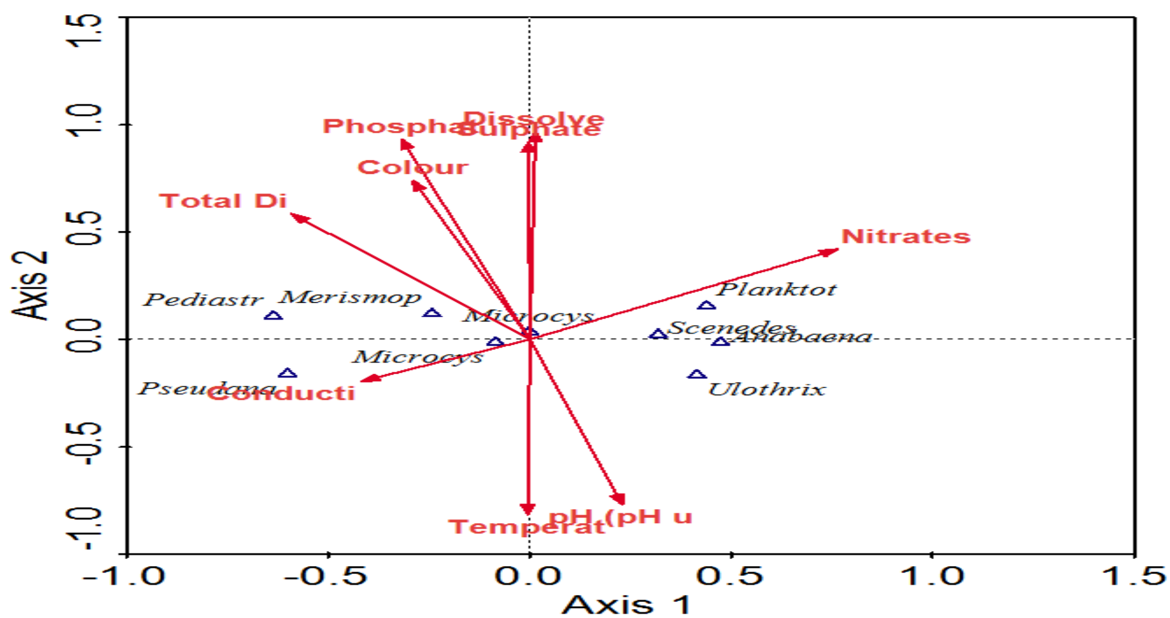


Figure 16: CCA ordination plot for phytoplankton – physico-chemical relationship in 2012

4.3.2 Zooplankton – Physico-chemical Relationship

The relationship between zooplankton species abundance and measured environmental variables in 2011 and 2012 are shown in Figures 17 and 18. In 2011, the strongest explanatory factors were conductivity, TDS, pH, DO and phosphates. About 64 % of variations in the species abundance data were accounted for by the physico-chemical factors measured. The *Daphnia sp.* and *Diaphanosoma sp.* were more sensitive to moderate to high levels of conductivity and TDS but to low levels of sulphate and DO. *Ceriodaphnia sp.* and *Polyphemus sp.* preferred moderate DO values or more likely low temperature and phosphates. *Cyclops sp.* on the other hand preferred moderate phosphate levels but lower sulphate and nitrate levels, while *Cypridopsis sp.* preferred high pH levels. *Diaptomus sp.* and *Leptodora sp.* were extremely positioned.

In 2012, the strongest explanatory factors were conductivity, pH, temperature, phosphates, sulphates, and nitrates. About 41 % variations in the species abundance data were accounted for by measured physico-chemical parameters. It was observed that the frequency of *Leptodora sp.*, *Polyphemus sp.* and *Ceriodaphnia sp.* were associated with low levels of conductivity, phosphates and sulphates. *Diaphanosoma sp.* Was, however, associated with pH and temperature. *Moina sp.*, *Ceriodaphnia sp.*, *Limnocalanus sp.* and *Daphnia sp.* were extremely positioned.

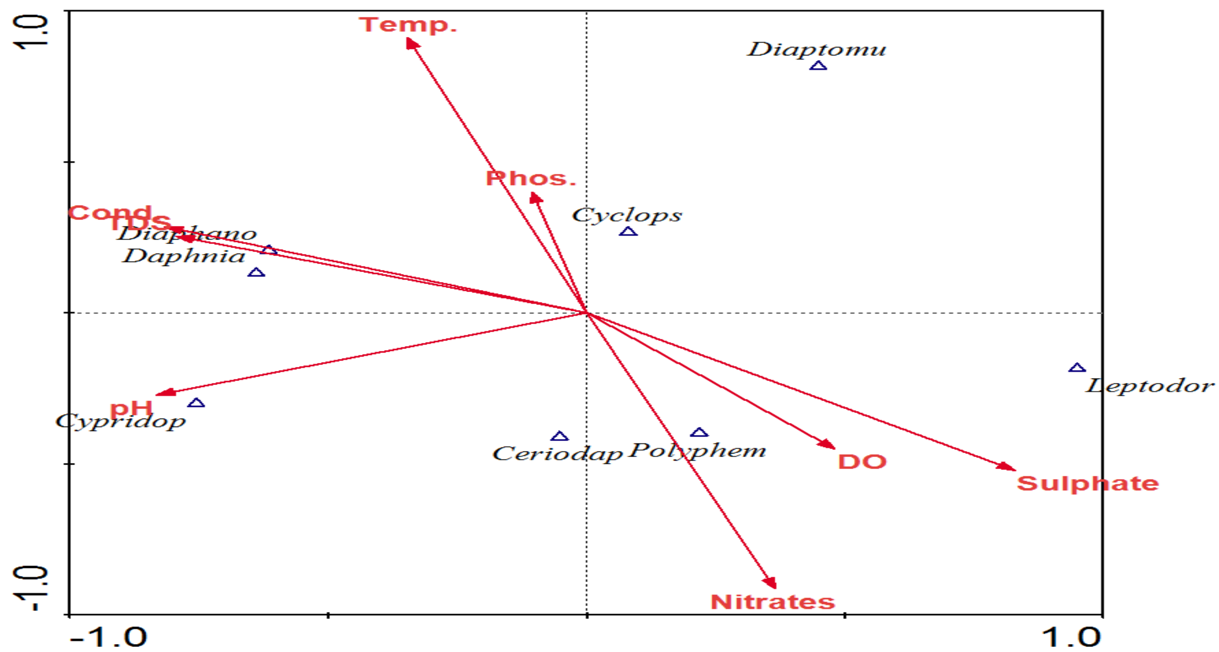


Figure 17: CCA ordination plot for zooplankton – physico-chemical relationship in 2011

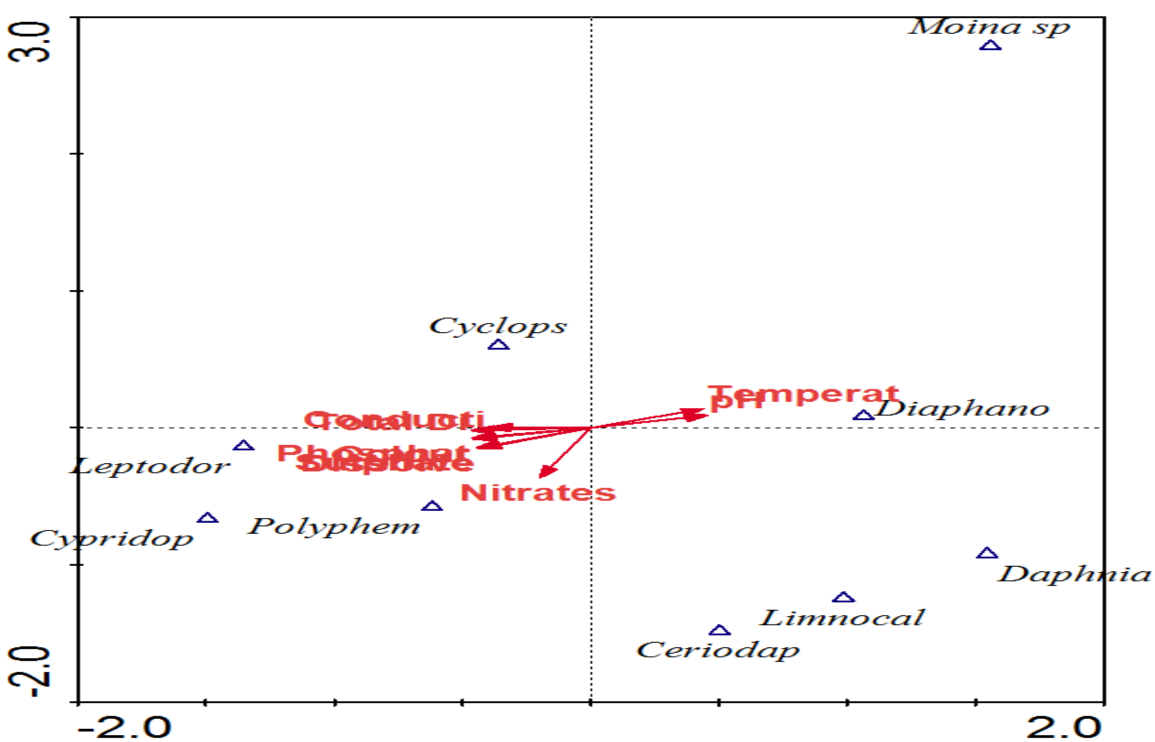


Figure 18: CCA ordination plot for zooplankton – physico-chemical relationship in 2012

4.4 The Fisheries

4.4.1 Commercial gill net catches

Table 13 shows a checklist of fish species identified during the pre- and post-impoundment periods, while Plates I – N show the key fish species identified during the study period. Appendix VI presents a summary of fish species abundance by number and weight (kg) during the study period. Twenty families, thirty-eight genera and sixty-three fish species were recorded in 2011. The following species dominated the catches in this order: *Synodontis sorex* > *Schilbe mystus* > *Heterobranchus bidorsalis* representing 12.46 %, 7.47 % and 6.44 %, respectively by number. *Bagrus docmak* > *Synodontis sorex* > *Labeo coubie* dominated the fish catches representing 7.12 %, 5.83 % and 4.47 %, respectively by weight. By comparison, thirteen fish families, twenty-four genera and forty-two species were identified in 2012. The species: *Labeo senegalensis* > *Sarotherodon galilaeus* > *Brycinus nurse* dominated the catches representing 18.14 %, 11.2 % and 7.57 %, respectively by number. *Labeo senegalensis* > *Labeo coubie* > *Sarotherodon galilaeus* representing 20.44 %, 10.99 % and 5.27 %, respectively by weight.

The fish families Mochokidae > Alestidae > Clariidae clearly dominated the catches representing 33.63 %, 11.28 % and 11.07 %, respectively by number. Mormyridae > Clariidae > Mochokidae dominated the catches representing 14.18 %, 12.7 % and 12.6 %, respectively by weight during 2011. The following fish families dominated the fish catches by number: Cyprinidae (25.71 %) > Alestidae (17.9 %) > Mochokidae (14.27 %) in 2012 (Figure 19). The fish diversity indices for 2011 and 2012 sampling years are shown in Appendix XI.

Table 13: Checklist of fish species identified during the pre- and post-impoundment periods

FAMILY/SPECIES	Pre-impoundment (Mar - May 2011)	Immediate post-impoundment (Jun - Dec 2011)	Late post-impoundment (Jan - Dec 2012)
ALESTIDAE			
Alestes baremoze	+	+	+
Alestes dentex	-	+	+
Brycinus leuciscus	+	-	+
Brycinus macrolepidotus	+	+	+
Brycinus nurse	-	+	+
Hydrocynus brevis	+	+	-
Hydrocynus forskalii	+	+	+
ANABANTIDAE			
Ctenopoma petherici	-	+	-
BAGRIDAE			
Bagrus bajad	+	+	+
Bagrus docmak	+	+	+
CICHLIDAE			
Chromidotilapia guntherii	+	+	+
Hemichromis fasciatus	-	+	-
Oreochromis niloticus	-	+	-
Sarotherodon galilaeus	-	+	+

Table 13 (continued)

Steatocranus irvinea	+	-	+
Tilapia dageti	+	-	-
Tilapia zillii	+	+	-
CENTROPOMIDAE			
Lates niloticus	+	+	+
CITHARINIDAE			
Citharinus citharus	+	+	-
CLARIIDAE			
Clarias anguillaris	-	+	+
Clarias gariepinus	-	+	+
Heterobranchus bidorsalis	-	+	-
Heterobranchus isopterus	-	+	-
CLAROTIDAE			
Auchenoglanis occidentalis	+	+	+
Chrysichthys auratus	+	+	+
Chrysichthys nigrodigitatus	+	+	+
CLUPEIDAE			
Odaxothrissa mento	+	+	-
CYPRINIDAE			
Labeo coubie	+	+	+
Labeo parvus	-	+	+

Table 13 (continued)

Labeo senegalensis	+	+	+
Raiamas senegalensis	+	-	-
DISTICHODONTIDAE			
Distichodus brevipinnis	+	+	-
Distichodus engycephalus	+	+	+
Distichodus rostratus	-	+	+
HEPSETIDAE			
Hepsetus odoe	+	+	+
MALAPTERURIDAE			
Malapterurus electricus	+	+	+
MASTACEMBELIDAE			
Aethiomastacembelus nigromarginatus	-	+	-
MOCHOKIDAE			
Hemisynodontis membranaceus	+	+	+
Synodontis clarias	-	+	+
Synodontis eupterus	+	-	-
Synodontis filamentosus	+	+	+
Synodontis nigrita	+	+	+
Synodontis ocellifer	+	+	+

Table 13 (continued)

Synodontis schall	+	+	+
Synodontis sorex	+	+	+
Synodontis velifer	-	+	+
MORMYRIDAE			
Campylomormyrus tamandua	+	-	-
Hyppopotamyrus pictus	-	+	+
Hyperopesus bebe	-	+	+
Marcusenius abadii	-	-	+
Marcusenius senegalensis	-	+	+
Mormyrops anguilloides	+	+	+
Mormyrops breviceps	-	+	+
Mormyrus macrophthalmus	-	+	+
Mormyrus rume	-	+	+
Petrocephalus bovei	-	+	-
OSTEOGLOSSIDAE			
Heterotis niloticus	+	+	-
POLYPTERIDAE			
Polypterus birchir	+	-	-
Polypterus endlicheri	+	-	-
Polypterus senegalus	+	-	-

Table 13 (continued)

SCHILBIDAE			
Schilbe intermedius	-	+	-
Schilbe mystus	-	+	+
Siluranodon auritus	-	+	+
TETRAODONTIDAE			
Tetraodon lineatus	+	+	-

+ means present; - means absent

Plate I: *Synodontis sorex*

Plate J: *Heterobranchus bidorsalis*

Plate K: *Brycinus nurse*

Plate L: *Labeo coubie*

Plate M: *Labeo senegalensis*

Plate N: *Sarotherodon galilaeus*

The following fish families also dominated by weight: Cyprinidae (33.81 %) > Alestidae (14.28 %) > Mochokidae (10.48 %) in 2012. The results showed that there were more families, genera and species of fish recorded in 2011 than 2012 (Figure 19).

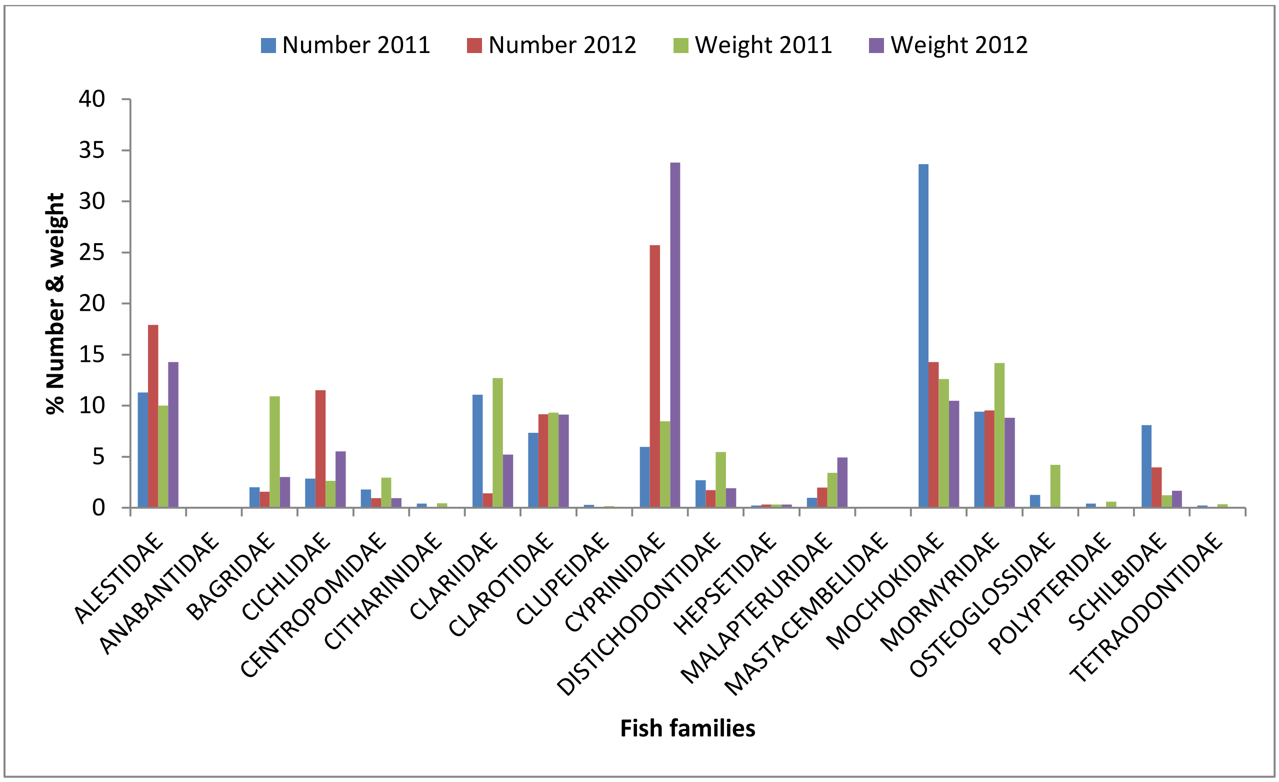


Figure 19: Fish family abundance by number and weight (kg) in 2011 and 2012

Seasonal abundance and diversity of fishes were also observed during the study period (Figure 20). The post-wet season recorded the highest fish catches, while the pre-wet recorded the lowest in 2011. By comparison, the dry season recorded the highest fish catches, while the post-wet season recorded the lowest in 2012. Diversity of fishes on the other hand, decreased from the post-wet season to the wet season in 2011, while the wet season recorded the highest diversity of fishes and post-wet season recorded the lowest in 2012.

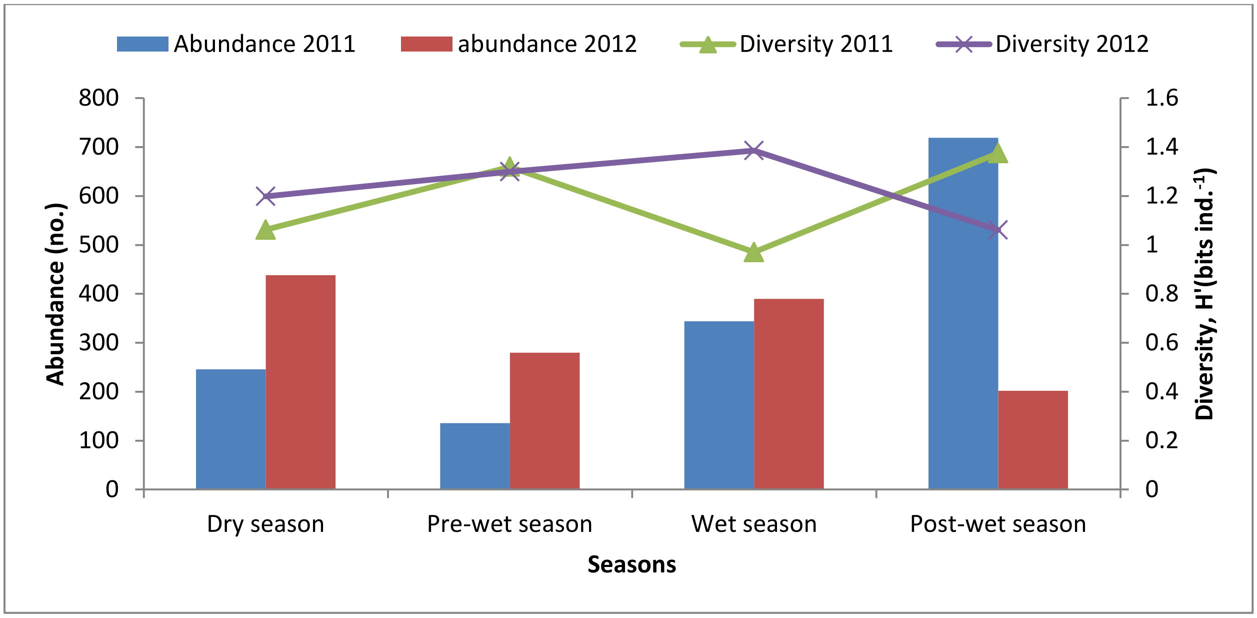


Figure 20: Seasonal variations in abundance and diversity of fish in 2011 and 2012

Table 14 shows the distribution of fish families during the pre- and post-impoundment periods with emphasis on the impact of the dam on the abundance of the fish families. Three families: Clarotidae, Distichodontidae and Mochokidae differed significantly (p < 0.05) during the pre- and post-impoundment periods indicating that the dam had impact on the abundance of only these three fish families during the study period (Figure 21).

Table 14: Fish family abundance by number during the pre- and post-impoundment periods (means ± standard error)

Families	Pre-impoundment (Mar - May 2011)	Immediate post-impoundment (June - Dec 2011)	Late post-impoundment (Jan - Dec 2012)	*P* value
ALESTIDAE	25.0 ± 1.6	56.5 ± 8.5	54.3 ± 4.9	**0.54**
ANABANTIDAE	0.0 ± 0.0	0.5 ± 0.5	0.0 ± 0.0	**0.25**
BAGRIDAE	10.5 ± 0..5	4.0 ± 0.0	5.0 ± 0.8	**0.34**
CENTROPOMIDAE	9.5 ± 0.5	3.5 ± 0.5	3.0 ± 0.4	**0.54**
CICHLIDAE	4.5 ± 0.5	16.0 ± 0.7	28.5 ± 1.5	**0.25**
CITHARINIDAE	0.0 ± 0.0	0.5 ± 0.5	0.0 ± 0.0	**0.18**
CLARIIDAE	0.0 ± 0.0	80.0 ± 7.2	4.5 ± 1.9	**0.21**
CLAROTIDAE	55.0^{a} ± 0.0	13.0^{b} ± 0.9	29.0^{b} ± 4.7	**0.02^{*}**
CLUPEIDAE	0.5 ± 0.5	1.5 ± 0.5	0.0 ± 0.0	**0.31**
CYPRINIDAE	22.5 ± 2.5	20.5 ± 2.5	81.5 ± 17.7	**0.09**
DISTICHODONTIDAE	17.0^{a} ± 3.0	2.5^{b} ± 0.1	5.5^{b} ± 0.9	**0.00^{*}**

Table 14 (continued)

HEPSETIDAE	0.5 ± 0.0	1.0 ± 0.1	1 ± 0.7	**0.89**
MALAPTERURIDAE	5.0 ± 1.0	2.0 ± 0.2	6.3 ± 0.4	**0.68**
MASTACEMBELIDAE	0.0 ± 0.0	0.5 ± 0.5	0.0 ± 0.0	**0.25**
MOCHOKIDAE	$47.0^{a} \pm 3.0$	$195.5^{b} \pm 32.5$	$45.3^{a} \pm 3.3$	**0.01***
MORMYRIDAE	2.0 ± 0.0	66.5 ± 8.5	30.3 ± 2.7	**0.29**
OSTEOGLOSSIDAE	0.5 ± 0.0	8.5 ± 1.5	0.0 ± 0.0	**0.26**
POLYPTERIDAE	3.0 ±0. 2	0.0 ± 0.0	0.0 ± 0.0	**0.08**
SCHILBIDAE	0.0 ± 0.0	58.5 ± 9.5	12.5 ± 1.7	**0.15**
TETRAODONTIDAE	1.5 ± 0.5	0.0 ± 0.0	0.0 ± 0.0	**0.25**

**** on the P values indicates significant differences (p < 0.05); figures on the same row with different superscript letters are also significantly different (p < 0.05) from one another. Hence there was impact of the dam with respect to some fish family abundance between the pre- and post-impoundment periods.***

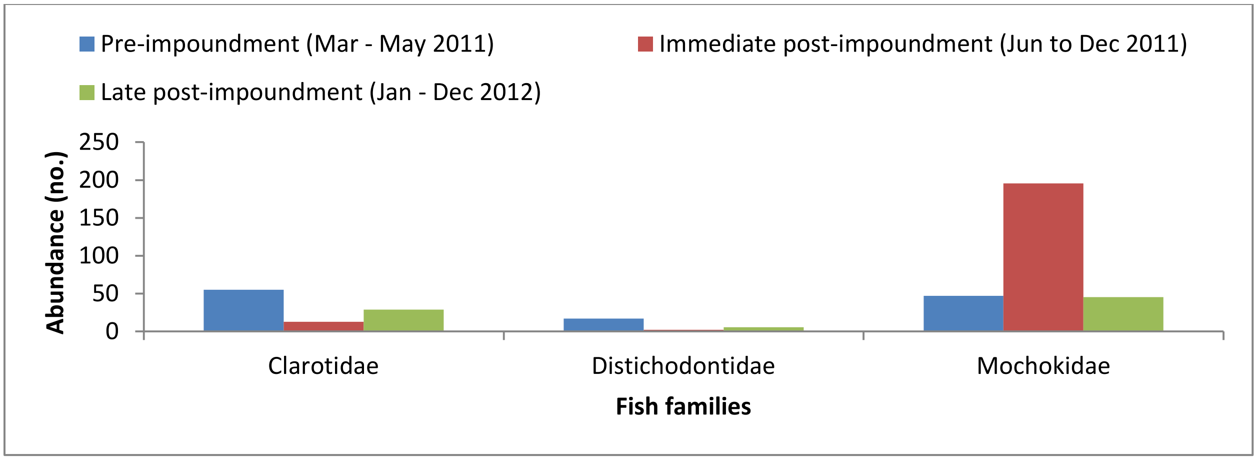


Figure 21: Variations in the abundance of fish families altered by the impoundment

4.4.2 Experimental gill net catches

A total number of six fish families, nine genera and thirteen species were recorded in experimental gill nets during the study period. In order of importance by number, the fish species followed this order: *Brycinus nurse* (16.3 %) > *Alestes dentex* (13.04 %) > *Synodontis nigrita* (11.95 %). By weight the species were dominated by *Clarias gariepinus* (14.02 %) followed by *Mormyrops anguiloides* (12.86 %) and *Alestes dentex* (12.54 %) (Figure 22). From the above, the fish species abundance by number and weight were lower in experimental catches compared to the commercial catches. Figure 23 shows the percentage composition of fish catches by mesh sizes in experimental gill nets. The fish catches decreased in number as the mesh size increased from 15 mm to 25 mm. Thus the mesh sizes that made the highest catches by number followed this order: 15 mm (33.69 %) > 17.5 mm (24.46 %) > 20 mm (15.76 %). By comparison, the mesh sizes that made the highest catches by weight followed this order, 25 mm (30.23 %) > 17.5 mm (28.06 %) > 15 mm (15.24 %).

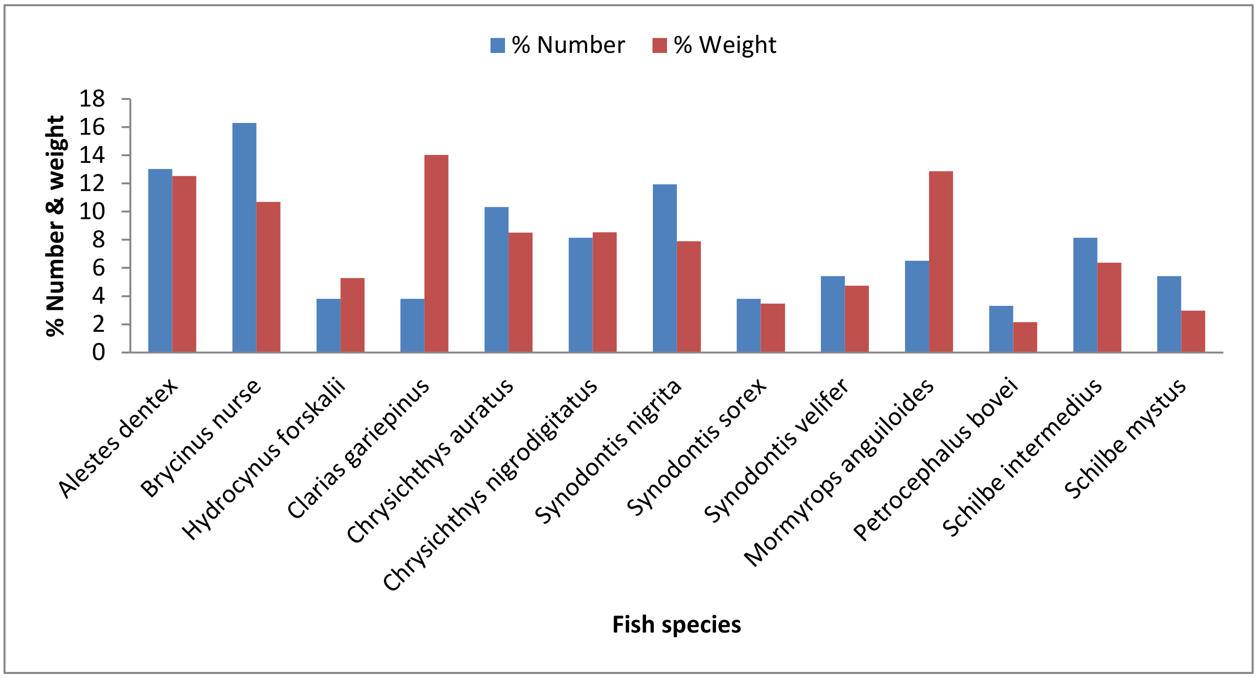


Figure 22: Fish species abundance by number and weight (kg) in experimental gill net catches

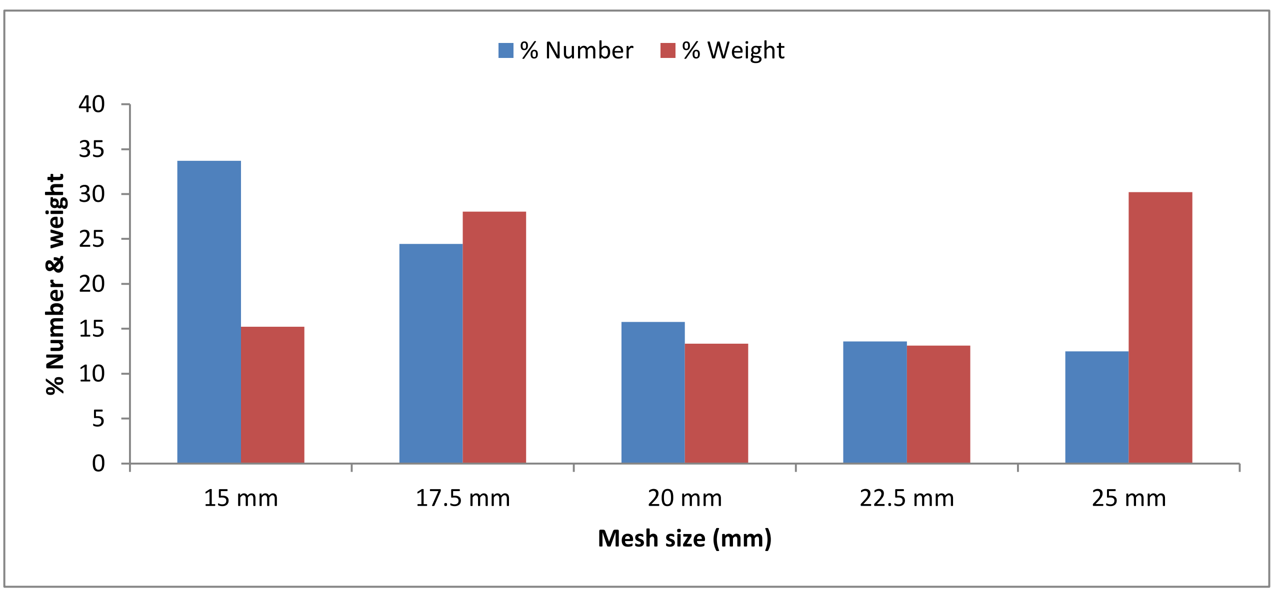


Figure 23: Composition by number and weight (kg) of fish catches by mesh sizes in experimental gill nets

4.4.3 Estimation of mean Catch per Unit Effort and total catch

Figure 24 shows the mean monthly variations in CPUE and total catch for canoes utilizing gill nets during 2011 and 2012. Mean monthly CPUE of fishes were lowest in August (0.71 kg/canoe/day) and highest in March (11.03 kg/canoe/day) with an annual mean of 5.16 kg/canoe/day in 2011. CPUE were highest in February (17.92 kg/canoe/day) and lowest in March (6.8 kg/canoe/day) with an annual mean of 10.29 kg/canoe/day in 2012. Mean monthly total catch of fish decreased from March (5361 kg) to August (37.3 kg) and a mean of 2568 kg in 2011. In 2012 however, total catch decreased from February (8064 kg) to March (3305 kg) and a mean of 5160 kg. Assuming 50 % of all local canoes utilizing gill nets were active, a total fish catch of 26748 kg or 26.75 metric tonnes was estimated for 2011 and 58655 kg or 58.65 metric tonnes for 2012 and a mean of 42702 kg or 42.70 metric tonnes for the whole study period.

Seasonal variations in CPUE levels were also observed during the study period. CPUE was lowest (3.4 kg/canoe/day) in the wet season and highest (10.59 kg/canoe/day) during the dry season in 2011. By comparison, in 2012 CPUE was lowest (8.42 kg/canoe/day) in the post-wet season and highest (12.52 kg/canoe/day) in the wet season. The mean estimated CPUE for the 2 years (2011 & 2012) was lowest (6.23 kg/canoe/day) in the post-wet season and highest (10.86 kg/canoe/day) in the dry season with a mean of 7.95 kg/canoe/day. Hence, the dry season was the most important season for catches, while the post-wet season was the lean season in the study area. CPUE also differed significantly ($p < 0.05$) between the pre- and post-impoundment periods indicating the dam had impact on fish production as measured by CPUE levels (Figure 25). Hence, the alternative hypothesis that fish production will change following the impoundment of the Black Volta by the Bui dam was accepted and the null rejected.

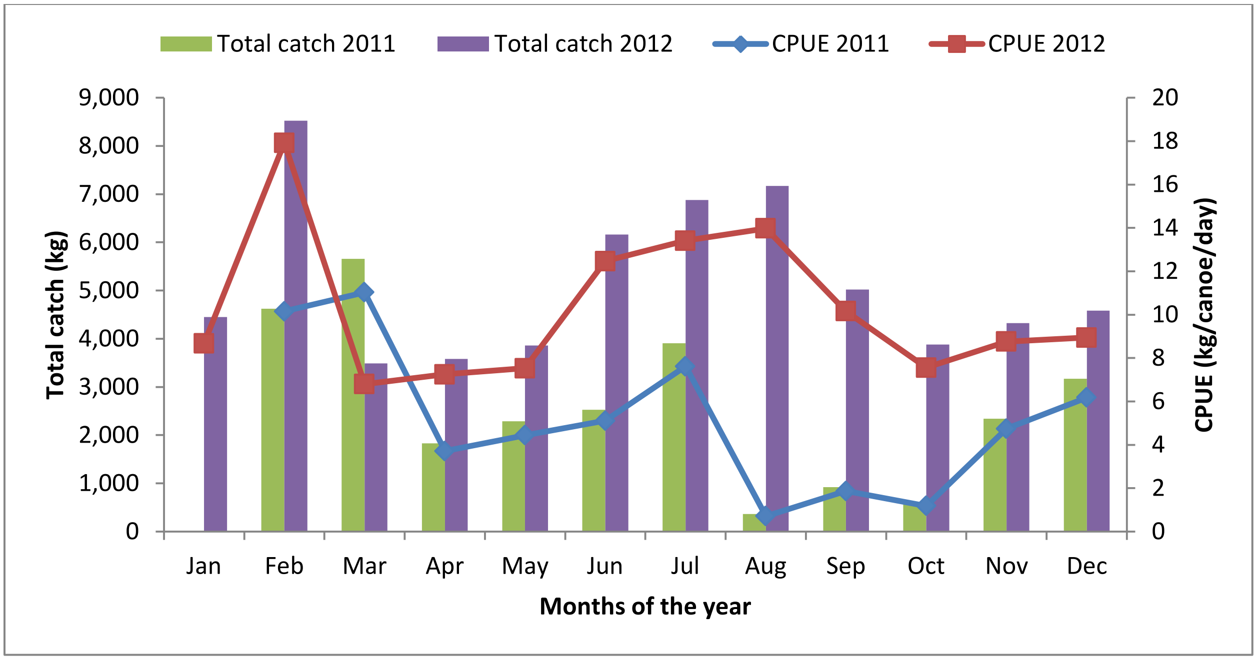


Figure 24: Mean monthly variations in CPUE and total catch of canoes utilizing gill nets in 2011 and 2012

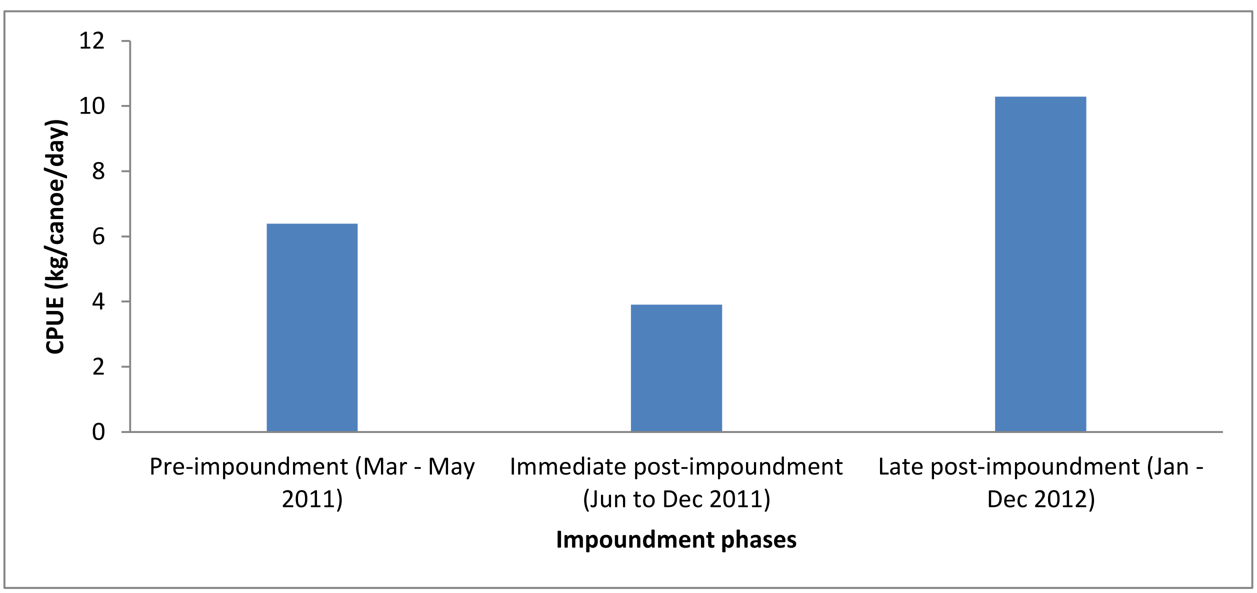


Figure 25: Variations in CPUE levels during the pre- and post-impoundment periods

4.4.4 Estimation of Potential fish yield of the Bui reservoir

Applying the equation used in estimating potential fish yield to the Bui reservoir during its formative period (June 2011 to December 2012) at the mean surface conductivity of 101.49 μScm^{-1} at a measured mean depth of 2.29 m, the MEI = 0.44. The potential fish yield (Y) was thus computed as: 23.281 x $0.44^{0.447}$ kg $ha^{-1}yr^{-1}$ or 16.13 $kgha^{-1}yr^{-1}$. This was found to be less than most reservoirs in Africa.

4.5 Relationship between Hydro-biological factors and Fish Production

4.5.1 Primary Productivity and Fish Production

Figure 26 shows the relationship between chlorophyll *a* and CPUE. The coefficient of determination, R^2 of 0.8545 indicates that 85 % of the variation in fish production as measured by CPUE was explained by primary productivity as measured by chlorophyll *a* concentration. Thus, when primary productivity was high, fish production was also high and vice versa. The regression analysis indicated that there was a significant (p = 0.000) relationship between both variables. Thus, knowing chlorophyll *a* concentration could be useful in predicting fish production as measured by CPUE levels.

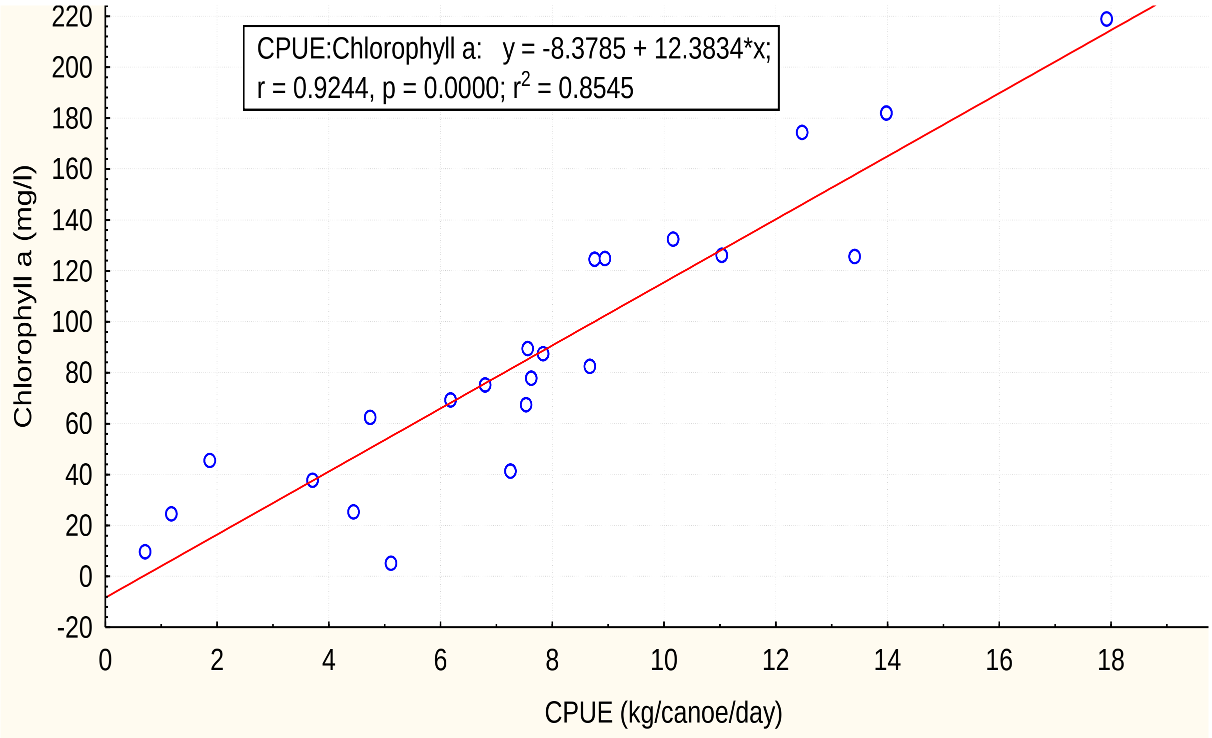


Figure 26: Relationship between chlorophyll *a* and CPUE

4.5.2 Phytoplankton densities and Fish Production

Figure 27 shows the relationship between phytoplankton densities and CPUE. The coefficient of determination, R^2 of 0.0395 indicates that 3.9 % of variation in the CPUE was explained by phytoplankton densities. There was no significant (p = 0.3631) relationship between both variables. Thus, knowing phytoplankton densities does not help in predicting fish production as measured by CPUE levels.

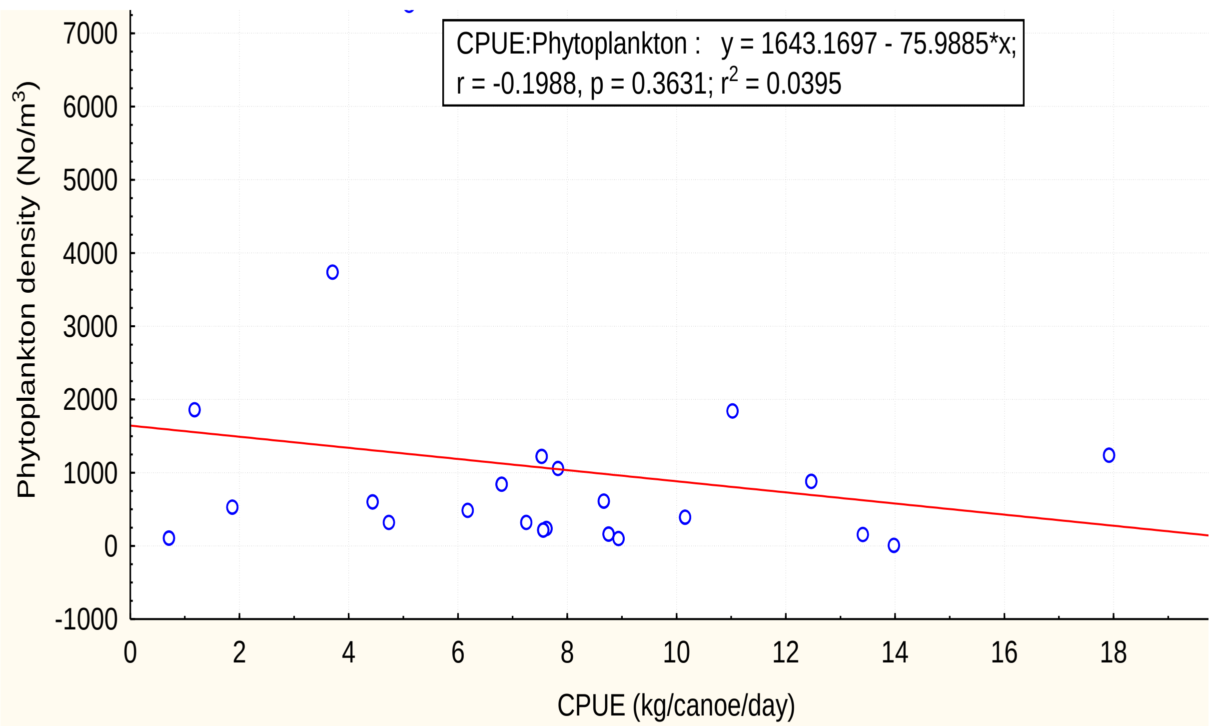


Figure 27: Relationship between phytoplankton densities and CPUE

4.5.3 Zooplankton densities and Fish Production

Figure 28 shows the relationship between zooplankton densities and CPUE. The coefficient of determination, R^2 of 0.099 indicates that 9.9 % of the variation in CPUE was explained by zooplankton densities. There was no significant (p = 0.1434) relationship between both variables. Thus, higher zooplankton densities do not help in predicting fish production as measured by CPUE levels.

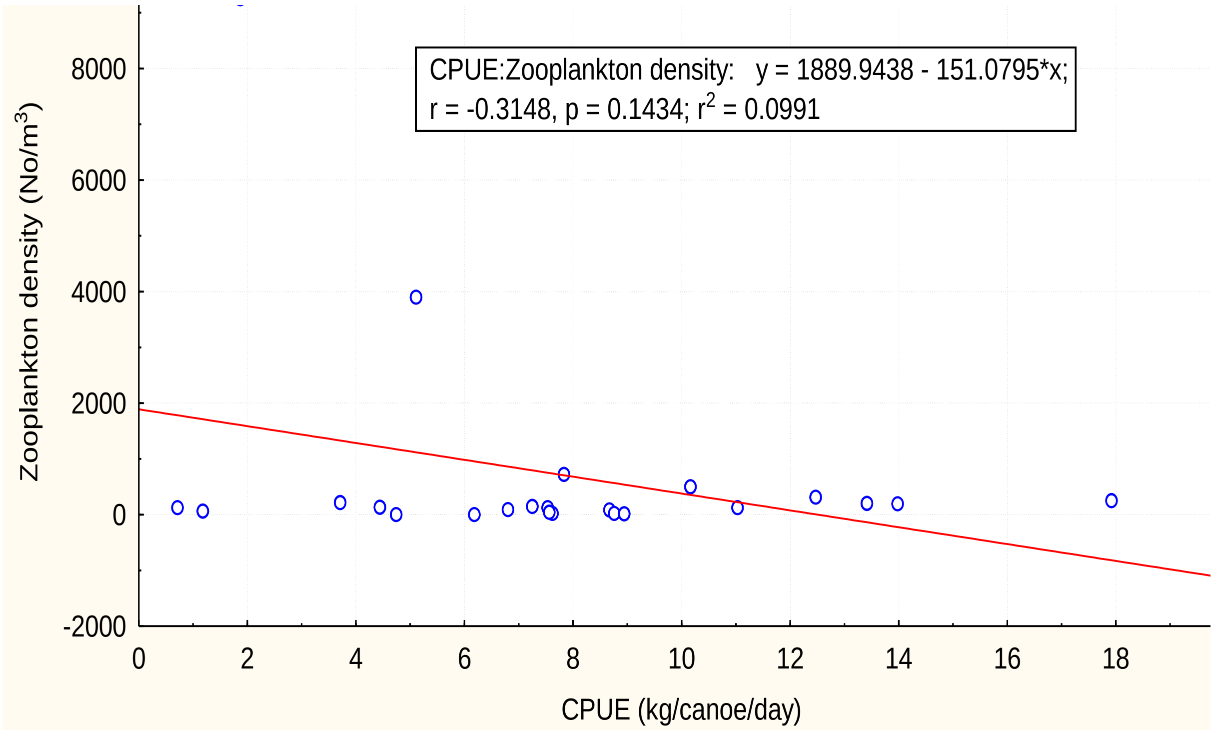


Figure 28: Relationship between zooplankton densities and CPUE

4.5.4 Flood regime and Fish Production

There was an inverse relationship between the flood regime as measured by water level and fish production (Figure 29). Thus, when water level increased fish production on the other hand, decreased and vice versa. The coefficient of determination, R^2 of 0.3168 indicates that 32 % of the variation in CPUE was explained by water level. There was a significant (p = 0.0052) relationship between both variables. Hence, knowing water levels could lead to the prediction of fish production as measured by CPUE levels.

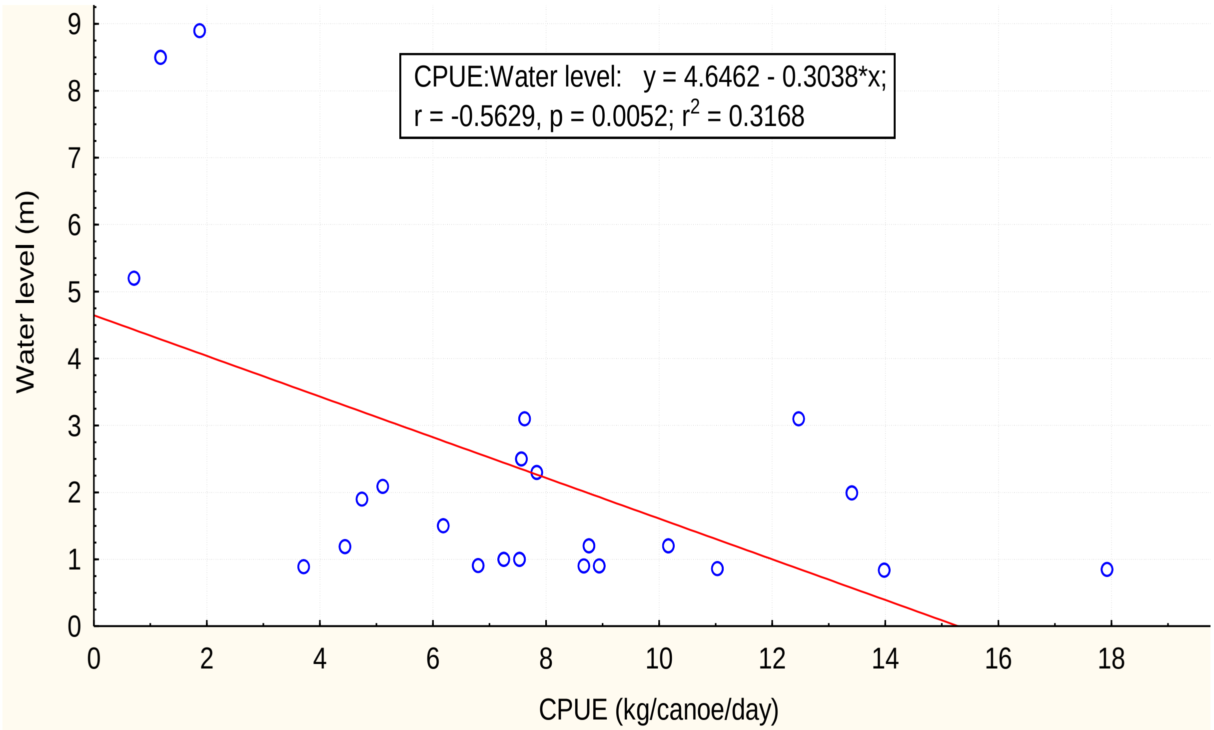


Figure 29: Relationship between water level and CPUE

4.5.5 Predictive fish catch Model

Figure 30 shows a plot of multiple variables (water level and chlorophyll *a*) against CPUE. Predictor variables that significantly explained CPUE levels were chlorophyll *a* (positive correlation) and water level (negative correlation) (p = 0.0002). The coefficient of determination, R^2 of 0.906 indicates that 91 % of the variations in CPUE were explained by water level and chlorophyll *a* concentration. Hence, knowing water level and chlorophyll *a* concentration, fish production as measured by CPUE levels could be predicted. The best, most prudent simple regression model to predict CPUE levels was described as:

CPUE = - (0.456 x water level) + (0.062 x chlorophyll *a*) + 3.363

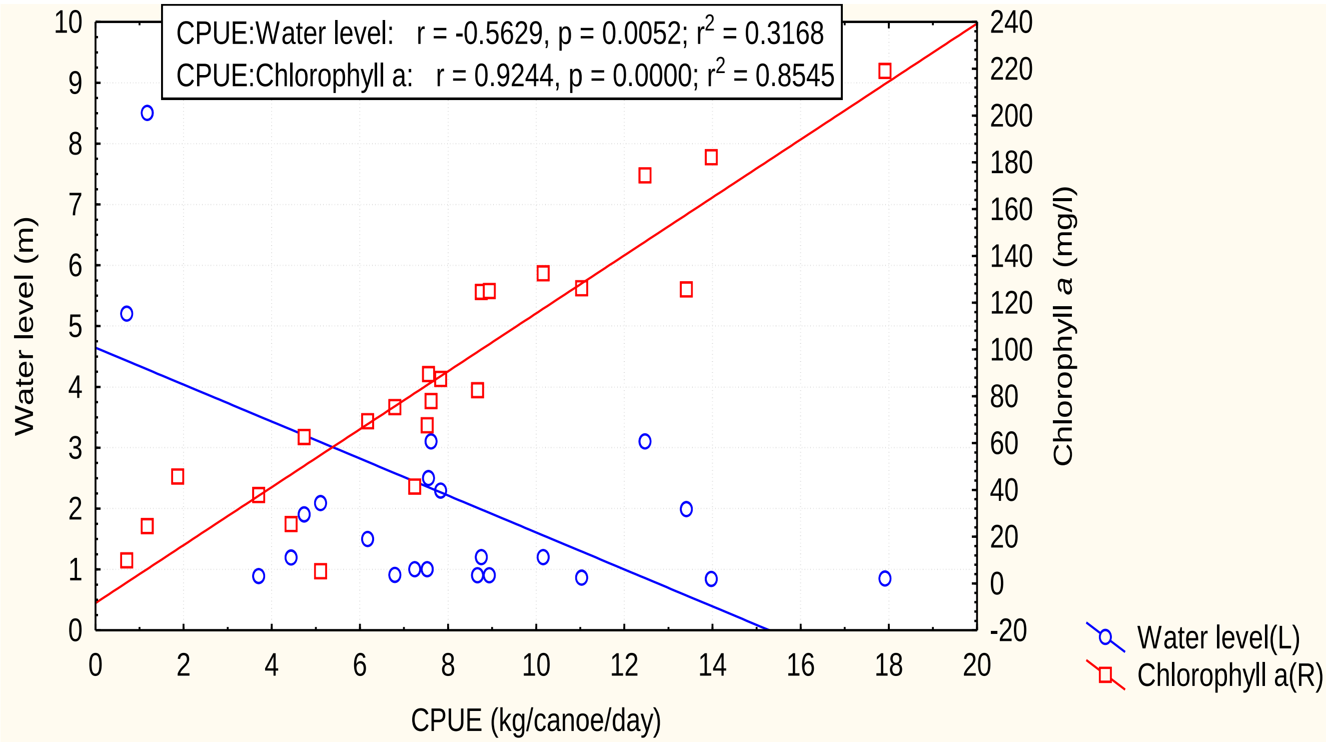


Figure 30: Effect of water level and chlorophyll *a* on CPUE

The standard error and significance levels of the constant, water level and chlorophyll *a* in the predictive model are shown in Appendix XVI. The R-square value is an indicator of how well the model fits the data (e.g., an R-square close to *1.0* indicates that almost all of the variability with the variables specified in the model was accounted for). For regression models, standardized regression coefficients are used as direct measures of model sensitivity. The use of the regression technique allows the sensitivity ranking to be determined based on the relative magnitude of the regression coefficient. Hence, the regression coefficient, R^2 = 0.906 in this model accounted for almost 91 % of the CPUE variability and therefore confirms the predictive power of this proposed model for estimating fish production as measured by CPUE levels.

The regression line expresses the best prediction of the dependent variable (CPUE), given the independent variables (water level & chlorophyll *a*). However, nature is rarely (if ever) perfectly predictable, and usually there is substantial variation of the observed points around the fitted regression line. The deviation of a particular point from the regression line (its predicted value) is called the residual value. The smaller the variability of the residual values around the regression line relative to the overall variability, the better the prediction. Residual analysis indicated that this model was adequate for describing CPUE since there was no trend in residuals along the CPUE gradient (Figure 31). This is therefore an indication that the resulting model is adequate to predict CPUE levels in the Black Volta near the Bui dam.

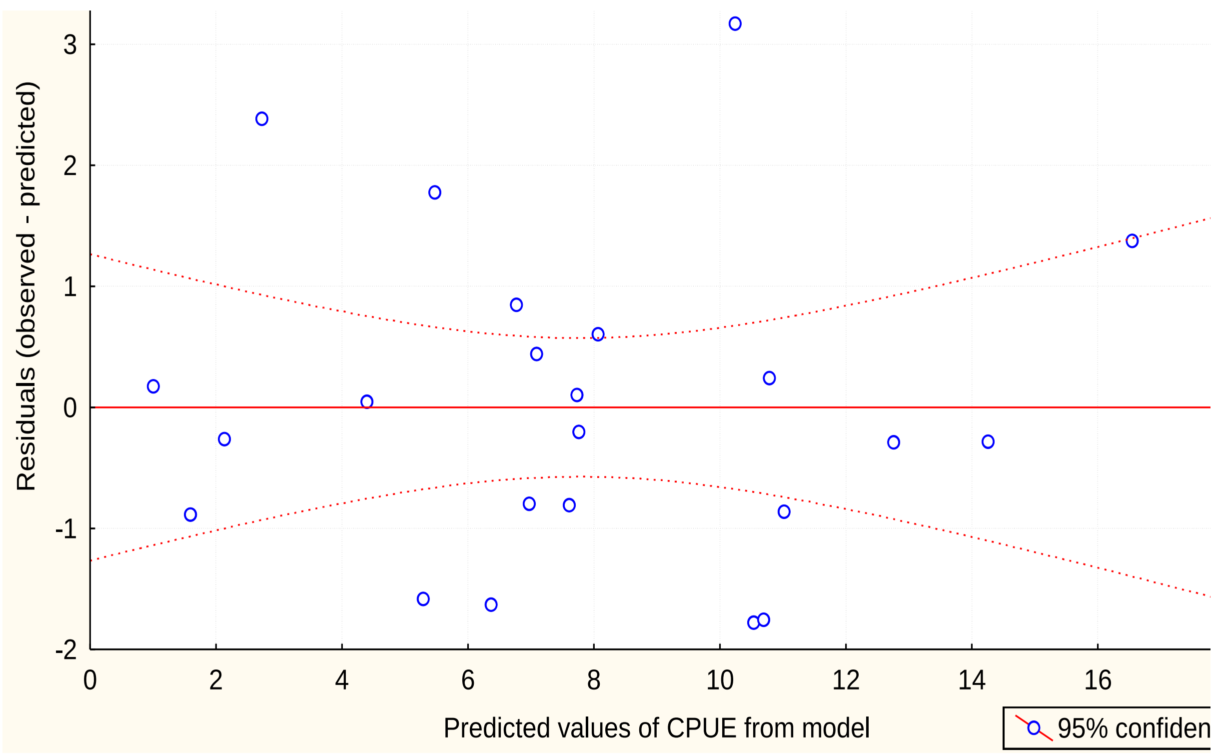


Figure 31: Relationship between residuals and predicted values of CPUE

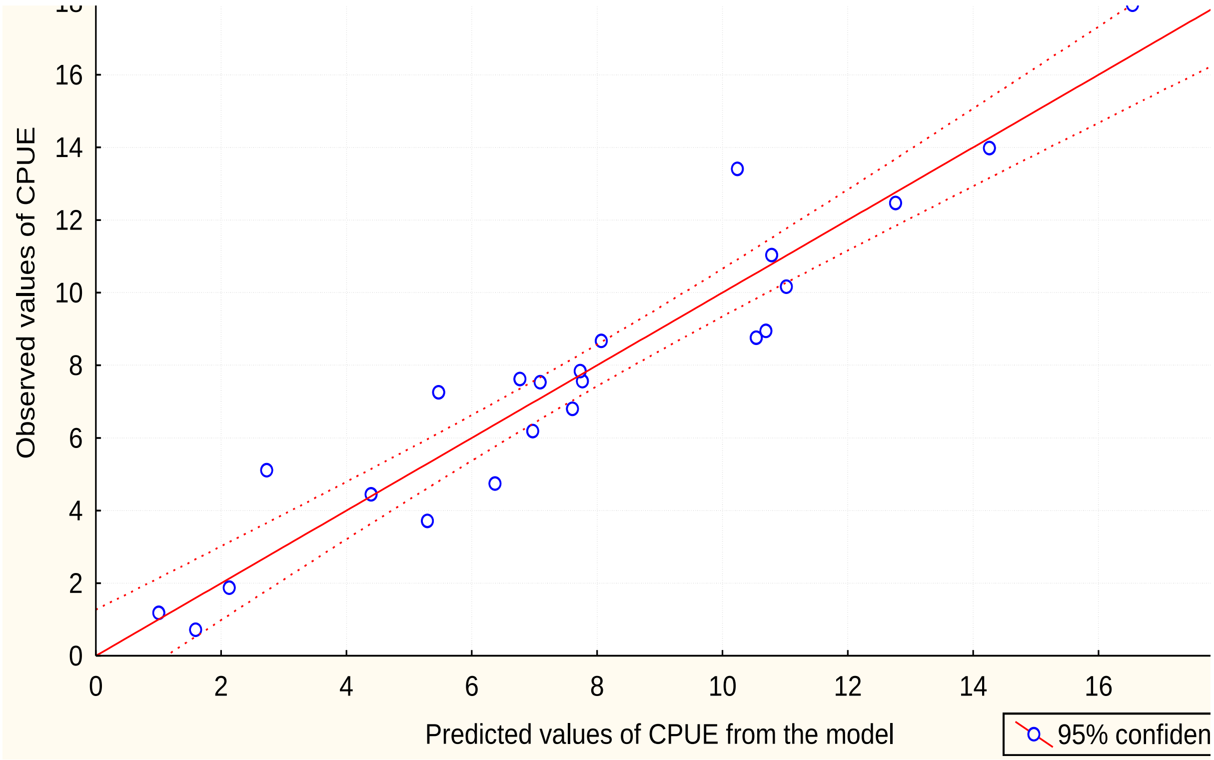


Figure 32: Relationship between CPUE predicted through the model and those observed in 2011 & 2012

To test the predictive power of the model, observed values of CPUE from the Black Volta during 2011 and 2012 were used. A significant relationship (R^2 = 0.908) was observed between CPUE predicted through the model and the CPUE measured independently in 2011 & 2012 (Figure 32). Therefore, independent validation also indicated that the model had potential to predict CPUE (as a measure of fish production) in the Black Volta near the Bui dam.

CHAPTER FIVE

5.0 DISCUSSION

5.1 Physico-chemical Characteristics

Building a dam across a river, and impounding water behind it, may cause profound changes in the limnological regime and biological productivity of the water body. These may include chemical and physical changes which may in turn affect the flora and fauna of the regulated water body (Egborge, 1979; Reynolds, 1997). The Black Volta near the Bui dam showed narrow differences in temperature during the four hydrological seasons, namely dry, pre-wet, wet and post-wet seasons. This could be due to the fact that river water showed little thermal stratification because of the turbulent flow which ensures that any heat received is evenly distributed. Aquatic organisms have their own tolerance limits to temperature and this affects their distribution and migration. The temperature range of 24.6 – 33 ºC recorded in this study was favourable and could support fisheries and aquatic life and similar to earlier findings (29.8ºC – 31.7 ºC) by Petr (1970) in the Black Volta. According to Alabaster and Lloyd (1980), the temperature of natural inland waters in the tropics generally varies between 25 – 35 ºC. Similar observations were made on the Volta Lake by Ofori-Danson and Ntow (2005).

Freshwaters with little change in pH are generally more conducive to aquatic life. The pH levels encountered in this study (6.9 – 8.3) were considered suitable for fish growth since the best pH values for the survival of fish in freshwaters have been reported to range from 5 – 9 (Jobling, 1995). Petr (1970) made similar observations for pH values of 7.7 – 8.6 in the Black Volta.

Abowei (2010) however noted that, pH higher than 7 but lower than 8.5 is ideal for biological productivity, while pH lower than 4 is detrimental to life in freshwaters.

Oxygen is essential to all forms of aquatic life as it affects the growth, survival, distribution and physiology of aquatic organisms (Solis, 1988). Oxygen concentration in water is controlled by four factors: photosynthesis, respiration, exchanges at the air-water interface, and supply of water to the water body (Erez *et al*., 1990). The mean DO levels of 3.37 mgl^{-1} in 2011 and 4.78 mgl^{-1} in 2012 were below the 5 mgl^{-1} threshold needed to support fish life in freshwaters (Hynes, 1970; Chapman and Kimstach, 1996). This could be due to the continuous input and decomposition of allochthonous detritus which tends to decrease the oxygen concentration even in periods of autotrophy (Carvalho *et al*., 2001).

There was a negative correlation ($r = -0.79$) between dissolved oxygen and temperature. According to Wetzel (2001), there is a direct effect of temperature on the solubility of gases, in particular dissolved oxygen. In addition, there is a direct temperature effect on decomposition rates due to microbial activity. Hence an increase in temperature can cause a two-fold to three-fold increase in bacterial activity, and consequently decreased dissolved oxygen concentrations (Rocha *et al*., 2009). The negative effect of temperature on oxygen concentration as reflected in this study corroborates other studies that showed that high temperatures found in tropical aquatic ecosystems are responsible for rapid detritus breakdown (Carvalho *et al*., 2005).

The low level of dissolved oxygen (2.27 mgl^{-1}) recorded during the immediate post-impoundment period indicated deteriorating water quality which probably resulted from the death and decay of aquatic macrophytes, increased active organic decomposition in the bottom sediment and the absence of flow induced turbulence which normally enhance oxygen dissolution in water (Ogbeibu and Oribhabor, 2002).

The conductivity of water is dependent on its ionic concentrations and temperature. It provides a good indication of the changes in water composition particularly its mineral concentration. There is also a relationship between conductivity and TDS in water. As more dissolved solids are added, the water's conductivity increases (Abowei *et al*., 2010). This scenario of direct relationship between conductivity and TDS ($r = 0.997$) was observed throughout the four hydrological seasons in this study.

Morpho-edaphic index (MEI), which is estimated by dividing the mean conductivity by the mean depth (Ryder *et al.,* 1974) of the Bui reservoir, was 0.44. This was however lower than the MEI of most reservoirs in Africa. Henderson and Welcomme (1974) recorded an MEI of 9.2 for the Nasser-Nubia reservoir in Egypt and 6.6 for Lake Kainji in Nigeria, Marshall (1984) recorded an MEI of 2.8 for Lake Kariba in Zimbabwe and Ofori-Danson (1999) recorded an MEI of 4.51 for Lake Volta in Ghana. The higher MEI values may be due to the large surface areas of these large reservoirs in Africa. The low MEI in this study on the other hand, could probably be due to the fact that the Bui reservoir had not yet stabilised during the study period.

There were high nutrient levels in the Black Volta near the Bui dam. Nitrate levels exceeded optimum considering the global average of 0.1 mgl^{-1} in freshwater (Meybeck and Helmer, 1989). The positive correlation ($r = 0.27$) between nitrates and dissolved oxygen in this study could be explained by the dependence of the nitrification process on oxygen supply (Wetzel, 2001).

The concentrations of sulphates also exceeded the average of 4.8 mgl^{-1} for freshwaters (Meybeck and Helmer, 1989). Mean seasonal sulphate concentrations measured in this study was 19.05 mgl^{-1} and 12.24 mgl^{-1} in 2011 and 2012 respectively. It may thus be inferred that sulphate was abundant in the Bui dam area the Black Volta. Though the optimum ranges of phosphate in freshwater is 0.005 – 0.05 mgl^{-1} (Dunne and Leopold, 1978), mean seasonal phosphate levels recorded in this study were 0.09 mgl^{-1} in 2011 and 0.06 mgl^{-1} in 2012. The concentration of phosphates in uncontaminated waters is reported to be about 0.01 mgl^{-1} (McNeely *et al*., 1979). The reasons for these high nutrient levels in the Bui dam area of the Black Volta was not clearly known in this study. It could however, be due to the breakdown of submerged dead plant materials as a result of the impoundment of the Black Volta. These high nutrient levels could however, be comparatively short-lived, and therefore requires further investigation by limnologists.

5.2 Hydro-biological Characteristics

Phytoplankton constitutes the primary producers of aquatic ecosystems and serves as bases for food chain that supports the commercial fisheries. Phytoplankton communities are also the major producers of organic carbon in large rivers, a food source for plankton consumers and may represent the primary oxygen source in low-gradient rivers (Wehr and Descy, 1998).

A total of thirty-five species of phytoplankton belonging to four classes were recorded during the studies. The number of species per class were dominated by Chlorophyceae (17 species), Cyanophyceae (12 species), Bacillariophyceae (4 species), and Euglenophyceae (2 species). The distribution pattern of phytoplankton during the study showed that all the species, except *Euglena sp.* and *Phacus pyrum* (Euglenophyceae) were fairly distributed in the four hydrological seasons. The lotic nature of the study area may be responsible for the relatively uniform seasonal distribution. Zabbey *et al*. (2008) reported that in lotic ecosystems, continuous flows ensure near uniform distribution of organisms. The impoundment of the Black Volta may, however, be the principal factor responsible for the discontinuous seasonal distribution of *Euglena sp* and *Phacus pyrum* observed in the study. Their absence during the post-impoundment period indicates that they could possibly not be adaptable to the impoundment-induced environmental changes.

The results also showed that variations occurred between seasons and in some phytoplankton groups. The dry and pre-wet seasons had significantly higher mean phytoplankton abundance than the wet season during the two-year sampling period. Phytoplankton abundance is associated with seasonal differences in flow. Densities of phytoplankton usually reach the peak in the dry season and diminish in the floods, unless otherwise influenced by temperature (Welcomme, 1985). In the Kafue River in Zambia, phytoplankton densities were found to be less during the floods and dense blooms occurring between August and November when the floods had receded

(Carey, 1971). Seasonal variations in phytoplankton abundance with higher values in the dry season have also been reported by Erondu and Chinda (1991), Ogamba *et al.* (2004) and Emmanuel and Onyema (2007).

There was also a significant variation in the mean values of phytoplankton abundance between 2011 and 2012. This may be due to the impoundment (or blocking of the normal flow) of the Black Volta in June 2011. This is because there was not continuous flow of water which might prevent the near uniform distribution of phytoplankton (Zabbey *et al*., 2008).

The CCA ordination showed that many phytoplankton species had affinity for higher nutrient concentrations, e.g. *Anabaena sp*, *Pseudoanabaena sp*, *Planktothrix sp* and *Scenedesmus sp* were clustered more specifically with nitrates. Nitrates and/or phosphates are considered the limiting nutrients in many aquatic environments (Sakka *et al*., 1999). In general, the CCA ordination plots showed a pattern where nitrates, phosphates, temperature, DO, conductivity and TDS and many phytoplankton species were correlated. The study therefore infers that in addition to hydro-biological factors (e.g. chlorophyll *a* concentration) changes in physico-chemical variables affected the composition of the phytoplankton community.

The biotic indices of Margalef, Pielou and Shannon-Wiener were fairly distributed in the four hydrological seasons of the two years. This indicated that phytoplankton community was stable among seasons of the study area. Minimal variations in the density of phytoplankton as reflected by Shannon-Wiener (H´ ($\log_{10}$), Pielous evenness and Margalef species richness may be attributed to uniform physical and chemical conditions (Ogamba *et al*., 2004). This may be attributed to the continuous flow in lotic ecosystems (Zabbey, 2008). According to Williams (1964), species diversity indices often reflect the impact of pollution on aquatic communities and pollution affects the distribution, standing crop and chlorophyll *a* concentration of phytoplankton (Barnes, 1980). The species in this study area were distributed in all four hydrological seasons. This therefore suggests that at present pollution has minimal effect on the Black Volta near the Bui dam. Thus the phytoplankton can support a good fishery, because phytoplankton are the major producers and a food source for plankton consumers (Wehr and Descy, 1998) and as well serve as the bases of food chain that supports commercial fisheries (Conde *et al*., 2007).

Zooplankton organisms are important components of the aquatic ecosystem and are considered indicators of water quality (Pinto-Coetuo *et al*., 2005) and fisheries health because they are a food source for organisms at higher trophic levels (Davies *et al*., 2009). The nature of species composition, diversity and seasonal abundance of zooplankton differ in water bodies (FAO, 2006). A total of sixteen species belonging to the Subclass Copepoda and Order Cladocera were recorded in the Bui dam area of the Black Volta. The low zooplankton densities recorded in this study corroborate the findings of Nilssen (1984) who reported that zooplankton communities are usually simplified, with low densities in tropical freshwaters. The dominance of Cladocera during the entire study period compares favourably with the findings of other workers (Yakubu *et al*., 1998; Tackx *et al*., 2004). Most of the zooplankton encountered in the study area appeared to be normal inhabitants of natural lakes, rivers and artificial impoundments in the tropics and subtropics (Jeje and Fernando, 1986; Aneni and Hassan, 2003; Ayodele and Adeniyi, 2005;

Okogwu and Ugwumba, 2006). There were seasonal variations of zooplankton groups during the two years of sampling. The mean values showed higher wet season than post-wet and dry season variations. This was in contrast with the findings of (Egborge, 1994; Onyema, 2007; Nkwoji *et al*., 2010a; and Okogwu, 2010) who reported that seasonal pattern of zooplankton densities in Nigerian freshwaters peak in the dry season. However, in consonance with this investigation, Muylaert *et al*. (2003) and Arimoro and Oganah (2010) reported that zooplankton abundance usually reach their peak during the wet season. This may be ascribed to the rains bringing in allochthonous nutrients from the drainage basin that will accelerate primary production resulting in zooplankton production and abundance (Evans *et al*., 1993; Okogwu and Ugwumba, 2006). The high population density of zooplankton during the wet season in this study was also traced to high population of phytoplankton food source which were abundant during the pre-wet season. According to Rocha *et al*. (1999), increase in phytoplankton abundance tends to be followed by increase in zooplankton densities.

There was also relatively low zooplankton abundance during the post-wet and dry seasons of the study period and more especially during 2011. Low availability of food source could be responsible for the decline in zooplankton during the post-wet and dry seasons (Achembach and Lambert, 1997). Oxygen is known to have a negative impact on zooplankton at levels below 2.5 mgl^{-1} (Aka *et al*., 2000). In response to low oxygen concentration, most zooplankton including cladocerans increase haemoglobin synthesis to enhance their oxygen extraction efficiency and survival rate (Hanazato and Dodson, 1995). This probably explains the absence of cladoceran population during the post-wet season of the 2011 sampling year when mean dissolved oxygen was low (1.77 mgl^{-1}).

The CCA ordination indicated that zooplankton organisms responded to a number of physico-chemical variables and 41 - 64 % of the variations in zooplankton densities were accounted for by the measured physico-chemical variables. The correlation coefficient indicated that several physico-chemical factors exert a considerable influence on the zooplankton abundance, especially DO, pH, TDS, conductivity, temperature, phosphates and sulphates. Consistent with the findings of this study, Sarkar and Choudhury (1999) reported significant multiple correlations between zooplankton abundance and several physico-chemical variables. The correlations of the zooplankton with phosphates and sulphates may not necessarily be a direct relationship of the zooplankton utilising the nutrients, but could be attributed to the dependence of the phytoplankton (which serves as food for the zooplankton) on these nutrients (Mustapha, 2009).

DO, conductivity, pH, TDS and phosphates have been found to be important to zooplankton in other tropical studies (Ogbeibu, 1998; Arora and Mehra, 2003; Pandey and Verma, 2004; Okogwu and Ugwumba, 2006). Size, structure and biomass of plankton population are closely related to physico-chemical conditions of the water body (Mitchell-Innes and Pitcher, 1992). Environmental factors such as dissolved oxygen, temperature, pH, nitrates and phosphates are reported to marshal the activities and composition of organisms (Collins, 1983) and their abundance and diversity reflect the physico-chemical conditions of aquatic ecosystem in general and its nutrient status in particular (Anene, 2003). Due to the strong interaction between the zooplankton abundance and physico-chemical variables, factors such as climatic changes and/or dam construction that will modify the flooding pattern of the river will inadvertently alter the zooplankton community structure in the newly created Bui reservoir and may have serious implications for fish production of the entire Black Volta ecosystem (Okogwu *et al*., 2009).

With the exception of the post-wet season, the biotic indices of Margalef's species richness, Pielou's evenness and Shannon-Wiener diversity were fairly distributed in the first three seasons of 2011. This was attributed to the fact that the post-wet season was between three and six months after the impoundment of the Black Volta and that the reservoir had not yet stabilised hydro-biologically. The higher variability in the biotic indices that were observed in the wet and post-wet seasons could also be a reflection of community instability in these seasons. This confirms changes in species composition, density and diversity measures caused by river impoundment. There was however, a fair distribution of these biotic indices during the 2012 sampling year indicating that the zooplankton community structure between eight and twenty months after the impoundment began to stabilise among the four hydrological seasons.

Four macrobenthic invertebrate taxa were identified during the study period. The overall community composition of the study stretch is comparable to earlier records for the Black Volta where chironomids have always dominated (Petr, 1970; Samman *et al*., 1992). All the taxa recorded in this study were also identified by earlier workers (Petr, 1970; Samman *et al*., 1992; Gordon *et al*., 2003) in the Black Volta. Gordon *et al*. (2003) did not, however, record any Tricoptera in their samples.

There was generally low macrobenthic invertebrate diversity in the Black Volta and this compares favourably with Nkowji *et al*. (2010b) who reported low values of Margalef's species richness and Shannon – Wiener diversity index in the Sombreiro River in Nigeria. The low diversity of macro-invertebrates could be ascribed to the high electrical conductivity of the water in this study. This therefore corroborates the findings of earlier workers (Bere and Tundisi, 2009; Ezekiel *et al*., 2011; Habeeba *et al*., 2012) who established a negative correlation between electrical conductivity and benthic macro-invertebrate abundance. Hence, when electrical conductivity was high, macro-invertebrate diversity was low and vice-versa.

5.3 Fish Production

A total of sixty-four fish species (out of ninety-six recorded for the Black Volta by Vanden Bosche and Bernacsek, 1990) belonging to thirty-eight genera and twenty families were recorded during the 2-year sampling period. This represented 66.7 % of recorded fish species for the Black Volta, suggesting high fish species diversity in the Bui dam area. The number of species recorded in this study was, however, higher than that recorded by earlier workers (Bennett and Basuglo, 1998; Gordon *et al*., 2003) in the same area. This could probably be due to the length of stretch (37.5 km) of the sampling along the Black Volta and relatively long duration of the sampling period (two years) of this study.

There were differences in fish family dominance (by number and weight) during the 2 years of sampling. The change from riverine to lacustrine conditions during the formation of the reservoir, led to the immediate reduction in the numbers of a variety of fish families, including

Centropomidae, Clarotidae and Distichodontidae which are very sensitive to oxygen depletion. The basic trend in this study was towards the development of a community of fish species which had vegetarian food habits such as *Sarotherodon galilaeus*, *Labeo coubie* and *Labeo senegalensis*. A similar trend was observed by Petr (1969) following the initial 2 years of Lake Volta's formation. Petr (1969) however, recorded a reduction in the numbers of insectivores such as *Alestis dentex* and *Brycinus macrolepidotus* in Lake Volta. These differences in fish dominance in the two sampling years of the present study could also be attributed to the fact that the fish distribution was not yet uniform, possibly because the impoundment was still in the interphase between the lacustrine and riverine conditions. This same phenomenon was also observed after the formation of Lake Kariba on the Zambezi River between Zambia and Zimbabwe (Jackson, 1961). The abundance of *S. galilaeus* in the 2012 sampling year was probably due to the establishment of a large static water body during the post-impoundment period (Petr, 1969).

Differences in seasonal abundance were also observed during the study period. The post-wet season recorded the highest numerical abundance of fishes during the two years. Fish populations in reservoirs usually increase rapidly in numbers after filling and thereafter occur seasonally or from year to year (Quarcoopone *et al*., 2011). According to Bhukaswan (1980), the alteration of existing ecological and bio-physical processes after impoundment such as the slowing of the flow of the river both upstream and downstream and the invasion of aquatic weeds have underpinned the changes in the relative abundance of fish species. The marked differences in seasonal abundance could also possibly be due to habitat preferences associated with water levels in the river (Gordon *et al*., 2003).

Freshwater fishes are among the most diverse groups of vertebrates in the world, exhibiting extraordinary taxonomic breadth and geographic scope in their distribution (Leveque *et al*., 2008). The rich taxonomic and functional diversity of freshwater fishes stem largely from the fact that streams, rivers, lakes and wetlands are embedded in terrestrial landscapes which limit the dispersal of freshwater organisms by promoting habitat isolation (Berra, 2007).

The diversity of fish species in the Black Volta near the Bui dam is presently high. Patterns of diversity (species richness, diversity and evenness) were investigated in relation to four hydrological seasons. In 2011, all three indices (richness, evenness and diversity) increased from the dry season to pre-wet season and dropped during the wet season and increased again in the post-wet season. During 2012 however, only species richness increased from the dry season to the pre-wet season and decreased from then till the post-wet season. Species evenness and diversity increased from the dry season through to the wet season and only dropped in the post-wet season. Spatio-temporal variations in the environmental characteristics as well as resource availability are among the main determinants of the species distribution (Grenouillet *et al*., 2002), species interaction (Zaret and Rand, 1971; Deus and Petrere Júnior, 2003) and habitat adaptations (Poff and Allan, 1995; Mérigoux *et al*., 2001).

A total of thirteen fish species of experimental gill net catches (out of the sixty-four recorded in the commercial gill net catches) belonged to nine genera and six families. This represented only 20.3 % of the commercial catches. Vanderpuye (1984) found high correlation between experimental and commercial gill net catches, and experimental catches proved to be a good indicator of trends in commercial catches. This scenario was not confirmed in the Black Volta because comparison of the species composition of the commercial and experimental gill net catches revealed large differences. This observation was further supported by the works of Ita and Mdaihli (1997) and Ikenweiwe (2005) on Kainji and Oyan lakes, respectively in Nigeria. Ita and Mdaihli (1997) and Ikenweiwe (2005) also found large differences between commercial and experimental catches and hence experimental gill net catches were not good indicators of trends in commercial gill net catches.

The data also revealed that size and species selectivity by the gill nets could not allow application of experimental catch to predict commercial catches. This could be attributed to the fact that other more efficient gears including drag nets, beach and purse seines, bamboo traps and other encircling gears could have been introduced (Braimah, 1989). There were also differences in the number of species that belonged to each family as reported by Ita and Mdaihli (1997). The fish catches in experimental gill nets decreased in number as the mesh size increased from 15 mm to 25 mm. This observation on the inverse relationship between fish catch and mesh size was similar to those of NIFFR (1991) on Asa Lake in Nigeria and Gordon *et al*. (2003) on the Black Volta in Ghana.

The Morpho-edaphic index was used to estimate the potential fish yield of the Bui reservoir in its formative years (June 2011 – December 2012). The potential fish yield of 16.13 $kgha^{-1}yr^{-1}$ was estimated for the newly formed Bui reservoir. Reservoirs in the tropics are noted to be more productive than those in the temperate regions (Jackson and Marmulla, 2001). The fish yield

(16.13 $kg^{-1}ha^{-1}yr^{-1}$) obtained for the Bui reservoir in the current research was lower than those estimated for some reservoirs in northern Ghana. For instance, Achubunyi (75.0 $kgha^{-1}yr^{-1}$) and Mahama (90.19 $kgha^{-1}yr^{-1}$) (Abban *et al.*, 1994); Bontanga (86.98 $kgha^{-1}yr^{-1}$) and Libga (97.19 $kgha^{-1}yr^{-1}$) (Quarcoopone, *et al.*, 2008). The potential fish yield of the Bui reservoir was however, higher than Lake Volta (15.55 $kgha^{-1}yr^{-1}$) in Ghana (Henderson and Welcomme, 1974) and Lake Kainja (4.7 $kgha^{-1}yr^{-1}$) in Nigeria (Balogun and Ibeun, 1995).

According to Marshall and Maes (1994), estimated fish yields in deep tropical reservoirs ranged from 10 – 50 $kgha^{-1}yr^{-1}$, while that of shallow tropical reservoirs ranged from 30 – 150 $kgha^{-1}yr^{-1}$. This implies the potential fish yield of the Bui reservoir is relatively low. This may be due to the fact that the study was undertaken during the formative years of the Bui reservoir. It is however, anticipated that there will be a quick turnover that will increase primary productivity of the reservoir as a result of breakdown of organic load from submerged vegetation as a result of the impoundment. This may increase the potential fish yield of the Bui reservoir in subsequent years as a result of increased primary productivity. On the basis of classification of reservoirs by potential fish yield, the Bui reservoir would be categorized as Mesohumic-mesotrophic (15 – 30 $kgha^{-1}yr^{-1}$) (after Braimah, 1995).

The monthly mean CPUE for 2011 decreased from 11.03 kg/canoe/day in March to 0.71 kg/canoe/day in August with a mean of 5.16 kg/canoe/day. These values were not yet uniform possibly because of the formation of the impoundment in June the same year. Hence, the reservoir was still in the interphase between the lacustrine and riverine conditions. The CPUE values for 2012 were however, uniform and decreased from 17.92 kg/canoe/day in February to

6.8 kg/canoe/day in March with a mean of 10.29 kg/canoe/day. This phenomenon could be due to the unusual low water level throughout the 2012 sampling year. Also, the unusual high CPUE from July to September (wet season) during 2012 was also due to the exceptionally low water level during the same period, since CPUE was inversely proportional to water level (Braimah, 1995).

The decline in CPUE immediately after the creation of the impoundment was apparently due to changes in trophic state, as well as to the construction of the dam upstream. The higher variability of CPUE that was observed before and after the impoundment was a reflection of fish community instability in these periods. This therefore corroborates other findings of this research that changes in community structure, density and diversity of flora and fauna were caused by river impoundment.

From the estimated CPUE and fishing effort, the total quantity of fish produced by commercial fishermen using gill nets was 26748 kg or 26.75 metric tonnes for 2011 and 58655 kg or 58.65 metric tonnes for 2012 and a mean of 42702 kg or 42.70 metric tonnes for the 2 year of sampling. The mean total catch of 42.70 metric tonnes forms only 0.03 % of the total inland capture fishery production which is about 150,000 metric tonnes (Anon, 2009) and 0.01 % of an estimate of 319,000 metric tonnes (comprising 251,000 metric tonnes from the Volta Lake and 68,000 metric tonnes from other sources) for the year 2000 (MOFA, 2006). Hence the contribution of the Black Volta gill net fisheries near the Bui dam area to total inland fishery production in Ghana is negligible. This therefore confirms the assumption that Lake Volta is the main ‘component’ of inland fisheries in Ghana (MOFA, 2006).

From the estimated maximum permissible number of canoes of approximately 35 in this study, the right-based licensing system for fisheries management should be implemented in order to control “open access” to the fisheries resources. This should be done taking the precautionary approach to capture fisheries into consideration (FAO, 1996). At this level of fishing effort, the fisheries may be considered to be at high level of exploitation. This should serve as immediate need to adopt methods to limit access (e.g. not exceeding 35 canoes initially) to the fisheries through management strategies to avoid early exploitation due to possible migration of more fishers (canoes) into the newly created Bui reservoir (Ofori-Danson *et al*., 2012). For the purpose of management, all other management measures indicated in the Fisheries Law of Ghana (Act, 625 of 2002) for inland water bodies should apply.

The study revealed a direct relationship between primary productivity as measured by chlorophyll *a* content and fish production as measured by CPUE. Hence, when primary productivity was high, CPUE was also high and vice versa. This compares favourably with earlier findings of Oglesby (1977), Jones and Hoyer (1982) and Quiros (1990) who suggested that CPUE was strongly correlated to chlorophyll *a* concentration.

There was however, an inverse relationship between flood regime as measured by water level and fish production as measured by CPUE. Thus, when water level was high, CPUE was low and vice versa. This corroborates the findings of Braimah (1995). Using lake level fluctuations and monthly commercial fish catches recorded in the Yeji portion of the lake (Stratum VII) from July 1989 to December 1991, Braimah (1995) showed an inverse relationship between fish catch and lake level: fish catches were high when lake levels were low and vice versa. Abobi *et al*. (2013) also showed an inverse relationship between water level and CPUE in the lower reaches of the White Volta near Yapei. Increasing CPUE with decreasing water level was observed in all the landing sites during the post flood season (October to December 2011). The dry season (January to March 2012), on the other hand had fluctuating CPUE with the decreasing water levels (Abobi *et al*., 2013). According to de Graaf and Ofori-Danson (1997), CPUE in Lake Volta was however, directly related to the water level. This perhaps still calls for an urgent and comprehensive assessment of Lake Volta fisheries through re-analysis of the fragmented but long-term data sets that have been collected in the past (Béné, 2007).

In the Bontanga Reservoir in the Northern Region of Ghana, increasing CPUE was also related to decreasing water levels between September 1995 and January 1996 as shown by CPUE values which rose from 12.35 kg to 16.73 kg (Quarcoopone *et al*., 2008). The relatively low CPUE associated with high water levels in the present study were also comparable to earlier works by Amarasinghe and Pitcher (1986), Amarasinghe (1987) and Blay Jr. and Asabere-Ameyaw (1993). These were attributed to the less success of reproduction and the generally poor representation of semi-pelagic fishes in the water bodies.

A multi-linear regression analysis of both water level and chlorophyll *a* content on the CPUE indicated that both partial coefficients were significant ($p < 0.05$). Hence, the best predictors of CPUE in the Black Volta were chlorophyll *a* and water level. Testing the present model using independent data set (obtained during 2011 and 2012 sampling years), the coefficient of determination, R^2 of 0.908 between the predicted values (model) and independent data demonstrated the substantial predictive power of the model.

This type of model may therefore be useful for estimating CPUE. The assumptions to be fulfilled in using this model, however, are that: a sampling unit was considered as a fisher utilizing a canoe normally with gill net; 50 % of local canoes were active; the contribution of other fishing gears to total fish catch is negligible; the water level was taken from the downstream station; and that the threshold for model input parameters were, water level = 0.5 m and chlorophyll *a* = 1.5 mgl^{-1}.

Simple regression models can be valuable and practical tools for understanding the dynamics of ecosystems if these models meet certain requirements such as simplicity, general applicability and validity (Rocha *et al*., 2009). It is therefore believed that this present model met these requirements and could be used to study other tropical freshwater bodies. The model here is also a valuable tool to predict future trends in the CPUE levels of the Black Volta.

CHAPTER SIX

6.0 CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

The general physico-chemical parameters monitored fell within the ranges suitable for fisheries and aquatic life in freshwater bodies with the exception of dissolved oxygen, nitrates, sulphates and phosphates which were outside recommended ranges.

The physico-chemical characteristics showed wide seasonal variations, while between sampling stations the differences were insignificant. The physico-chemical conditions were similar in both periods (pre- and post-impoundment) except for conductivity, total dissolved solids, colour and dissolved oxygen which differed significantly between these periods, hence reflecting the alterations in the river continuum due to impoundment.

The CCA ordination indicated that zooplankton organisms responded to a number of physico-chemical variables and 41 - 64 % of the variations in zooplankton densities were accounted for by the measured physico-chemical factors, while 8 - 12 % of the variations in phytoplankton densities were accounted for by the measured physico-chemical factors. This indicates that zooplankton communities responded to changes in the measured physico-chemical factors and this was seen in composition of species assemblages and abundance in the four hydrological seasons.

The total number of planktonic taxa and their overall density were significantly higher in the upstream than in the downstream station. River impoundment therefore, altered not only the physico-chemical characteristics but also the floral and faunal characteristics of the downstream station.

The following classes of phytoplankton (Bacillariophyceae, Chlorophyceae, Euglenophyceae) and total phytoplankton differed significantly between the pre- and post-impoundment periods. This therefore indicates that changes in their community structure and densities were caused by river impoundment.

Since the zooplankton were more abundant during the wet season, factors such as climatic changes and/or dam construction that will modify the flooding pattern of the river will inadvertently alter the zooplankton community structure in the newly created Bui reservoir and may have serious implications for fish production of the entire Black Volta ecosystem.

The change from riverine to lacustrine conditions during the formation of the reservoir, led to the immediate reduction in the numbers of a variety of fish families, including Centropomidae, Clarotidae and Distichodontidae which are very sensitive to oxygen depletion. The basic trend in this study was towards the development of a herbivorous fish community which included *Sarotherodon galilaeus*, *Labeo coubie* and *Labeo senegalensis*.

The mean estimated seasonal CPUE of fish by local canoes utilizing gill nets for the whole sampling period (2011 and 2012) was lowest (6.23 kg/canoe/day) in the post-wet season and highest (10.86 kg/canoe/day) in the dry season and thus making the dry season the most important season for fish catches in the Bui dam area. The CPUE decreased from 6.39 kg/canoe/day during pre-impoundment period to 3.91 kg/canoe/day in the immediate post-impoundment period and increased to a peak of 10.29 kg/canoe/day in the late post-impoundment. The CPUE also differed significantly between the pre- and post-impoundment periods and thus indicating the impact of river impoundment on fish production in the study area.

A multi-linear regression analysis of both water level and chlorophyll *a* concentration on the CPUE indicated that both partial coefficients were significant and validation with independent data showed that the model: **CPUE = - (0.456 x water level) + (0.062 x chlorophyll *a*) + 3.363**

had the potential to predict fish production as measured by CPUE levels in the Bui dam area of the Black Volta. The application of this model however, rests on the following assumptions:

i. a sampling unit was considered as a fisher utilizing a canoe normally with gill net;

ii. fifty percent (50 %) of local canoes were active;

iii. the contribution of other fishing gears to total fish catch were negligible;

iv. the water level was taken from the downstream station; and

v. that the threshold for model input parameters were, water level = 0.5 m and chlorophyll *a* = 1.5 mgl^{-1}.

The potential fish yield of the newly created Bui reservoir was 16.13 kgha^{-1}yr^{-1}. This was found to be less than most reservoirs in Africa. On the basis of classification of reservoirs by potential fish yield, the Bui reservoir would be categorized as Mesohumic-mesotrophic (i.e. 15 – 30 kgha^{-1}yr^{-1}) in its formative years.

6.2 Recommendations

Based on the stated findings, the following recommendations were made.

i. The following nutrients; nitrates, sulphates and phosphates were outside recommended ranges suitable for fisheries and aquatic life in freshwaters. The reasons for these high nutrient levels was still not clear in this study and therefore requires further investigations by limnologists. In the interim however, agricultural activities, especially the use of inorganic fertilizers within the catchment area, should be monitored and controlled by the Bole District Assembly, the Bamboi and Bui Traditional Councils.

ii. Changes in the measured environmental variables had significant effects on the structure of zooplankton assemblages. This feature could be used for bio-monitoring of the river health by the Bui Power Authority (BPA) to ensure protection of the aquatic biota.

iii. From the estimated maximum permissible number of canoes of approximately 35, the right-based licensing system for fisheries management should be implemented by the Fisheries Commission and BPA so that fishermen could cooperate in the management options of the newly created Bui reservoir in order to prevent over-exploitation of the fisheries resources by fishermen.

iv. Predictor variables that significantly explained CPUE were chlorophyll *a* (direct relationship) and water level (inverse relationship). This model explained about 91 % of CPUE variability. Hence predictive models are valuable and practical tools for understanding the dynamics of fish populations and that predictive hydro-biology should be considered an approach by research scientists and graduate students in monitoring the CPUE of the Black Volta in the Bui dam area.

v. Environmental flow requirements (including management of flood releases) should be used by BPA to reduce the impacts of changed river flow regimes on the aquatic ecosystems downstream.

vi. Mitigation measures such as good information base; early co-operation between ecologists, the dam design team and affected people; and regular monitoring and feedback on the effectiveness of mitigation measures should be ensured by the BPA, the Bui Traditional Council and the local fishers.

vii. Fishermen and farmers whose livelihoods were affected as a result of the dam construction should be assisted by the BPA and local government authorities in the catchment area with alternative livelihoods (e.g. fish farming and livestock rearing).

viii. The illegal mining operations (‘Galamsey’) observed downstream during the studies should by controlled by the Bamboi Traditional Council and local government authorities in the area to prevent pollution and siltation of the water body and destruction of fish habitats.

ix. Similar studies should be conducted by research scientists and graduate students five years after impoundment to ascertain which species of plankton and fish would be established by which time the reservoir would have been more stabilized. This is because the reservoir had not yet stabilized during the study period.

x. Since the most abundant zooplankton species identified in the study area was *Cyclops sp*, a vector of Guinea worm which can contribute to an outbreak of the Guinea worm disease among communities along the Black Volta basin, studies on the nematode (*Dracunculiasis medinensis*) which infests *Cyclops sp* should be conducted by Scientists from the School of Public Health and other Health Research Centers in the country in order to help in the prevention and eradication of the disease in the area. This is because, the Guinea worm disease is mainly endemic in 13 countries (all in Sub-Saharan Africa), except for a few remote villages in the Rajastan desert of India and in Yemen.

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APPENDICES

Appendix I: Mean monthly physico-chemical parameters during the study period

Months	Conductivity (μS/cm)	TDS (mg/l)	pH	Temp (°C)	Colour (Hz)	DO (mg/l)	Nitrates (mg/l)	Phosphates (mg/l)	Sulphates (mg/l)
March, 2011	140.6	84.8	7.13	29.3	30	2.7	0.26	0.058	3.89
April	142.0	85.6	7.77	28.9	45	2.49	0.69	0.005	1.81
May	143.8	86.0	7.58	29.6	75	2.5	2.27	0.001	9.31
June	104.1	62.7	7.48	25.1	375	2.4	4.0	0.05	31.9
July	94.8	56.8	7.51	29.9	250	2.9	3.08	0.031	34.94
August	90.5	54.1	6.98	24.6	312.5	2.5	3.07	0.001	39.81
September	69.2	41.5	7.01	27.6	312.5	2.8	2.72	0.001	34.1
October	79.2	47.5	6.93	29.9	215	2.0	2.6	0.001	21.7
November	73.0	37.0	8.3	29.5	175	1.5	1.35	0.63	22.5
December	73.0	37.0	8.1	29.5	185	1.8	1.55	0.23	21.0
January, 2012	99.5	59.7	7.73	29.0	62.5	2.2	0.74	0.001	1.6
February	103.1	61.9	7.41	30.5	10.0	2.2	1.26	0.001	5.69
March	109.6	65.9	8.05	31.1	15.0	5.5	1.35	0.001	5.813
April	106.5	64.1	7.92	33.0	15.0	5.0	1.81	0.001	6.81
May	89.9	54.9	7.45	26.7	20.0	4.7	3.85	0.001	16.14
June	99.5	59.7	7.73	29.0	62.5	2.2	0.74	0.001	16.06

Appendix I (continued)

July	149.5	90.2	7.43	26.9	262.5	5.9	2.23	0.12	26.4
August	121.6	73.2	7.07	27.3	70.5	6.1	2.02	0.11	11.7
September	96.5	66.5	7.36	27.4	85.3	6.0	2.51	0.11	25.3
October	86.2	70.1	7.24	28.6	69.3	5.9	2.3	0.09	20.3
November	81.3	63.9	7.73	28.7	64.4	5.8	1.2	0.15	16.2
December	79.4	62.8	7.7	29.1	68.9	5.9	1.1	0.09	12.4
Mean	**101.49**	**62.99**	**7.53**	**28.69**	**126.4**	**3.68**	**1.94**	**0.08**	**17.52**

Appendix II: Seasonal changes in the quantitative distribution of phytoplankton in 2011

TAXA	**Dry season (no/m^3)**	**Pre-wet season (no/m^3)**	**Wet season (no/m^3)**	**Post-wet season (no/m^3)**	**Mean (no/m^3)**
BACILLARIOPHYCEAE					
Gyrosigma sp.	0	216	0	10	57
Navicula sp.	207	297	49	27	145
Surirella sp.	0	40	0	0	10
Synedra ulna	44	520	24	86	169
Sub total	251	1073	73	123	380
CHLOROPHYCEAE					
Ankistrodesmus sp.	63	873	208	404	387
Carteria sp.	0	80	0	0	20
Chlamydomonas sp.	501	80	0	0	145
Chlorella sp.	8	200	0	15	56
Chlorogonium sp.	0	40	0	0	10
Closterium sp.	56	640	0	0	174
Coelastrum sp.	54	0	234	0	72
Cosmarium sp.	0	40	0	0	10

Appendix II (continued)

Dictyosphaerium sp.	0	400	0	0	100
Micrasterias sp.	0	40	0	0	10
Pediastrum sp.	109	840	0	0	237
Scenedesmus sp.	65	80	16	0	40
Schroederia sp.	0	60	0	0	15
Staurastrum sp.	14	40	0	0	14
Stigeoclonium sp.	0	80	0	0	20
Ulothrix sp.	486	1272	581	701	760
Volvox sp.	0	0	277	0	69
Sub total	1356	4765	1316	1120	2139
CYANOPHYCEAE					
Anabaena sp.	73	1200	170	46	372
Chroococcus sp.	32	680	0	0	178
Coelosphaerium sp.	6	440	0	0	112
Lyngbya circumcreta	2	240	16	38	74
Merismopedia puntata	0	0	346	0	87
Microcystis aeruginosa	0	1640	242	261	536
Microcystis wesenbergii	0	0	595	686	320

Appendix II (continued)

Oscillatoria sp.	15	100	51	77	61
Planktothrix sp.	86	499	103	337	256
Pseudanabaena sp.	13	865	461	280	405
Rivularia sp.	0	58	0	0	15
Spirulina sp.	5	0	0	0	1
Sub total	232	5722	1984	1725	2416
EUGLENOPHYCEAE					
Euglena sp.	0	80	0	0	20
Phacus pyrum	0	80	0	0	20
Sub total	0	160	0	0	40
GRAND TOTAL	**1839**	**11720**	**3373**	**2968**	**4974**

Appendix III: Seasonal changes in the quantitative distribution of phytoplankton in 2012

TAXA	Dry season (no/m^3)	Pre-wet season (no/m^3)	Wet season (no/m^3)	Post-wet season (no/m^3)	Mean (no/m^3)
BACILLARIOPHYCEAE					
Gyrosigma sp.	0	0	0	0	0
Navicula sp.	13	35	5	7	15
Surirella sp.	0	0	0	0	0
Synedra ulna	5	47	12	9	18
Sub total	18	82	17	16	33
CHLOROPHYCEAE					
Ankistrodesmus sp.	71	67	23	18	45
Carteria sp.	0	0	0	0	0
Chlamydomonas sp.	0	0	0	12	3
Chlorella sp.	82	71	0	4	39
Chlorogonium sp.	0	0	0	0	0
Closterium sp.	0	0	0	0	0
Coelastrum sp.	0	0	0	0	0
Cosmarium sp.	0	0	0	0	0
Dictyosphaerium sp.	0	0	0	0	0

Appendix III (continued)

Micrasterias sp.	12	12	0	10	9
Pediastrum sp.	112	18	39	14	46
Scenedesmus sp.	97	175	24	19	79
Schroederia sp.	0	0	0	0	0
Staurastrum sp.	8	15	0	2	6
Stigeoclonium sp.	0	0	0	0	0
Ulothrix sp.	209	380	22	15	157
Volvox sp.	0	0	0	0	0
Sub total	591	738	108	94	383
CYANOPHYCEAE					
Anabaena sp.	89	215	20	16	85
Chroococcus sp.	0	0	0	0	0
Coelosphaerium sp.	0	0	0	0	0
Lyngbya circumcreta	16	73	15	16	30
Merismopedia puntata	341	192	86	75	174
Microcystis aeruginosa	426	390	84	86	247
Microcystis wesenbergii	551	414	105	89	290
Oscillatoria sp.	15	15	5	4	10

Appendix III (continued)

Planktothrix sp.	90	227	19	41	94
Pseudanabaena sp.	363	55	52	36	127
Rivularia sp.	0	0	0	0	0
Spirulina sp.	0	0	0	0	0
Sub total	1891	1581	386	363	1055
EUGLENOPHYCEAE					
Euglena sp.	0	0	0	0	0
Phacus pyrum	0	0	0	0	0
Sub total	0	0	0	0	0
GRAND TOTAL	**2500**	**2401**	**511**	**473**	**1471**

Appendix IV: Seasonal changes in the quantitative distribution of zooplankton in 2011

Taxa	**Dry season (no/m^3)**	**Pre-wet season (no/m^3)**	**Wet season (no/m^3)**	**Post-wet season (no/m^3)**	**Mean (no/m^3)**
CLADOCERA					
Alonella sp.	0	0	10	0	2.5
Bosmina sp.	0	3	0	0	0.8
Ceriodaphnia sp.	0	736	110	0	212
Daphnia sp.	7	500	0	0	127
Diaphanosoma sp.	9	338	0	0	87
Leptodora sp.	1	0	3,460	0	865
Macrothrix sp.	0	2	0	0	0.5
Moina sp.	19	0	0	0	5
Polyphemus sp.	0	474	2,660	0	783
Sida sp.	0	0	16	0	4
Sub total	36	2,053	6,256	0	2,086
COPEPODA					
Canthocamptus sp.	0	8	0	0	2
Cyclops sp.	68	960	3,130	73	1,058
Cypridopsis sp.	0	480	0	1	120

Appendix IV (continued)

Diaptomus sp.	23	0	16	0	10
Eubranchanpus sp.	0	12	0	0	3
Limnocalanus sp.	0	740	0	0	185
Sub total	91	2,200	3,146	74	1,378
GRAND TOTAL	**127**	**4,253**	**9,402**	**74**	**3,464**

Appendix V: Seasonal changes in the quantitative distribution of zooplankton in 2012

Taxa	**Dry season (no/m^3)**	**Pre-wet season (no/m^3)**	**Wet season (no/m^3)**	**Post-wet season (no/m^3)**	**Mean (no/m^3)**
CLADOCERA					
Alonella sp.	0	10	0	2	3
Bosmina sp.	0	14	0	0	4
Ceriodaphnia sp.	0	36	29	0	16
Daphnia sp.	22	98	21	11	38
Diaphanosoma sp.	36	60	34	0	33
Leptodora sp.	24	22	224	20	73
Macrothrix sp.	4	6	0	0	3
Moina sp.	84	26	6	0	29
Polyphemus sp.	12	46	117	0	44
Sida sp.	0	12	5	0	4
Sub total	182	330	436	33	247
COPEPODA					
Canthocamptus sp.	0	6	4	2	3

Cyclops sp.	165	98	334	36	158
Cypridopsis sp.	0	8	54	0	15
Diaptomus sp.	69	0	3	0	18
Eubranchanpus sp.	0	14	6	0	5
Limnocalanus sp.	15	142	71	10	60
Sub total	249	268	472	48	259
GRAND TOTAL	**431**	**598**	**908**	**81**	**506**

Appendix VI: Fish species abundance by number and weight (kg) encountered in gill net catches in 2011 and 2012

FAMILY/SPECIES			**2011**				**2012**	
	No.	**Wt (kg)**	**% No.**	**% Wt**	**No.**	**Wt (kg)**	**% No.**	**% Wt**
ALESTIDAE								
Alestes baremoze	32	4.1	2.21	1.58	21	5.82	1.66	1.25
Alestes dentex	1	0.04	0.07	0.02	36	21.61	2.84	4.65
Brycinus leuciscus	7	0.8	0.48	0.31	22	1.08	1.73	0.23
Brycinus macrolepidotus	44	6.52	3.04	2.51	45	20.53	3.55	4.42
Brycinus nurse	41	0.88	2.84	0.34	96	14.8	7.57	3.19
Hydrocynus brevis	5	2.4	0.35	0.92	-	-	-	-
Hydrocynus forskalii	33	11.24	2.28	4.33	7	2.5	0.55	0.54
Sub total	**163**	**25.98**	**11.28**	**10.01**	**227**	**66.34**	**17.9**	**14.28**
ANABANTIDAE								
Ctenopoma petherici	**1**	**0.2**	**0.07**	**0.08**	-	-	-	-

BAGRIDAE								
Bagrus bajad	21	9.82	1.45	3.78	13	10	1.02	2.15
Bagrus docmak	8	18.48	0.55	7.12	7	3.96	0.06	0.85
Sub total	**29**	**28.3**	**2**	**10.9**	**20**	**13.96**	**1.58**	**3.01**
CICHLIDAE								
Chromidotilapia guntherii	5	0.54	0.35	0.21	2	0.48	0.16	0.1
Hemichromis fasciatus	3	0.88	0.21	0.34	-	-	-	-
Oreochromis niloticus	7	1.15	0.48	0.44	-	-	-	-
Sarotherodon galilaeus	16	2.48	1.1	0.95	142	24.5	11.2	5.27
Steatocranus irvinea	4	0.08	0.28	0.03	2	0.7	0.16	0.15
Tilapia dageti	1	0.8	0.07	0.31	-	-	-	-
Tilapia zillii	5	0.88	0.35	0.34	-	-	-	-
Sub total	**41**	**6.81**	**2.84**	**2.62**	**146**	**25.68**	**11.52**	**5.53**
CENTROPOMIDAE								
Lates niloticus	**26**	**7.64**	**1.8**	**2.94**	**12**	**4.38**	**0.95**	**0.94**

CITHARINIDAE								
Citharinus citharus	**6**	**1.12**	**0.42**	**0.43**	-	-	-	-
CLARIIDAE								
Clarias anguillaris	1	2.18	0.07	0.84	9	5.99	0.71	1.29
Clarias gariepinus	51	10.03	3.53	3.86	9	18.23	0.71	3.92
Heterobranchus bidorsalis	93	12.13	6.44	0.05	-	-	-	-
Heterobranchus isopterus	15	8.64	1.03	0.03	-	-	-	-
Sub total	**160**	**32.98**	**11.07**	**12.7**	**18**	**24.22**	**9.15**	**5.21**
CLAROTIDAE								
Auchenoglanis occidentalis	23	8.84	1.6	3.4	33	14.97	2.6	3.22
Chrysichthys auratus	64	10.64	4.43	4.1	29	9.55	2.29	2.06
Chrysichthys nigrodigitatus	19	4.7	1.31	1.81	54	17.81	4.26	3.83
Sub total	**106**	**24.18**	**7.34**	**9.31**	**116**	**42.33**	**9.15**	**9.11**
CLUPEIDAE								
Odaxothrissa mento	**4**	**0.4**	**0.28**	**0.15**	-	-	-	-

CYPRINIDAE								
Labeo coubie	21	11.6	1.45	4.47	82	51.07	6.47	10.99
Labeo parvus	28	3.06	1.94	1.18	14	11.04	1.1	2.38
Labeo senegalensis	36	7.33	2.49	2.82	230	94.96	18.14	20.44
Raiamas senegalensis	1	0.02	0.07	0.01	-	-	-	-
Sub total	**86**	**22.01**	**5.95**	**8.47**	**326**	**157.07**	**25.71**	**33.81**
DISTICHODONTIDAE								
Distichodus brevipinnis	24	11.28	1.66	4.34	-	-	-	-
Distichodus engycephalus	14	2.9	0.97	1.12	13	3.06	1.02	0.66
Distichodus rostratus	1	0.02	0.07	0.01	9	5.9	0.71	1.27
Sub total	**39**	**14.2**	**2.7**	**5.47**	**22**	**8.96**	**1.73**	**1.93**
HEPSETIDAE								
Hepsetus odoe	**3**	**0.84**	**0.21**	**0.32**	**4**	**1.43**	**0.32**	**0.31**
MALAPTERURIDAE								
Malapterurus electricus	**14**	**8.9**	**0.97**	**3.43**	**25**	**22.85**	**1.97**	**4.92**

MASTACEMBELIDAE								
Aethiomastacembelus nigromarginatus	**1**	**0.08**	**0.07**	**0.03**	-	-	-	-
MOCHOKIDAE								
Hemisynodontis membranaceus	27	4.22	1.87	1.62	54	16.34	4.26	3.52
Synodontis clarias	11	2.58	0.76	0.99	20	6.4	1.58	1.38
Synodontis eupterus	4	0.14	0.28	0.05	-	-	-	-
Synodontis filamentosus	7	0.26	0.48	0.1	2	1	0.16	0.22
Synodontis nigrita	86	2.12	5.95	0.82	25	4.24	1.97	0.91
Synodontis ocellifer	45	5.12	3.11	1.97	21	7.19	1.66	1.55
Synodontis schall	87	2.52	6.02	0.97	8	1.88	0.63	0.4
Synodontis sorex	180	15.15	12.46	5.83	29	7.64	2.29	1.64
Synodontis velifer	39	0.62	2.7	0.24	22	4.02	1.73	0.86
Sub total	**486**	**32.73**	**33.63**	**12.6**	**181**	**48.71**	**14.27**	**10.48**
MORMYRIDAE								
Campylomormyrus tamandua	1	0.04	0.07	0.02	-	-	-	-
Hyppopotamyrus pictus	14	2.71	0.97	1.04	6	1.56	0.47	0.34

Hyperopesus bebe	21	8.46	1.45	3.26	2	1.35	0.16	0.29
Marcusenius abadii	-	-	-	-	11	2.46	0.87	0.53
Marcusenius senegalensis	19	0.56	1.31	0.22	33	4.75	2.6	1.02
Mormyrops anguilloides	20	8.1	1.38	3.12	5	4.7	0.39	1.01
Mormyrops breviceps	5	5.44	0.35	2.09	12	6.8	0.95	1.46
Mormyrus macrophthalmus	36	3.46	2.5	1.33	21	8.32	1.66	1.79
Mormyrus rume	19	7.74	1.31	2.98	31	10.98	2.44	2.36
Petrocephalus bovei	1	0.32	0.07	0.12	-	-	-	-
Sub total	**136**	**36.83**	**9.41**	**14.18**	**121**	**40.92**	**9.54**	**8.81**
OSTEOGLOSSIDAE								
Heterotis niloticus	**18**	**10.9**	**1.25**	**4.2**	-	-	-	-
POLYPTERIDAE								
Polypterus birchir	1	0.5	0.07	0.19	-	-	-	-
Polypterus endlicheri	4	0.74	0.28	0.28	-	-	-	-
Polypterus senegalus	1	0.33	0.07	0.13	-	-	-	-
Sub total	**6**	**1.57**	**0.42**	**0.6**	-	-	-	-

SCHILBIDAE								
Schilbe intermedius	8	0.69	0.56	0.27	-	-	-	-
Schilbe mystus	108	2.42	7.47	0.93	40	5.26	3.15	1.13
Siluranodon auritus	1	0.02	0.07	0.01	10	2.44	0.79	0.53
Sub total	**117**	**3.13**	**8.1**	**1.21**	**50**	**7.7**	**3.94**	**1.66**
TETRAODONTIDAE								
Tetraodon lineatus	**3**	**0.92**	**0.21**	**0.35**	-	-	-	-
GRAND TOTAL	**1445**	**259.72**	**100**	**100**	**1268**	**464.55**	**100**	**100**

- indicates absent

Appendix VII: Summary of Fish catches per mesh size of experimental gill nets

FAMILY/SPECIES	15 mm		17.5 mm		20 mm		22.5 mm		25 mm		30 mm		40 mm		Total	
	No	Wt (kg)	No	Wt (kg)	No	Wt (kg)	No	Wt (kg)	No	Wt (kg)	No	Wt (kg)	No	Wt (kg)	No	Wt (kg)
ALESTIDAE																
Alestes dentex	9	1.22	9	2.52	2	0.58	3	1.04	1	0.58	-	-	-	-	24	5.94
Brycinus nurse	15	0.8	10	0.94	-	-	2	0.92	3	2.4	-	-	-	-	30	5.06
Hydrocynus forskalii	5	1.2	1	0.6	1	0.7	-	-	-	-	-	-	-	-	7	2.5
CLARIIDAE																
Clarias gariepinus	-	-	4	2.1	1	0.64	-	-	2	3.9	-	-	-	-	7	6.64
CLAROTIDAE																
Chrysichthys auratus	5	0.68	1	0.62	8	0.68	4	0.6	1	1.45	-	-	-	-	19	4.03
Chrysichthys nigrodigitatus	7	0.72	6	1.9	1	0.66	1	0.76	-	-	-	-	-	-	15	4.04
MOCHOKIDAE																
Synodontis nigrita	-	-	-	-	3	0.34	8	0.7	11	2.7	-	-	-	-	22	3.74
Synodontis sorex	-	-	-	-	3	0.42	3	0.66	1	0.56	-	-	-	-	7	1.64
Synodontis velifer	-	-	4	0.31	3	0.82	2	0.6	1	0.52	-	-	-	-	10	2.25

MORMYRIDAE																	
Mormyrops anguiloides	-	-	10	4.3	-	-	1	0.64	1	1.15	-	-	-	-	12	6.09
Petrocephalus bovei	2	0.3	-	-	4	0.72	-	-	-	-	-	-	-	-	6	1.02
SCHILBIDAE																
Schilbe intermedius	11	1.64	-	-	2	0.56	1	0.3	1	0.52	-	-	-	-	15	3.02
Schilbe mystus	8	0.66	-	-	1	0.2	-	-	1	0.54	-	-	-	-	10	1.4
TOTAL	**62**	**7.22**	**45**	**13.29**	**29**	**6.32**	**25**	**6.22**	**23**	**14.32**	**-**	**-**	**-**	**-**	**184**	**47.37**

- indicates absent

Appendix VIII: Phytoplankton species diversity during 2011 and 2012

Diversity indices	**Dry season**		**Pre-wet season**		**Wet season**		**Post-wet season**	
	2011	**2012**	**2011**	**2012**	**2011**	**2012**	**2011**	**2012**
Species (S)	19	17	30	17	15	14	13	18
Number (N)	1839	2500	11720	2401	3373	511	2968	473
Richness (d)	2.3946	2.045	3.0953	2.0556	1.7234	2.0845	1.5008	2.7601
Evenness (J')	0.7374	0.7969	0.8439	0.8414	0.8618	0.8718	0.7956	0.8431
Diversity, H'(log10)	0.9429	0.9806	1.2465	1.0353	1.0135	0.9992	0.8863	1.0583

Appendix IX: Zooplankton species diversity during 2011 and 2012

Diversity indices	**Dry season**		**Pre-wet season**		**Wet season**		**Post-wet season**	
	2011	**2012**	**2011**	**2012**	**2011**	**2012**	**2011**	**2012**
Species (S)	6	9	11	15	7	13	2	6
Number (N)	127	431	4253	598	9402	908	74	81
Richness (d)	1.0322	1.3188	1.1968	2.1897	0.65583	1.7618	0.23234	1.7378
Evenness (J')	0.73314	0.8008	0.80262	0.8353	0.60243	0.7026	0.10328	0.7914
Diversity, H'(log10)	0.57046	0.7642	0.83584	0.9823	0.67368	0.7827	2.70E-02	0.6158

Appendix X: Macrobenthic invertebrate diversity during 2011 and 2012

Diversity indices	**Dry season**		**Pre-wet season**		**Wet season**		**Post-wet season**	
	2011	**2012**	**2011**	**2012**	**2011**	**2012**	**2011**	**2012**
Species (S)	4	4	4	3	3	4	4	4
Number (N)	37	45	27	32	25	24	39	48
Richness (d)	0.8308	0.7881	0.9102	0.5771	0.6213	0.9439	0.8189	0.7749
Evenness (J')	0.7788	0.803	0.788	0.9755	0.6804	0.8129	0.7817	0.7224
Diversity, H'(log10)	0.4689	0.4835	0.4744	0.4655	0.3246	0.4894	0.4706	0.4349

Appendix XI: Fish species diversity during 2011 and 2012

Diversity indices	**Dry season**		**Pre-wet season**		**Wet season**		**Post-wet season**	
	2011	**2012**	**2011**	**2012**	**2011**	**2012**	**2011**	**2012**
Species (S)	24	35	32	35	29	35	41	24
Number (N)	250	395	136	283	339	390	658	202
Richness (d)	4.166	5.6867	6.31	6.0226	4.806	5.6988	6.164	4.3329
Evenness (J')	0.7692	0.7758	0.8761	0.8413	0.6636	0.8973	0.8525	0.7685
Diversity, H'(log10)	1.062	1.1979	1.319	1.299	0.9705	1.3855	1.375	1.0607

Appendix XII: Estimated mean CPUE of canoes utilizing gill nets in 2011

Months	**Feb**	**Mar**	**Apr**	**May**	**June**	**July**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Mean**
Total fishing days (x)	24	27	26	27	26	27	27	26	27	26	27	26.4
Mean CPUE (Kg/canoe/day (y))	10.15	11.03	3.71	4.44	5.11	7.62	0.71	1.87	1.18	4.74	6.18	5.16
Total expected estimated canoes (or total effort)	35	35	35	35	35	35	35	35	35	35	35	35
50 % Canoes assumed active (z)	18	18	18	18	18	18	18	18	18	18	18	18
Total catch (kg) (xyz)	4385	5361	1736	2158	2391	3703	345	875	573	2218	3003	2452

Appendix XIII: Estimated mean CPUE of canoes utilizing gill nets in 2012

Months	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**June**	**July**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Mean**
Total fishing days (x)	27	25	27	26	27	26	27	27	26	27	26	27	26.5
Mean CPUE (Kg/canoe/day (y))	8.67	17.92	6.8	7.25	7.53	12.47	13.41	13.98	10.16	7.56	8.76	8.94	10.29
Total expected estimated canoes (or total effort)	35	35	35	35	35	35	35	35	35	35	35	35	35
50 % Canoes assumed active (z)	18	18	18	18	18	18	18	18	18	18	18	18	18
Total catch (kg) (xyz)	4214	8064	3305	3393	3659	5836	6517	6794	4755	3674	4099	4345	4908

Appendix XIV: CPUE, flood regime and primary productivity during the study period

Months	**CPUE (kg/canoe/day)**	**Water level (m)**	**Chlorophyll *a* (mgl^{-1})**
March, 2011	11.03	0.86	126.1
April	3.71	0.89	37.7
May	4.44	1.19	25.42
June	5.11	2.09	5.12
July	7.62	3.1	77.8
August	0.71	5.2	9.73
September	1.87	8.9	45.6
October	1.18	8.5	24.53
November	4.74	1.9	62.5
December	6.18	1.5	69.31
January, 2012	8.67	0.9	82.5
February	17.92	0.85	218.9
March	6.8	0.91	75.2
April	7.25	1.0	41.41
May	7.53	1.0	67.48
June	12.47	3.1	174.4
July	13.41	1.99	125.6
August	13.98	0.84	182
September	10.16	1.2	132.4
October	7.56	2.5	89.4
November	8.76	1.2	124.6
December	8.94	0.9	124.9
Mean	**7.73**	**2.29**	**87.39**

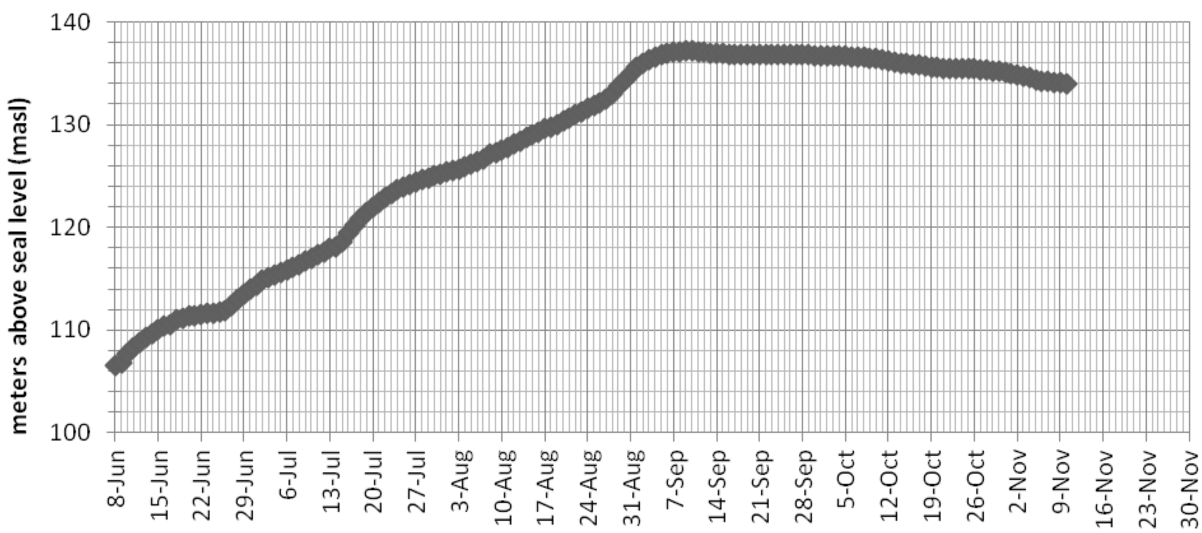
Appendix XV: Observed CPUE and estimated CPUE from the predictive model

Months	Observed values	Predicted values	Residuals	Standard error of predicted values
March, 2011	11.03	10.78821	0.24179	0.363530
April	3.71	5.29434	-1.58434	0.482581
May	4.44	4.39629	0.04371	0.509757
June	5.11	2.72752	2.38248	0.545227
July	7.62	6.77272	0.84728	0.300302
August	0.71	1.59544	-0.88544	0.540681
September	1.87	2.13229	-0.26229	0.891427
October	1.18	1.00846	0.17154	0.836409
November	4.74	6.37131	-1.63131	0.327541
December	6.18	6.97584	-0.79584	0.331580
January, 2012	8.67	8.06707	0.60293	0.347961
February	17.92	16.54573	1.37427	0.733345
March	6.8	7.60996	-0.80996	0.360828
April	7.25	5.47419	1.77582	0.458784
May	7.53	7.09035	0.43965	0.372004
June	12.47	12.76125	-0.29125	0.603171
July	13.41	10.24204	3.16796	0.345399
August	13.98	14.26274	-0.28274	0.559956
September	10.16	11.02376	-0.86376	0.368353
October	7.56	7.76538	-0.20538	0.284268
November	8.76	10.54021	-1.78021	0.347886
December	8.94	10.69558	-1.75558	0.359291
Minimum	0.71	1.00846	-1.78021	0.282253
Maximum	17.92	16.54573	3.16796	0.891427
Mean	**7.73**	**7.73367**	**0.00000**	**0.458806**

Appendix XVI: Results of multi-linear regression analysis of CPUE against chlorophyll *a* and water level

					95% Confidence Interval for B	
Predictor variables	**B**	**std error**	**t**	**sig.**	**Lower Bound**	**Upper Bound**
Constant	3.363	0.748	4.487	0.000	1.792	4.925
Water level	-0.456	0.141	-3.238	0.004	-0.751	-0.161
Chlorophyll *a*	0.062	0.006	10.928	0.000	0.05	0.074

Dependent Variable: CPUE, R^2 = 0.906; p = 0.0002; and the standard error of the estimate = 1.354



Appendix XVII: Post-impoundment trend in the water level of the Bui reservoir from June 2011 (after Ofori-Danson *et al*., 2012)

PLATES (continued)

Plate O: Pre-impoundment nature of the Black Volta near Bui (picture taken, February 2011)

Plate P: The post-impoundment nature of the Black Volta near Bui (picture taken, October 2011)

Plate Q: A view of the early spill-over at the Bui dam (Picture taken, November 2011)

Plate R: A view of the late construction works at the Bui dam (Picture taken, November 2012)

Plate S: Fish mongers waiting for catch at the Bui reservoir (Picture taken, October 2011)

Plate T: Fish mongers rushing for catch at the Bui reservoir (Picture taken, October 2011)

Plate U: Fishermen on the Black Volta near Bamboi (picture taken, October 2011)

Plate V: Non-motorised canoes at the newly created Bui reservoir (picture taken, October 2011)