MERCURY LEVELS IN FISH, SEDIMENT AND WATER FROM THE BLACK VOLTA RIVER BEFORE THE CONSTRUCTION OF THE HYDRO ELECTRIC DAM-BASELINE STUDIES AT BUI DAM SITE

BY

SOLOMON KWAKU DANSO-ANKAMAH

(BSc Agriculture Technology)

A THESIS SUBMITTED TO THE DEPARTMENT OF MATERIALS ENGINEERING

KWAME NKRUMAH UNIVERSITY OF SCIENCE AND

TECHNOLOGY, KUMASI

IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCIENCE

(ENVIRONMENTAL RESOURCES MANAGEMENT)

FACULTY OF CHEMICALS AND MATERIALS ENGINEERING

COLLEGE OF ENGINEERING

MARCH 2009

DECLARATION

It is hereby declared that this thesis is my own work towards the MSc and that, to the best of my knowledge, it contains no material previously published by another person nor material which has been accepted for the award of any degree of the University, except where due acknowledgement has been made in the text.

Solomon Kwaku Danso-Ankamah (20047065) ..... Signature 11-12-09 ..... Date

Certified by:

MR. R. B. Voegborlo Supervisor ..... Signature 11-12-09 ..... Date

Certified by:

PROF. F. W. Y. Momade Supervisor ..... Signature 11/12/09 ..... Date

Certified by:

DR. A. A. Adjaottor Head of Department ..... Signature 15/12/09 ..... Date

DEDICATION

I dedicate this work to the Glory of God without whom man labours in vain.

ACKNOWLEDGEMENTS

My utmost gratitude goes to the Almighty God for His grace, mercy and divine guidance throughout this program. I am grateful to my parents for their care, support and love.

I acknowledge with profound gratitude my supervisors, Mr. R. B. Voegborlo and Prof. F. W. Y. Momade for their careful corrections, constructive criticisms and suggestions during the preparation of the manuscript. Special thanks to Mr. Eric Selorm Agorku and Mr. Daniel Adu-Ampratwum for their technical assistance.

I cannot deny myself the pleasure of acknowledging the Manager of Bui National Park, Mr. C. A. Fumey-Nasssh, a principal technical officer of Bui National Park, Mr. Dankwa, a fisherman at Bator, Mr. Maxwell Gbedago and Maame Mansa Dodovi, a fishmonger at Bui Dam site for their immense support during the sample collection.

Finally, I am also grateful to Mr. Dassah, a lecturer at the Fisheries Department, Institute of Renewable Natural Resources (IRNR), Kwame Nkrumah University of Science and Technology (KNUST) for the identification of the fish species.

ABSTRACT

The risk of elevated Hg concentration in fish has become one of the most important issues in assessing the environmental impact of hydroelectric reservoirs. The construction of such reservoirs has led to increased rates of converting Hg in the aquatic system into methylmercury (MeHg), the form easily accumulated by fish. People and wildlife that eat fish from hydroelectric reservoirs especially newly constructed have an elevated risk of accumulating too much MeHg.

Total mercury (Hg) concentrations were determined in fish, sediments and water from the Black Volta River Basin at Bator and Dam site all at Bui prior to the impoundment for the construction of the Bui Hydroelectric Dam. Cold Vapour Atomic Absorption Spectrophotometry (CVAAS) technique using an automatic mercury analyser was employed after digestion of the samples. One hundred and eighty-five (185) fish samples comprising twenty-one (21) species; one hundred and twenty (120) sediment samples and twelve (12) water samples were collected and analysed for total mercury.

Mercury concentration (ng/g wet weight) in the muscle tissue of fish from Bator ranged from 59.25 to 181.29 (mean =127.03±42.49) for *Lates niloticus*, from 57.49 to 208.19 (mean = 137.76±49.11) for *Schilbe mystus*, from 19.97 to 27.04 (mean = 22.90±3.68) for *Alestes dentex*, from 14.39 to 39.68 (mean=27.88±7.67) for *Brycinus nurse*, from 68.38 to 121.39 (mean= 94.82±21.02) for *Chrysichthys auratus*, from 69.78 to 99.57 (mean = 84.68±21.07) for *Mormyrus sp*, from 14.28 to 24.93 (mean = 18.58±4.28) for *Labeo coubie*, from 18.36 to 208.54 (mean = 88.01±58.53) for *Hydrocynus sp*, from 32.01 to

68.43 (mean= 52.34±15.32) for *Synodontis oceillifer*, from 42.41 to 86.55 (mean= 61.23±18.72) for *Synodontis sp*, from 45.5 to 62.04 (mean = 56.07±7.45) for *Mormyrus macrophthalmus*, from 104.86 to 105.89 (mean = 105.38±0.73) for *Irvinea voltae* and 28.55 for *Barbus sp*, 68.73 for *Marcusenius abadii*, and 94.36 for *Hepsetus odoe*. Mean mercury levels in sediment and water were 69.07±32.20 ng/g and 0.06 ±0.03 ng/L respectively.

There was a significant correlation between Hg concentration in fish muscle and fresh weight of fish for *Lates niloticus* (r^2 = 0.505) and an inverse correlation between Hg concentration in fish muscle and fresh weight of fish for *Synodontis oceillifer* (r^2 = 0.836). A good correlation between Hg concentration in fish muscle and total length of fish was also observed for *Lates niloticus* (r^2 = 0.604) and *Hydrocynus forkali* (r^2=0.545) and an inverse correlation between Hg concentration and total length of fish was observed for *Synodontis oceillifer* (r^2 = 0.911) and *Synodontis sp* (r^2 = 0.682). All the rest of the fish species showed poor correlation between Hg concentration in muscle and total length and fresh weight.

In general, mercury concentration in fish from Dam site ranged between 35.02 to 345.89ng/g. *Hydrocynus forkali* recorded the highest level of 345.89 ng/g whilst the lowest recorded Hg concentration of 35.02 ng/g was in *Brycinus imberi*.

Positive correlation between Hg concentration in *Synodontis Oceilifer* and river sediment was observed at Bator. Correlation between Hg concentration in fish and in sediment at

the other two sampling sites was not significant. The correlation between Hg concentration in fish and water from the two sampling site was insignificant with the exception of *Synodontis Oceilifer* which showed significant correlation (r^2= 0.685) at Bator. No correlation was observed between the total Hg concentration in the sediment and the water at any of the sampling sites.

All the fish samples studied showed mercury concentrations below the Word Health organization (WHO) limit of 500 ng/g wet weight. The results obtained from this study therefore showed that fish from the Black Volta River Basin are unlikely to constitute a significant mercury exposure to the public through consumption before the construction of the hydroelectric reservoir.

TABLE OF CONTENTS

LISTS OF FIGURES

LIST OF TABLES

ABBREVIATIONS

AAS	Atomic Absorption Spectrophotometry
CVAAS	Cold Vapour Atomic Absorption Spectrophotometry
IAEA	International Atomic Energy Agency

CHAPTER ONE

1. INTRODUCTION

Mercury (Hg) is found everywhere in the environment. It is in the air, soil and vegetation, as well as in lakes and rivers. Mercury can be released into the air naturally, by volcanoes and forest fires, or as a result of human activities such as coal burning and waste incineration. Mercury can be transported through the atmosphere over long distances, and falls into lakes and forests with dust particles and rain. But this mercury, present mainly in inorganic form, is relatively harmless, because it is not readily assimilated by living beings (Schetagne and Plante, 2006). However, once it enters lakes and rivers, inorganic mercury is converted by bacteria that process sulphate (SO_4) into methylemercury that is easily taken up by living organisms (Schetagne and Plante, 2006). This form of mercury (methylmercury) can become toxic in high concentrations. The concentrations of methylmercury increase as it passes through the food chain, from plankton to aquatic insects and on to fish. Therefore, fish such as northern pike and walleye that eat other fish contain more mercury than fish such as Lake Whitefish and brook trout that feed on insects (Schetagne and Plante, 2006).

Human exposure to methylmercury is therefore through the consumption of fish. The levels of methylmercury may reach hazardous levels in humans through repeated consumption of contaminated fish. Hg toxicity is well established and its dangers to people have been well-known and several cases of Hg toxicity in the environment have been reported (USEPA, 2001). The most serious occurred in Minamata Bay area of Japan from 1953 – 1960 as a result of Hg, released into the bay from manufacturing plants. Mercury levels of 5 to 20ppm were found in seafood eaten by 111 people

diagnosed with "Minamata disease". Of these, 45 died as a result of an apparent poisoning (Yoshino *et al.*, 1966; Irukayama *et al.*, 1977).

Mercury contamination of fish as a result of the construction of hydroelectric dams to generate power and to supply community water has motivated recent scientific researches (Bodaly *et al.*, 1997; Kelly *et al.*, 1997; Akagi and Ikingura, 2003). Mercury bioaccumulation process in fishes in man-made reservoirs is a phenomenon that has been recognized in several countries such as U S A, Canada, Sweden, Finland, Brazil. In many cases, no specific pollutant source was identified and many occurrences of elevated Hg levels in tissues of fish have been detected in regions considered to be remote from sources of Hg (Bodaly *et al.*, 1984; Verdon *et al.*, 1991; Schetagne *et al.*, 2000).

Several studies worldwide have indicated considerable mercury levels in fish samples collected from hydroelectric power generation reservoirs (Bodaly *et al.*, 1984 and 1990). The increase in fish mercury levels was generally believed to be a consequence of increased bacterial methylation of elemental Hg to form methylmercury, stimulated by benthic conditions as a result of decomposition of flooded organic materials (Bodaly *et al.*, 1984; 1990; Coquery, 2001). In recently flooded hydroelectric reservoirs, the green parts of the vegetation, in other words the ground cover, leaves and moss, provide food for the bacteria that convert inorganic mercury to methylmercury. Fish in and downstream of reservoirs consequently contain more mercury shortly after flooding. However, the phenomenon is temporary, since the green parts of the vegetation are quickly decomposed by the bacteria. Hence the duration and extent of Hg elevation in

fish from new reservoirs cannot be reliably predicted, since there are differences between regions and different characteristics of the reservoirs (Rodgers *et al.*, 1995).

Methylmercury, which can be absorbed directly from water across the gills or be obtained from food, is the form in which most of the mercury is concentrated in fish. Fish and seafood retain methylmercury, of which up to 90% can be stored in their body tissues and transferred to humans on consumption (Harris and Snodgrass, 1993; Ullrich *et al.*, 2001). Methylmercury is dominant in edible flesh of fish and aquatic mammals making fish a primary source of mercury in the human diet (Clarkson, 1992). Since elimination is slow compared to the relative rate of uptake, and the effects are presumed to be irreversible, there is bioaccumulation.

In the last decades, numerous studies have shown that a relationship exists between elevated levels of mercury in fish and inundation of land due to construction of hydroelectric power plants. Mercury concentrations have been reported to increase significantly, often by three to four times, in newly formed hydroelectric reservoirs (Bodaly *et al.*, 1984; Verdon *et al.*, 1991). In many cases, the recommended dietary limit has been exceeded. A range of factors influence the bioavailability as well as the bioaccumulation of mercury in freshwater biota. Mercury bioavailability is determined by a biotic factor and often increased at low pH, high concentration of organic matter or low O_2 saturation (Verta, 1984; Tropp, 2000).

The increase of mercury bioavailability due to impoundments is usually related with quality and amount of flooded vegetation, bacterial activity in sediments and high level

of humosity of the surface waters, (Ramsey, *et al*., 1986; Stokes, *et al*., 1987; Jackson, 1988). Dissolved organic acids, abundant in dark water aquatic systems, increase the reactivity of all forms of mercury both present in flooded sediments and deposited from atmosphere (Meech *et al*., 1998; Costa *et al*., 1999; Sjoblom *et al*., 2000). The recent discovery of water-soluble species of mercury in the atmosphere, usually produced by coal-wood combustion, named reactive gaseous mercury (RGM), has heightened concerns that this form of mercury can react quickly in large surface reservoirs increasing the bioavailability of the pollutant (Lindberg and Stratton, 1998; Lindberg, 1999).

The risk of elevated mercury (Hg) concentrations in fish has become one of the most important issues in assessing the environmental impact of hydroelectric reservoirs. Although the primary reason is not a direct Hg contamination, the problem is anthropogenically caused and the effect on human health can be as severe as when Hg itself is emitted into water systems (Wasserman *et al*., 2003). The increase in Hg concentrations in fish when areas are flooded at newly constructed reservoirs has mainly been studied in temperate areas, where predatory and non-predatory fish species responded to impoundment with increased levels of Hg within two years (Lodenius *et al*.,1983; Bodaly *et al*., 1984 ; Porvari, 1998).

According to Porvari (1998), model predictions of Hg levels in pike in a planned reservoir in Northern Finland showed that the levels would exceed 1mg Hg/kg for the first 12 years after the flooding. The timing of increase and decline in Hg levels however depends on the species as was shown in Canada (Porvari, 1998). Some species

in reservoirs had a steep increase but only a short period of increased Hg concentration, while others had a flatter curve of increased Hg concentration, extending over a longer period (Jackson, 1991). A comprehensive data that exist for dams constructed 6 – 67 years ago indicate that it could take 20 – 30 years before the Hg concentrations in fish return to pre-impoundment levels (Verdon *et al.*, 1991).

Monitoring of reservoir fish has also shown that mercury levels in insect-eating fish such as Lake Whitefish return to levels equivalent to those in natural lakes after 10 to 20 years (Schetagne and Plante, 2006). In fish that feed on other fish, such as northern pike, the return to normal levels takes longer and is usually complete only after 20 to 30 years. The increase in mercury levels is temporary, because the main mechanisms involved in the production of methylmercury and its transfer to fish are intense shortly after reservoir impoundment but are relatively short-lived. The increased methylmercury production generally ends 8 to 10 years after impoundment, due to the rapid depletion of the readily decomposable components of the flooded soil and vegetation, which provide food for the bacteria that convert the inorganic mercury to methylmercury. After this, methylmercury transfer to fish by periphyton, zooplankton and insect larvae is greatly reduced (Schetagne and Plante, 2006).

Fish has always been an important source of food for the people living in the Black Volta River Basin and surplus catches made by local fishermen are sold for cash income. The Bui Dam proposed for the Black Volta River would be the third major dam in Ghana. It would flood nearly a quarter of the Bui National Park (IRN, 2001).

A recent survey has revealed that the Black Volta River abounds with 46 species of fish from 17 families, all of economic importance (WRM, 2006). Elevated Hg concentrations in fish from the Black Volta River during the construction phase and years after the construction may affect the health of local people.

Methylmercury is highly toxic, and the nervous system is its principal target tissue. In adults, the earliest effects are non-specific symptoms such as paresthesia, malaise, and blurred vision: with increasing exposure, signs appear such as concentric constriction of the visual field, deafness, dysarthria, ataxia, and ultimately coma and death (Harada, 1995). The developing central nervous system is more sensitive to methylmercury than the adult. In infants exposed to high levels of methylmercury during pregnancy, the clinical picture may be indistinguishable from cerebral palsy caused by other factors, the main pattern being microcephaly, hyperreflexia, and gross motor and mental impairment, sometimes associated with blindness or deafness (Harada, 1995; Takeuchi and Eto, 1999). In milder cases, the effects may only become apparent later during the development as psychomotor and mental impairment and persistent pathological reflexes (WHO/IPCS, 1990; NRC, 2000). Studies from one population exposed to methylmercury from fish also suggest an association with increased incidence of cardiovascular system diseases (Salonen *et al.*, 1995, Rissanen *et al.*, 2000).

Methylmercury production occurs mainly in sediments and the rate depends on levels of Hg in sediment in addition to other factors. Considerable data exist on the accumulation of mercury in sediments of numerous lakes (Koeman *et al.*, 1975). Recent mercury accumulation rates in sediments agree fairy well with limited measurements of

atmospheric mercury deposition. Thus river sediments may be useful archives of historical, natural and anthropogenic inputs of mercury in rivers. In 1970, analysis carried out showed that more than 90% of the surface sediments in Onondaga Lake contained mercury at concentrations greater than 0.10ppm. In that same year, mercury levels in fish were found to exceed 0.50ppm, the maximum permissible levels established by WHO. Some fish had mercury concentrations as high as 3.6ppm (Hunter *et al.*, 1987).

An extensive body of literature documenting a positive relationship between fish mercury concentration, size and length within an individual water body exist (Lange *et al.*, 1994). Good correlation normally existed among carnivorous species while herbivorous species normally show poor correlation (Lange *et al.*, 1994). Mercury in fish measured in Deep Creek pickerel in 1992 showed that the fish examined that was 48cm long contained 0.98 mgHg/kg whereas those with length 20cm long had average Hg concentration of 0.3mg/kg (Gremillion *et al.*, 2004)

It is therefore important to study the effects of impoundments, especially the influence on fish Hg levels due to Hg possibly becoming more bioavailable after inundation. This can only be achieved when current levels of mercury contamination in the reservoir are established. In Ghana, a study conducted on the levels of mercury in fish from the Akosombo and Kpong hydroelectric reservoirs indicates low levels of mercury which are well below the WHO's standard value of 0.5ppm (Agorku *et al.*, 2008). However, there was no available data on the pre-impoundment levels of mercury in fish from the reservoirs. Hence it was not possible to indicate whether there was an initial rise in Hg

levels after impoundment for sometime before returning to the current levels which can be considered background. Hence this research will serve as the basis for assessing long term mercury trends during the construction phase of the Bui Hydroelectric and years after the construction.

1.1 Research Objectives:

The objectives of this research are:

- To determine the total mercury concentrations in various fish species, sediment and water samples from the Black Volta River.
- To determine if there is any correlation between total mercury concentration in fish and sediments from the study area.
- To determine if there is any correlation between total mercury concentration in fish and water.
- To determine if there is any correlation between total mercury concentration in sediment and water.
- To determine if there is any correlation between total mercury concentration and size as well as length of fish.
- To determine whether the levels of mercury in fish from the river are at levels of potential human health concern.

CHAPTER TWO

2. LITERATURE REVIEW

2.1 Chemical and Physical Properties of Mercury

Mercury occurs naturally in the environment and exists in a large number of forms. Like lead or cadmium, mercury is a constituent element of the earth. In pure form, it is known as "elemental" or "metallic" mercury (expressed as Hg^0). Mercury is rarely found in nature as the pure, liquid metal, but rather within compounds and inorganic salts. Mercury can be bound to other compounds as monovalent or divalent mercury (expressed as Hg (I) and Hg (II) or Hg^{2+}, respectively). Many inorganic and organic compounds of mercury can be formed from Hg (II).

Several forms of mercury occur in the environment. The most common forms found in the environment are metallic mercury, mercuric sulphide, mercuric chloride and methylmercury. Some micro-organisms and natural processes can change the mercury in the environment from one form to another. Elemental mercury in the atmosphere can undergo transformation into inorganic mercury forms, providing a significant pathway for deposition of emitted elemental mercury.

The most common organic mercury compound that micro-organisms and natural processes generate from other forms is methylmercury. Methylmercury is of particular concern because it can build up (bioaccumulate and biomagnify) in many edible fish and marine mammals to levels that are many thousands of times greater than levels in the surrounding water (WHO, 1976).

Methylmercury can be formed in the environment by microbial metabolism (biotic processes) and by chemical processes that do not involve living organisms (abiotic processes). Although, it is generally believed that its formation is predominantly due to biotic processes, significant direct anthropogenic (human-generated) sources of methylmercury are currently not known, although historic sources have existed (WHO, 1976). Indirectly, however, anthropogenic releases contribute to the methylmercury levels found in nature because of the transformation of other forms. Examples of direct release of organic mercury compounds are the Minamata methylmercury-poisoning event that occurred in the 1950's where organic mercury by-products of industrial-scale acetaldehyde production were discharged into the local bay (WHO, 1976). Also, new research has shown that methylmercury can be released directly from municipal waste landfills (Lindberg *et al.*, 2001) and sewage treatment plants (Sommar *et al.*, 1999), but the general significance of this source is still uncertain.

2.2 Sources of Environmental Mercury Pollution

The sources of environmental Hg pollution include natural occurrence and anthropogenic sources.

2.2.1 Natural Occurrence

Natural sources include volcanoes, evaporation from soil and water surfaces, degradation of minerals and forest fires. Mercury in small, but varying concentrations can be found virtually in all geological media. Elemental and some forms of oxidized mercury are permanently coming to the atmosphere due to their volatility. High temperature in the earth mantle results in high mercury mobility and mercury

continuously diffuses to the surface. In the zones of deep geological fractures these processes go on more intensively. Here are located so-called mercury geochemical belts where mercury concentrations in the upper layer appreciably exceed their average values. In some parts of mercury belts the intensive accumulation of mercury resulted in the formation of extractable deposits (Jonasson and Boyle, 1971; Bailey *et al.*, 1973). Regions with high concentrations in surface rocks are characterized by high mercury emissions to the atmosphere.

The natural mercury emissions are beyond control, and must be considered part of local and global living environment. In some areas of the world, the mercury concentrations in the earth's crust are naturally elevated, and contribute to elevated local and regional mercury concentrations in those areas. Today's emissions of mercury from soil and water surfaces are composed of both natural sources and re-emission of previous deposition of mercury from both anthropogenic and natural sources. This makes it very difficult to determine the actual natural mercury emissions. For example, total estimates of re-emission from soil and water surfaces in Europe exist, but they include mercury originating from both natural and anthropogenic sources (Pirrone *et al.*, 2001).

Attempts to directly measure natural emissions are ongoing (Coolbaugh *et al.*, 2002). Nonetheless, available information indicates that natural sources account for less than 50 percent of the total releases. A number of attempts have been made to estimate the regional and global natural emissions of mercury. It is, however, difficult to do so with any precision and research is still done in this field at several institutions (AMAP, 2000).

2.2.2 Anthropogenic Sources

Mercury is naturally present in coal and other fossil fuels, as well as in minerals like lime for cement production and soils (such as agricultural soils subject to acidification management) and metal ores including for example zinc, copper and gold ore. Coal-fired power production is today deemed the single largest global source of atmospheric mercury emissions (Pacyna and Pacyna, 2000). This is due to the increasing global power consumption, and also to the fact that emissions from intentional use of mercury are gradually diminishing in many of the industrialised countries.

A large portion of the mercury present in the atmosphere today is the result of many years of releases due to anthropogenic activities. The natural component of the total atmospheric burden is difficult to estimate, although a study by Munthe *et al*., (2001) has suggested that anthropogenic activities have increased the overall levels of mercury in the atmosphere by roughly a factor of 3. While there are some natural emissions of mercury from the earth's crust, anthropogenic sources are the major contributors to releases of mercury to the atmosphere, water and soil.

Available global estimates of atmospheric emissions from waste incineration, as well as other releases originating from intentional uses of mercury in processes and products, are deemed underestimated and to some degree incomplete. Anthropogenic emissions from a number of major sources have decreased during the last decade in North America and Europe due to reduction efforts (UNEP, 2002).

The intentional use of mercury in products and processes is still deemed a significant source of mercury to the environment. The recorded global primary production of virgin

mercury is still large compared to current estimates of global atmospheric mercury emissions. When assessing the releases of mercury to the environment, it is generally difficult to quantify diffuse releases from the life cycle of mercury-containing products. These sources have not always been included fully in regional or global inventories for mercury releases to the environment. Some national studies do however give a certain insight in the contributions from this source category. The contribution from intentional mercury uses in a number of products in the European region was also assessed by Munthe and Kindbom (1997). They found that in the mid-1990's three dominating groups of intentional mercury uses in products contributed about 18 percent of the total mercury emissions to air in this region. Additional contributions from dental amalgam use were not included in the assessment.

2.3 Methylation of Mercury

The methylation of inorganic mercury in the sediment of lakes, rivers and other waterways, as well as in the oceans, is a key step in the transport of mercury in aquatic food chains. It was first demonstrated by Jensen and Jernelov, (1967) that microorganisms in lake sediments could methylate mercury. They later showed that the degree of methylation correlated well with the overall microbial activity in the sediment (Jensen and Jernelov, 1969). The following general conclusions have been drawn by Bisogni and Lawrence, (1973) concerning methylation by microorganisms:

a) mono-methylmercury is the predominant product of biological methylation near neutral pH.

b) the rate of methylation is greater under oxidising conditions than under anaerobic conditions.

c) the output of methylmercury doubles for a ten-fold increase in inorganic mercury,

d) temperature affects methylation as a result of its effect on overall microbial activity,

e) higher microbial growth rate increases mercury methylation,

f) methylation rates are inhibited by the addition of sulfide to anaerobic systems.

The formation of new or enlarged artificial lakes considerably increases the production of methylmercury, although this increase was found to be short-lived in new lakes in Finland (Simola and Lodenius, 1982; Alfthan *et al.*, 1983).

A similar problem of increased mercury in new lakes, which was taken up by fish and fish-eating mammals, occurred in the scheme to divert the Churchill River in Manitoba, Canada (Canada-Manitoba, 1987). Methylation rates in one lake, which had been flooded 20 years previously, had returned to normal. Methylation rates in the new lake, which had flooded arboreal forest, were high and were expected to remain high for decades. The source of mercury in all of these artificial lakes appeared to be natural rather than anthropogenic in origin. Anaerobic conditions after the flooding of large amounts of organic material and the subsequent increase in microbial activity are thought to be the causes of the increased availability of mercury through methylation.

2.3.1 Mercury species and transformation in aquatic environments

The formation of methylmercury in aquatic systems is influenced by a wide variety of environmental factors. The efficiency of microbial mercury methylation generally depends on factors such as microbial activity and the concentration of bioavailable mercury (rather than the total mercury pool), which in turn are influenced by parameters

such as temperature, pH, redox potential and the presence of inorganic and organic complexing agents (Ullrich *et al.*, 2001).

Certain bacteria also demethylate mercury and this tendency increases given increasing levels of methylmercury, thereby forming some natural constraints on build-up of methylmercury (Marvin-Dipasquale *et al.*, 2000; Bailey *et al.*, 2001). Since both methylation and demethylation processes occur, environmental methylmercury concentrations reflect net methylation rather than actual rates of methylmercury synthesis. Numerous bacterial strains capable of demethylating methylmercury are known, including both aerobic and anaerobic species, but demethylation appears to be predominantly accomplished by aerobic organisms. Bacterial demethylation has been demonstrated both in sediments and in the water column of freshwater lakes. Degradation of methyl and phenyl mercury by fresh water algae has also been described (Ullrich *et al.*, 2001). Purely chemical methylation of mercury is also possible if suitable methyl donors are present. The relative importance of abiotic versus biotic methylation mechanisms in the natural aquatic environment has not yet been established, but it is generally believed that mercury methylation is predominantly a microbially mediated process (Ullrich *et al.*, 2001).

Methylmercury is the predominant mercury species in fish. The United States Environmental Protection Agency (US EPA) states in a mercury overview paper that in most adult fish, 90 to 100 percent of mercury content is methylmercury (US EPA, 2001). As a consequence, the US EPA recommends that the cheaper total mercury chemical analysis be used for evaluation of risk from consuming local fish, and that results should

be used as if mercury was present as 100 percent methylmercury in order to be most protective of human health.

Mason and Fitzgerald (1996; 1997) have reviewed aspects of the cycle of mercury in oceans and other waters. From open ocean studies, it is apparent that elemental mercury, dimethylmercury and, to a lesser extent, methylmercury are common constituents of the dissolved mercury pool in deep ocean waters. In open ocean surface waters dimethylmercury is lacking, may be as a result of decomposition in the presence of light and an additional potential loss via evaporation from the water surface. Recent results suggest that low oxygen conditions are not necessary for the formation of dimethylmercury in the open oceans. Studies in freshwater and estuarine environments have shown that methylation of mercury is primarily taking place under low oxygen conditions and mainly by sulphate-reducing bacteria. Here methylmercury is the product of methylation of ionic mercury.

2.4 Toxicology of mercury

The toxicity of mercury depends on its chemical form, and thus symptoms and signs are rather different in exposure to elemental mercury, inorganic mercury compounds, or organic mercury compounds such as alkylmercury compounds, methylmercury and ethylmercury salts, and dimethylmercury. The sources of exposure are also markedly different for the different forms of mercury. For alkylmercury compounds, among which methylmercury is by far the most important, the major source of exposure is diet, especially fish and other seafood. For elemental mercury vapour, the most important source for the general population is dental amalgam, but exposure at work may in some

situations exceed this by many times. For inorganic mercury compounds, diet is the most important source for the majority of people. However, for some segments of populations, use of skin-lightening creams and soaps that contain mercury; and use of mercury for cultural or ritualistic purposes or in traditional medicine, can also result in substantial exposures to inorganic or elemental mercury. While it is fully recognised that mercury and its compounds are highly toxic substances for which potential impacts should be considered carefully, there is ongoing debate on how toxic these substances, especially methylmercury, are. New findings during the last decade indicate that toxic effects may be taking place at lower concentrations than previously thought, and potentially larger parts of the global population may be affected (UNEP, 2002).

2.4.1 Toxicity of Methylmercury

Like other alkylmercury compounds, the toxicity of methylmercury is much higher than that of inorganic mercury. Methylmercury is a potent neuro-toxin, hence human exposure to methylmercury is clearly unwelcome and should be regarded with concern. It is present worldwide in fish and marine mammals consumed by humans. Methylmercury is formed naturally (from anthropogenic and naturally released mercury) by biological activity in aquatic environments, and it is bio-magnified in the food chain, resulting in much higher concentrations in higher predatory fish and mammals than in water and lower organisms. Most of the total mercury concentrations in fish are in the form of methylmercury (close to 100 percent for older fish). Most people are primarily exposed through the diet, above all through consumption of fish which is an extremely valuable component of the human diet in many parts of the world. In 1991, a joint committee from FAO and WHO revised guideline levels for mercury in fish aimed at

human consumption. The recommended dietary limit was set at 0.5 μg Hg/g wet weight for non predatory fish and 1.0 μg Hg/g wet weight for predatory fish (FAO/WHO. 1991).

Consumption of contaminated fish and marine mammals is the most important source of human exposure to methylmercury (WHO/IPCS, 1990; US EPA, 1997). The highest concentrations are found in large predatory fish like shark, king mackeral, swordfish and tuna, as well as in some freshwater fish like pike, walleye, bass, perch, and eels, and in mammals like seals and whales (UNEP, 2002). Mercury bioaccumulation in fish is influenced by physiological factors such as sex, age, size, growth rate or metabolic rate (Huckabee *et al.*, 1979) and ecological factors such as trophic position or food chain length (Cabana *et al.*, 1994; Kidd *et al.*, 1995). Due to long-range atmospheric emission transport and ocean currents, methylmercury is also present in the environment far away from local or regional mercury sources. This implies that population groups particularly dependent on or accustomed to marine diets, such as the Inuits of the Arctic, as well as marine and freshwater fish dependent populations anywhere else on the globe, are particularly at risk due to methylmercury exposure.

Methylmercury is highly toxic, and the nervous system is its principal target tissue. In adults, the earliest effects are non-specific symptoms such as paresthesia, malaise, and blurred vision; with increasing exposure, signs appear such as concentric constriction of the visual field, deafness, dysarthria, ataxia, and ultimately coma and death (Harada, 1995). The developing central nervous system is more sensitive to methylmercury than the adult. In infants exposed to high levels of methylmercury during pregnancy, the clinical picture may be indistinguishable from cerebral palsy caused by other factors, the

main pattern being microcephaly, hyperreflexia, and gross motor and mental impairment, sometimes associated with blindness or deafness (Harada, 1995; Takeuchi and Eto, 1999). In milder cases, the effects may only become apparent later during the development as psychomotor and mental impairment and persistent pathological reflexes (WHO/IPCS, 1990; NRC, 2000). Studies from one population exposed to methylmercury from fish also suggest an association with increased incidence of cardiovascular system diseases (Salonen *et al.*, 1995; Rissanen *et al.*, 2000). From research on animals there is evidence of genotoxicity and effects on the immune system and the reproductive system.

2.5 Mercury concentrations and transformations in surface waters.

Freshwater ecosystems are among the most sensitive to Hg pollution. Under conditions of high total Hg loading, MeHg production can vary widely, depending on the methylation efficiency of a particular ecosystem (Krabbenhoft *et al.*, 1999). Mercury enters remote surface waters through direct atmospheric deposition and through soil water, wetland, or groundwater drainage. Streams and rivers can exhibit marked temporal variation in Hg concentrations, which is associated with variations in concentrations of dissolved organic carbon (DOC) or suspended matter. Large increases in Hg concentrations can occur during high flow events (Shanley *et al.*, 2005). Some inputs of Hg to lakes are removed from the water column by the volatilization of Hg^0 and by sediment deposition. In freshwater lakes, photochemical processes are largely responsible for the reduction of ionic Hg to Hg^0 (Amyot *et al.*, 1997). Microbial reduction has been observed in laboratory studies, but only at higher than ambient concentrations of Hg (Morel *et al.*, 1998). Biogeochemical processes in lakes also result

in net production of MeHg due to methylation in anoxic sediments and in the water column. Areas of elevated Hg concentrations in surface waters can be explained by high concentrations of DOC, as in the Adirondacks; by high inputs of suspended solids, from rivers along Lake Champlain, related to high flow events; and by elevated atmospheric Hg deposition, as in lakes in Southeastern New Hampshire and Eastern Massachusetts. A large portion of the variation in total Hg and MeHg across the region can be explained by variation in DOC (Dennis *et al.*, 2005). Areas with the highest mean surface water Hg concentrations also have the greatest range in Hg concentrations. This variation may be attributed to heterogeneity in watershed characteristics or to high flow events (Shanley *et al.*, 2005).

2.5.1 Mercury in sediment

It has been estimated that sediment is an important sink for both Hg and MeHg in the aquatic environment (Mason *et al.*, 1999) and after atmospheric deposition and runoff from surrounding catchments, Hg can be converted to MeHg from in situ production by natural bacteria in anoxic sediments and soils (Gilmour *et al.*, 1992). The amount of MeHg in aquatic regions varies among ecosystems, as does atmospheric Hg deposition. Therefore, MeHg bioaccumulation in fish does not only depend on how much Hg enters the ecosystem, but also on the ability of the ecosystem to convert that Hg to MeHg (Heyes and Gilmour, 1999). For example, methylation of Hg has been found to be enhanced in wetlands but can be produced in other anoxic regions as well. Increased runoff from highly urbanized areas and as the result of impervious surfaces in and around the watershed may contribute to higher than normal concentrations of Hg and MeHg in aquatic systems. Whereas MeHg has high affinity for particles and organic

matter, the extent to which sediment is a source of MeHg to the fish largely depends on the size of particles and organic matter content of the sediment (Benoit *et al.*, 1998; Mason, 2001).

2.5.2 Movements of mercury in and between environmental compartments

Mercury is a natural element that cannot be created or destroyed and the same amount has existed on the planet since the earth was formed. A significant amount of research indicates that natural and anthropogenic activities can redistribute this element in the atmospheric, soil and water ecosystems through a complex combination of transport and transformations.

Mercury is emitted to the atmosphere from a variety of point and diffuse sources and is dispersed and transported in the air, deposited to the earth and stored in or redistributed between water, soil and atmospheric compartments. Therefore, mercury cycling and mercury partitioning between different environmental compartments are complex phenomena that depend on numerous environmental parameters. Wet deposition was, until recently, assumed to represent the primary mechanism for transfer of mercury and its compounds from the atmosphere to aquatic and terrestrial receptors. However, studies by US EPA, the Florida Department of Environmental Protection and US Department of Energy have all shown that dry deposition of divalent gaseous mercury species can be equal or greater than wet deposition, even in moist climatic areas such as the Florida Everglades and the Great Lakes Region with relatively high annual precipitation (Rea *et al.*, 2000; 2001; Landis *et al.*, 2002; Vette *et al.*, 2002).

The chemical and physical form of mercury in air affects the mechanisms by which it is transferred to the earth surface and ultimately influences the total depositional flux. An increase in ambient air concentrations of mercury will result in an increase of direct human exposure and an increase of mercury flux entering terrestrial and aquatic ecosystems leading to elevated concentrations of methylmercury in freshwater and marine biota. Extensive research conducted on mercury deposition in Boreal forests systems has shown that the main source of mercury and methylmercury to the forest floor is litter fall (Iverfeldt, 1991, Munthe *et al.*, 1995). This mercury and methylmercury mainly originate from the atmosphere and adsorbs on plants surfaces via dry deposition. Figure 2.1 illustrates pathway of mercury into the environment.

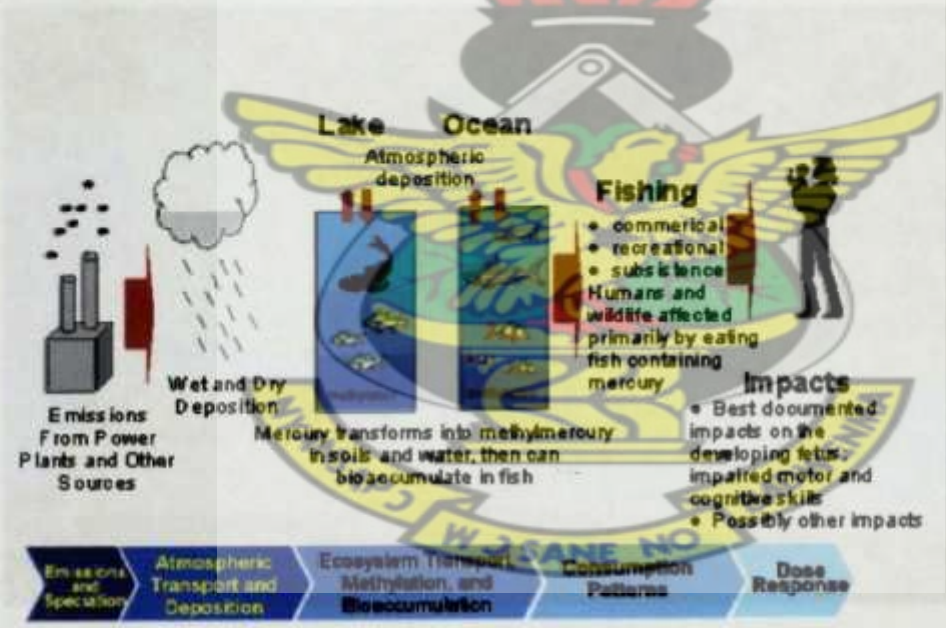


Figure 2.1 pathway of mercury into the environment
Source http://www.epa.gov/mercury/exposure.htm

2.6 Dams

Dams are structural barriers built to obstruct or control the flow of water in rivers and streams. They are designed to serve two broad functions. The first is the storage of water

to compensate for fluctuations in river discharge (flow) or in demand for water and energy. The second is the increase of hydraulic head, or the difference in height between water levels in the lake created upstream of the dam and the downstream river. By creating additional storage and head, dams can serve one or more purposes; generating electricity, supplying water for agricultural, industrial, and household needs, controlling the impact of floodwaters and enhancing river navigation. They can be operated in a manner that simultaneously augments downstream water quality, enhances fish and wildlife habitat, and provides for a variety of recreational activities, such as fishing, boating, and swimming (www.waterencyclopedia.com/Re-St/Reservoirs-Multipurpose.html). Figure 2.2 illustrates a dam.

Figure 2.2 A cross section of a hydroelectric dam

2.6.1 Types of Dams

Dams are of numerous types, and type classification is sometimes less clearly defined. However, broad classification into two generic groups can be made in terms of the principal construction material employed.

1. Embankment dams which are constructed of earthfill and/or rockfill. Upstream and downstream face slopes are similar and of moderate angle, giving a wide section and a high construction volume relative to height.
2. Concrete dams are constructed of mass concrete. Face slopes are dissimilar, generally steep downstream and near vertical upstream (Novak, *et al*, 2007).

Embankment dams are numerically dominant for technical and economic reasons, and account for an estimated 85 – 90% all dams built (Novak, *et al.*, 2007). The Bui dam in Ghana is an example of embankment dam.

2.6.2 Embankment Dams

An Embankment dam can be defined as a dam constructed from natural materials excavated or obtained close by. The materials are utilised to the best advantage in relation to their characteristics as an engineered bulk fill in defined zones within the dam section. Embankment construction is an almost continuous and highly mechanized process, weather and soil conditions permitting, and is thus plant intensive rather than labour intensive (Novak *et al.*, 2007).

2.6.3 Types of Embankment Dams

Embankment dams can be classified in broad terms as being earthfill or rockfill dams, depending upon how the available materials are utilised.

1. Earthfill Embankments

An embankment may be categorised as an earthfill dam if compacted soils account for over 50% of the placed volume of material. An earthfill dam is constructed primarily of

selected engineering soils compacted uniformly and intensively in relatively thin layers and at a controlled moisture content.

2. Rockfill Embankments

Rockfill embankment includes a discrete impervious element of compacted earthfill or a slender concrete or bituminous membrane. The designation rockfill embankment is appropriate where over 50% of the fill material may be classified as rockfill, i.e. coarse-grained frictional material.

2.6.4 Construction of Embankment Dams

The construction operations of embankment dams which follow initial site development fall into four principal groups of activities, namely (1) material source development (2) foundation preparation and construction (3) fill construction and control and (4) ancillary works construction.

Material source development activities involve the opening out of borrow areas or quaries, including the installation of fixed plant, e.g. crushers, conveyors etc. Access and haulage roads are also constructed between the various borrow areas and the embankment site, and excavation and haulage plant is mobilised.

Foundation preparation activities, including river diversion, can proceed concurrently with the development of the fill sources. Temporary river diversion is commonly effected by driving a flanking tunnel, which in most cases subsequently houses the outlet works. Topsoil and weathered surface drift deposits etc. are removed at this stage. Foundation instrumentation is also installed at this stage to monitor pore pressures and

cut-off performance. Foundation construction is completed with the laying of the drainage blankets which will underlie the downstream shoulder.

Fill construction is an exercise in efficient plant utilisation within the terms of the specification requirements as to materials compliance and compaction technique. Control of placing is centered upon supervision of water content, layer thickness and compaction procedure. The installation of instrumentation in the core and shoulders proceeds in parallel with the placing of fill. Fill construction is concluded with the completion of upstream rock armouring or other face protection works.

Ancillary works construction embraces the construction of spillway and stilling basins, culverts or tunnels for outlet works etc., Valve towers and similar control works. It also includes completion of crest details e.g. roadway, drainage works, wavewall etc. and, where climatic conditions allow, grassing of the downstream face slope.

2.6.5 Bui Dam

The main dam, located in Bui Gorge, will be a gravity roller compacted concrete dam with a maximum height of 110 m above the foundation level, or 90 m above ground level (ERM, 2006). The dam incorporates an emergency spillway and water intake to regulate and control the flow of water upstream and downstream of the dam respectively. The Bui Dam would flood nearly a quarter of the Bui National Park. It would also completely destroy habitats for the rare Black Hippopotamus. It is also home to a stunning collection of many globally endangered amphibians, lions, and various

primates. The dam would affect a large number of the native wildlife species, forcibly resettle about 2,600 people and affect thousands more (IRN 2001; UNEP, 1999).

2.6.6 Reservoir

A reservoir is the artificial body of water that forms adjacent to a storage dam. Many of the modern reservoirs that operate today in unison with dams serve two or more purposes. The most common purposes of these reservoirs are to generate hydroelectric power, provide flood control, store water, enable irrigation, and provide recreational opportunities.

2.6.7 Mercury in Reservoirs and Wetlands

Reservoirs and wetlands are often mentioned as sources of methylmercury due to the methylation of inorganic mercury in the sediment (UNEP, 2002). According to the (UNEP, 2002), the creation of reservoirs is an important source of mercury contamination of fish in Canada, because the mercury present in newly flooded land becomes more available, and then more toxic due to the increased rate of conversion to methylmercury. Most fish caught in new reservoirs have mercury concentrations that exceed the consumption limit of 0.2 mg/kg wet weight recommended by Health Canada for people who frequently consume fish (UNEP, 2002).

In an investigation of mercury in feathers of birds from a number of tropical locations, Burger, (1997) reported that although fish-eating birds generally had the highest mercury content, a similar content was found in Cattle Egrets from the Aswan dam area, although this species is an insect-eating bird. The author suggested that this may have

been caused by more methylmercury in the food web due to a recent flood in the area initiating the methylation process. An experiment in a wetland and pond at the Experimental Lakes Area in North western Ontario demonstrated that natural wetlands are important sites of mercury methylation, and that flooding of wetlands increases methylation rates by a factor of more than 30 (UNEP, 2002). Increased concentrations of methylmercury were found in water, the food chain and eventually fish. Monitoring of boreal reservoirs indicates that concentrations of methylmercury in fish may return to normal 10 to 50 years after flooding.

2.7 Environmental Impact of Dams

Although an inexpensive energy resource, the environmental damage caused by hydropower can be serious. The most obvious effect is that fish are blocked from moving up and down the river. When a dam is constructed, a river habitat is replaced by a lake habitat. Dams can create large reservoirs submerging what used to be dry land, producing many problems. This land is often composed of wetlands, which are important wildlife habitats, and low-lying flood plains, usually the most fertile crop land in the area. Population density is typically higher along rivers, leading to mass dislocation of urban centers.

Another problem that can occur when the land area behind the dam is flooded without proper preparation is that, as the plants and trees that would be submerged begin to rot, they will reduce the oxygen content of the water, killing off the plants and fish in the water. Moreover, the rotting plants will give off large quantities of methane, a powerful global warming gas. A similar problem occurred in Canada, in hydro projects built by

Hydro Quebec (http://www.irn.org/programs/threeg/resettle.html). The stones and soil in the flooded area contain naturally occurring mercury. When the land was flooded, the mercury dissolved into the water, and then into the local fish populations. The creatures that ate the fish from bears and eagles, to the native Cree people suffered mercury poisoning. (http://www.irn.org/programs/threeg/resettle.html).

Impoundments used for hydropower can cause many other effects on water quality and aquatic life. Rivers and lakes can be filled with sediment from erosion. Water falling over spillways can force air bubbles into the water, which can be absorbed into fish tissue, ultimately killing the fish. By slowing down rivers, the water can become stratified, with warm water on top and cold water on the bottom. Since the cold water is not exposed to the surface, it loses its oxygen and becomes uninhabitable for fish.

Another important habitat disruption comes from the operation of the dam to meet electric demand. Water is stored up behind the dam and released through the turbines when power demand is greatest. This causes water levels to fluctuate widely on both sides of the dam, stranding fish in shallow waters and drying out the habitat. There are many competing pressures on dam operators to produce power, to provide water for recreational use both on the reservoir and downstream, to provide drinking and irrigation water, and to preserve habitat for fish and plant species. In many cases, nature loses out to boaters, farmers, and electric customers.

It is also important to compare the environmental effects of hydropower with alternatives. The damage to aquatic habitat from dams may be significant, but acid rain,

nitrogen deposition, and thermal pollution from coal plants also lead to aquatic damage, as well as to air pollution and global warming.

2.8 Mobilisation of mercury due to changes in land use

Under some conditions anthropogenic changes in land use may result in substantial mobilisation of mercury already present in the environment (originating from natural and/or anthropogenic sources). For example, in some environments, anthropogenic modifications including farmlands, recent clear-cuttings and water reservoirs (hydroelectric, aquaculture, irrigation) may considerably enhance the release of mercury into aquatic systems and the bio-accumulation of mercury in organisms. There is a growing body of evidence that the soils of forested watersheds contain considerable stores of both methylmercury and inorganic divalent forms. Both in North America and in Northern Europe, evidence is gradually accumulating, which points to the effect of terrain disturbance as a factor in the mobilisation and transport of both the inorganic and methylmercury stored in watersheds, and apparently also in the production of methylmercury (UNEP, 2002). Investigations in connection with hydro-electric reservoirs revealed the importance of understanding transport phenomena involving flooded soils. Watershed-scale hydrology is emerging as an increasingly important explanatory variable (UNEP, 2002).

CHAPTER THREE

3. MATERIALS AND METHODS

3.1 The Study Area

The study was carried out on the Black Volta River at the Bui dam site in north-western part of the Brong Ahafo Region of Ghana. The project area drains into the Volta Lake by the Black Volta River and other numerous tributaries of different sizes and lengths. The Black Volta had its headwaters in Burkina Faso, where it is called the Mohoun River. From there, it flows some 400 km to the northeast before it is joined by the Sourou River, with a combined catchment area to this point of some 47,000 km^2. Downstream of this confluence, the Mohoun flows southeast, then south for a further 510 km before reaching the Koulbi-Noumbiel dam site. The Black Volta then flows directly south until it reaches the Bui dam site 200 km downstream of Noumbiel. It has a catchment area of 123,000 km^2. It then forms a big loop northeastwards to enter the Volta Lake near Mpaha (ERM, 2006). The map of Ghana showing the geographical layout of the Black Volta River and sampling sites is shown in figure 3.1 below.

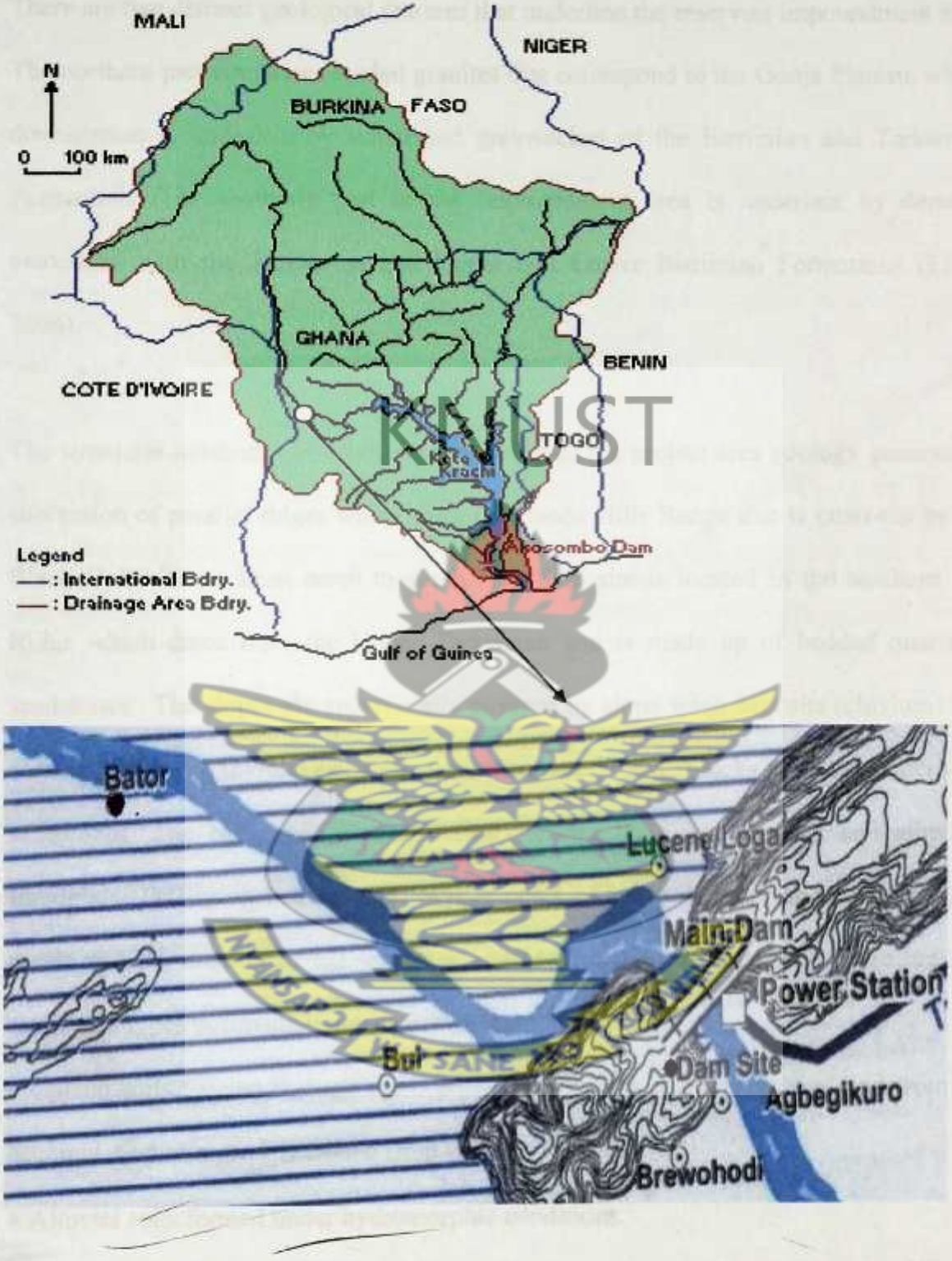


Fig. 3.1 Map of Ghana showing the geographical layout of Black Volta and the sampling sites.

3.1.1 Geology of the Area

There are two distinct geological features that underline the reservoir impoundment area. The northern part comprises eroded granites that correspond to the Gonja Plateau, whilst downstream is underlain by schist and greywackes of the Birrimian and Tarkwaian Formations. The southerly part of the impoundment area is underlain by deposits associated with the Tarkwaian and Upper and Lower Birrimian Formations (ERM, 2006).

The structural north-east to south-west direction of the project area geology generates a succession of parallel ridges which form the Banda Hills Range that is cross-cut by the Black Volta River, from north to south. The dam site is located in the southern Bui Ridge which dates from the Upper Tarkwaian and is made up of bedded quartzitic sandstones. The slopes are superficially covered by slope wash deposits (eluvium) that correspond to sandy-clayey soils with fragments and blocks of the underlying sandstones. The banks are generally characterized by Talus deposits and alluvium floodplain. Overlying bedrock are various types of loose deposits and accumulations resulting from erosion of the upstream and local bedrock formations. There are two major soil types occurring in the project area and these are:

- Upland soils formed through the deposition of parent materials in situ, and from the accumulation of materials eroded from upper slopes; and
- Alluvial soils formed under hydromorphic conditions.

These soils have been developed over weathering products of a number of geological formations including Tarkwaian, Birimian, Voltaian, Granitic rocks, and sediments of

the Black Volta and its major tributaries. Small scale mining operations were said to be carried out along the river.

3.2 Apparatus

All glassware used were soaked in detergent solution overnight; rinsed with distilled water and soaked in 10% (v/v) HNO_3 overnight. They were rinsed with distilled water followed by 0.5% (w/v) $KMnO_4$ and finally rinsed with distilled water and dried before use.

Automatic Mercury Analyzer Model HG-5000 (Sanso Seisakusho Co., Ltd, Japan), equipped with mercury lamp operated at a wavelength of 253.7 nm was used for the determinations. The signals were obtained on a Yokogawa strip chart recorder Model 3021.

Digestion apparatus was thick walled 50 ml volumetric digestion flasks and a Clifton hot plate.

3.3 Reagents

All reagents used were of analytical reagent grade (BDH Chemicals Ltd, Poole, England) unless otherwise stated. Doubled distilled water was used for the preparation of all solutions.

Mercury stock standard solution (1000 mg L^{-1}) ml was prepared by dissolving 0.0677 g of $HgCl_2$ in 14ml of an acid mixture of HNO_3:$HClO_4$:H_2SO_4 (2:2:10) in a 50 ml digestion flask with heating on a hot plate at a temperature of 200^0 C for thirty (30) minutes. The working standard solutions were freshly prepared by diluting an

appropriate aliquot of the stock solution through intermediate solutions using blank solution. Blank solutions were prepared by adding 1ml of distilled H_2O, 2ml of HNO_3 and $HClO_4$ (1:1) and 5ml of H_2SO_4 in a digestion flask. The mixture was heated at 200^0C for 30 minutes. Distilled water was added later to make up to 50ml after the mixture was allowed to cool.

Stannous Chloride solution (10% w/v) was prepared by dissolving 10 g of the salt in 100 ml 1M HCl solution. The solution was aerated with nitrogen gas at 50 ml min^{-1} for 30 min to expel any elemental mercury from it.

Sulfuric acid solution (10M) was prepared by transferring about 100ml of distilled water into 250ml volumetric flask and 135ml of concentrated (18.4M) H_2SO_4 was added gradually while stirring in an ice bath. Distilled water was later added to make a final volume of 250ml after it gains room temperature.

Potassium permanganate solution (0.5% w/v) was prepared by dissolving 0.5g of $KMnO_4$ in distilled water to make a final volume of 100ml.

Sodium hydroxide solution (10M) was prepared by dissolving 200g of NaOH in distilled water to make a final volume of 500ml.

Hydroxylamine hydrochloride solution (10% w/v) was prepared by dissolving 10g of $NH_2OH.HCl$ in distilled water to make a final volume of 100ml.

Ethylenediaminetetraacetate solution (10% w/v) was prepared by dissolving 10g of disodium salt of EDTA in distilled water to make a final volume of 100ml.

Dithizone-toluene solution (0.01% w/v) was prepared by dissolving 0.01g dithizone in 100ml of toluene.

3.4 Sampling and Sample preparation

3.4.1 Fish

The fish species were collected from random commercial catches made by the local fishermen between October 2007 and March 2008 in six batches, depending on the species available for sale. The samples obtained were reflective of species meant for consumption. A total of one hundred and eighty five (185) samples covering twenty-one (21) species were obtained from the two sites namely Bator and Dam site. The samples were placed in clean plastic bags and stored on ice in an ice chest. They were then transported to the laboratory, identified and kept in a freezer prior to preparation for chemical analysis. The fish samples were later taken from the freezer and allowed to thaw. They were washed with distilled water, dried on tissue paper and the length and body weight of each were taken. A portion of the edible muscle tissue was removed from the dorsal part of each fish, homogenized and stored in transparent polythene bags and kept in a freezer until chemical analysis.

3.4.2 Sediment

Ten sediment samples were collected each month from ten (10) different locations at each site namely Bator and Dam site at 5m intervals along the banks of the Black Volta River using the grab method. The samples were stored in polythene bags and kept cool during transportation to the laboratory. Sediment samples were air-dried and sieved through a 2mm nylon sieve. A total of one hundred and twenty (120) sediments samples were collected within six months (October to March).

3.4.3 Water

The sampling containers were soaked in 10% HNO_3 for 24 hours, rinsed under running tap water, followed by 0.5% (w/v) $KMnO_4$ and finally rinsed with distilled water before use. Water was sampled into the cleaned plastic containers (5m intervals) each month after rinsing the container with the water to be sampled from three locations at each site namely Bator and Dam site. The containers were filled with water from ten centimeters below the water surface. The three samples were combined together to form one composite sample and preserved with 5 ml concentrated HNO_3/L and transported to the laboratory prior to chemical analysis. A total of twelve (12) water samples were collected at both sites.

3.5 Digestion Procedure for Fish and Sediment

The fish and sediment samples were digested for total mercury determination by an open flask procedure developed at the National Institute for Minamata Disease (NIMD) in Japan by Akagi and Nishimura (1991) as shown in chart 3.1 below. In the procedure,

0.5g of homogenized fish, or 0.1g of sediment was weighed into 50 ml digestion flask and 1 ml H_2O, 2 ml HNO_3:$HClO_3$ (1:1) and 5 ml H_2SO4 were added in turns. The mixture was then heated at a temperature of 200^{0}C for 30 minutes. The sample solution was then cooled and diluted to 50 ml with double distilled water.

Solutions of 25 and 50 µl of 1 µg ml^{-1} standard Hg solution were also subjected to the same digestion procedure. The concentrations of the standard solution digest obtained were 25 and 50 ng.

Sample (0.5g fish or 0.1g sediment in 50ml digestion flask)

↓ H_2O, 1ml

↓ HNO_3: $HClO_4$ (1:1), 2ml

↓ H_2SO_4, 5ml

↓ Heat at 200^0 C for 30min

Digested Sample

↓ Cool to room temperature

↓ H_2O,

↓ Make up to 50 ml

Sample Solution, 5ml

↓ 10% $SnCl_2$, 0.5ml

CVAAS (Analyzer)

Chart 3.1 Analytical procedure for total mercury determination in fish and sediment samples

3.6 Extraction Procedure for Water

Extraction technique used for the water sample was developed at the National Institute for Minamata disease (NIMD) in Japan by Akagi and Nishimura (1991). In the procedure, one liter (1L) of water sample was transferred into 1.5L capacity separatory funnel and 10 ml of 10M H_2SO_4 and 5 ml of 0.5% $KMnO_4$ solution were added, mixed

by shaking and was allowed to stand for 5 min. The solution was neutralized with 20 ml of 10M NaOH, and 5 ml of 10% $NH_2OH.HCl$ solution was added and shaken. The mixture was allowed to stand for 30 minutes and 5 ml of 10% EDTA solution was added and mixed by shaking. Precise addition of 10 ml of purified 0.01% dithizone-toluene was made followed by vigorous shaking for 1 min. to extract mercury from the sample. The solution was allowed to stand for 1 hr, avoiding direct sunlight. This led to the formation of an aqueous phase and organic phase. The aqueous phase (lower phase) was discarded and the organic phase was transferred into a 10ml conical centrifuge tube fitted with a glass stopper and centrifuged at 1200 rpm for 3 min with the glass stopper in place. When an emulsion was formed, 0.5 g of anhydrous sodium sulfate was added and the mixture shaken, followed by centrifugation to separate the phases. Exactly 7ml of the organic phase was transferred into a 50 ml volumetric digestion flask. The solution was then evaporated to dryness on a water bath at 60^0 C with a rotary evaporator. The residue was then subjected to wet digestion with the addition of 1 ml of H_2O, 2 ml of concentrated HNO_3:$HClO_4$ (1:1) and 5 ml of concentrated of H_2SO_4 and heated on a Hot plate at 200^oC for 30 min. The sample solution was allowed to cool and distilled water was added to make it up to a volume of 50 ml. Exactly 5 ml of the test solution was injected into a mercury analyzer and the response was recorded in the form of a peak.

Water Sample, 1L (1.5L capacity separatory funnel)

↓ Add 10 ml of 10M H_2SO_4 and mix

↓ Add 5 ml of 0.5% $KMnO_4$ solution and mix.

↓ Let stand for 5 min.

↓ Add 20ml of 10M NaOH and mix to neutralize

↓ Add 5ml of 10% $NH_2OH.HCl$ solution, mix, and allow standing for 20 min.

↓ Add 5ml of 10% EDTA solution and mix

↓ Add 10ml of purified 0.01% dithizone-toluene and vigorously shake for 1 min

↓ Allow To stand for at least 1 hr.

Organic phase | Aqueous phase

↓ (When an emulsion is formed, add 0.5g of anhydrous $NaSO_4$ and shake.)

↓ Centrifuge at 1,200 rpm for 3 min.

Organic phase, 7ml (sample digestion flask)

↓ Evaporate to dryness

Residue

↓ Distilled water, 1 ml

↓ HNO_3:$HClO_4$ (1:1), 2ml

↓ H_2SO_4, 5ml

↓ Heat at 200°C for 30 min.

Digested sample

↓ Allow to cool

↓ Make up to 50ml with distilled water

Test solution, a fixed volume (5 ml)

↓ 10% $SnCl_2$ solution, 0.5 ml

CV AAS

Chart 3.2 Analytical procedure for determining total mercury in water

3.7 Determination of mercury

Determination of mercury in all the digests was carried out by Cold Vapour Atomic Absorption Spectrophotometry using an Automatic Mercury Analyzer model HG-5000 (Sanso Seisakusho co., Ltd, Japan) developed at National Institute for Minamata Disease (NIMD). The analyzer is an instrument designed specifically for the measurement of mercury using the cold vapour technique. It makes use of the batch mercury cold vapour generating system. The analyzer consists of an air circulation pump, a reaction vessel, $SnCl_2$ dispenser, an acidic gas trap and a four-way stop-cock with tygon tubes to which is attached a ball valve. The operations of the ball valve and the air circulation pump are controlled by a microprocessor. A schematic diagram of the system is shown in Fig.3.2. During the determination, a known volume of the sample solution normally 5 ml is introduced into the reaction vessel using a micropipette (5ml). The reaction vessel is immediately stoppered tightly and 0.5 ml of 10% (w/v) $SnCl_2.2H_2O$ in 1M HCl is added from a dispenser for the reduction reaction ($Hg^{2+} + Sn^{2+} \longrightarrow Hg^0 + Sn^{4+}$). During this time, air is circulated through the four-way stopcock to allow the mercury vapour to come to equilibrium and the acidic gases produced by the reaction also swept into the sodium hydroxide solution. After 30 seconds the four-way stopcock is rotated through 90^0 and the mercury vapour is swept into the absorption cell. Response is recorded on the strip chart recorder as very sharp peaks. Peaks heights were used for computations. Standards used for calibration of the analyzer included solutions containing 25 and 50 ng Hg.

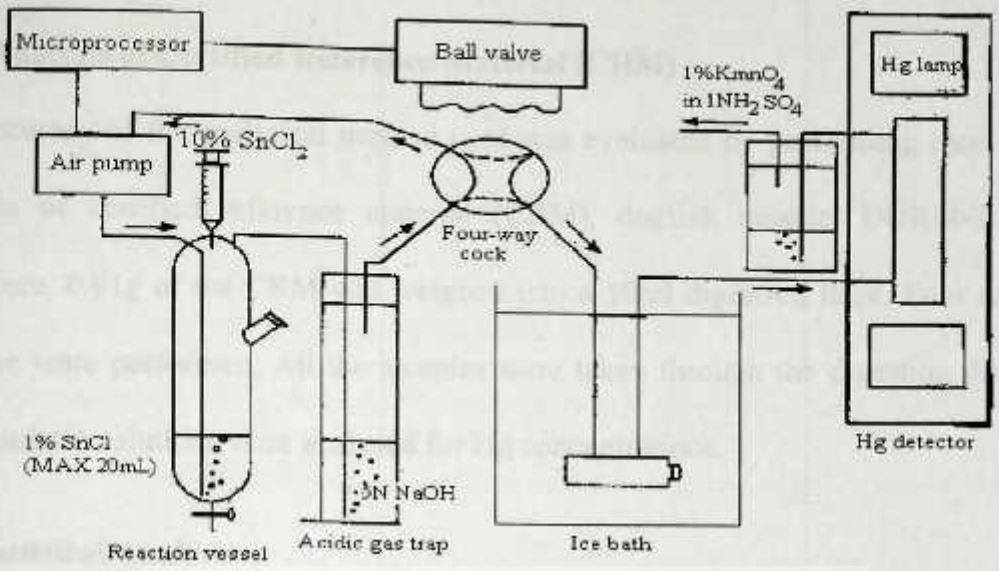


Fig. 3.2 Apparatus for mercury analyses by Cold Vapour Atomic Absorption Spectrophotometry

3.8 Recovery of Mercury

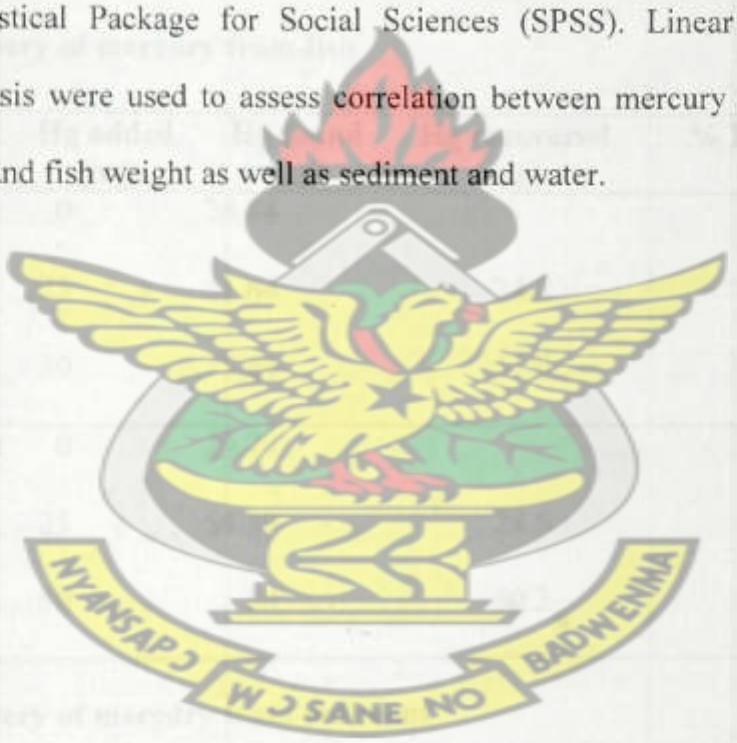
Recovery of mercury from fish was determined by adding increasing amounts of mercury chloride solution to known weights of two different fish species namely *Brycinus nurse* and *Brycinus imberi*. The first flask contained only 0.5g of each sample, the second flask contained 0.5g of each sample and 25ng Hg, and the third flask contained 0.5g of each sample and 50ng Hg. They were all taken through the digestion procedure. Similarly, recovery of mercury from sediment was determined also in the same manner using one sediment sample. Recovery of mercury from water was determined by adding increasing amounts of mercury chloride solution to distilled water which was taken through the extraction procedure. The resulting solutions were analysed for mercury concentration and the results obtained are reported in Table 4.1 to 4.3.

3.8.1 Analysis of Certified Reference Material (CRM)

The accuracy of the analytical method used was evaluated by performing recovery and analysis of certified reference material (CRM), dogfish muscle, DORM-2. In the procedure, 0.01g of the CRM was weighed into a 50ml digestion flask. Four replicates of these were performed. All the samples were taken through the digestion procedure. The resultant solutions were analysed for Hg concentrations.

3.9 Statistical Analysis

The data obtained in this study were subjected to statistical analysis using Microsoft Excel and Statistical Package for Social Sciences (SPSS). Linear regression and correlation analysis were used to assess correlation between mercury concentration in fish, fish length and fish weight as well as sediment and water.



4. RESULTS AND DISCUSSION

The accuracy and precision of total mercury determinations was determined by carrying out analysis of certified reference material and recoveries. The validity of the procedure has been proved by the agreement between the determined (4425 - 4640 ng/g) and the certified (4150 - 4790 ng/g) concentration of total Hg in dogfish muscle (DORM-2) reference material. Recoveries were 85% to 104.4%. The recovery results are reported in tables 4.1 to 4.3.

Table 4.1 Recovery of mercury from fish

Sample	Hg added (ng)	Hg found (ng)	Hg recovered (ng)	% Recovered
Brycinus nurse	0	26.84	-	-
	25	51.64	24.8	99.2
	50	77.84	51.0	102.0
Brycinus imberi	0	29.79	-	-
	25	54.39	24.6	98.4
	50	80	50.2	100.4

Table 4.2 Recovery of mercury from sediment

Sample	Hg added (ng)	Hg found (ng)	Hg recovered (ng)	% Recovered
October Sampling Point 2	0	166.75	-	-
	25	192.85	26.1	104.4
	50	216.45	49.7	99.4

Table 4.3 Recovery of mercury from distilled water

Sample	Hg added (ng)	Hg found (ng)	Hg recovered (ng)	% Recovered
Distilled Water	0	12.5	-	-
	25	36.5	24	96
	50	55	42.5	85

4.1 Total Mercury Concentrations in Fish, Sediments and Water

Total mercury (Hg) concentrations were determined in fish muscle tissues, sediments and water from the Black Volta River Basin in Ghana at two different sites namely, Bator and Bui Dam site. In all, a total of one hundred and eighty-five fish samples covering twenty-one species were collected and analysed for total mercury. All the fish species analysed for total Hg are consumed by humans.

Samples covering fifteen species namely, *Lates niloticus* (n=10), *Schilbe mystus* (n=20), *Alestes dentex* (n=3), *Brycinus nurse* (n=50), *Chrysichthys auratus* (n=8), *Mormyrus sp* (n=2), *Labeo coubie* (n=5), *Hydrocynus sp* (n=23), *Synodontis sp* (n=4), *Synodontis oceillifer* (n=4), *Mormyrus maropthalmus* (n=4), *Irvinea voltae* (n=2), *Barbus sp* (n=1), *Marcusenius abadii* (n=1), *Hepsetus odoe* (n=1), sixty (60) sediment and six (6) water samples were collected from Bator and analysed for total Hg. From Dam site, samples covering seven (7) fish species namely, *Schilbe mystus* (n=16), *Schilbe mandibularies* (n=20), *Hydrocynus forkali* (n=4), *Alestes imberi* (n=4), *Bagrus sp* (n=1), *Heterobrunchus bidorsalis* (n=1), *Clarias sp* (n=1), sixty (60) sediments and six (6) water samples were collected and analysed for total Hg.

Summary of results of Hg concentrations, fresh weight and total length of fish from the two sites are presented in Table 4.4. Mercury concentrations in sediments and water are presented in Tables 4.5 and 4.6 respectively. Total Hg concentrations (ng/g wet weight) in the edible muscle of fish from Bator and Dam site ranged from 14.28 to 208.54 and from 35.02 to 345.89 respectively. All the samples studied showed mercury concentrations below the Word Health Organization (WHO/FAO) limit of 500 ng/g wet weight. Total mercury concentration in fish depends on the fish species and the concentrations also varied with factors such as total length of fish and fresh weight of fish. There was a significant variation between mercury concentrations, fish length and fish weight in this study. Although growth rate data of fish from the studied areas are not available, variations suggest that all the fish species are not growing at the same rate.

Table 4.4 Total Hg concentrations (ng/g) in fish muscle tissues from Bator and Dam site

Species Name	Sampling Site	Sample Size (n)	Fresh weight Range (g)	Mean Weight (g)	Hg concentration Range (ng/g)	Mean Hg Concentration (ng/g) ± s. d.
Lates niloticus	Bator	10	18.1-134.8	53.7	59.25-181.29	127.03±42.49
Schilbe mystus	Bator	20	10.8-32.7	19.9	57.49-208.19	137.76±49.11
	Dam site	16	24.0-50.0	33.5	49.94-279.44	131.31±64.08
Alestes dentex	Bator	3	21.0- 47.1	31.9	19.97-27.04	22.90±3.68
Brycinus nurse	Bator	50	16.0- 202.9	70.0	14.39-39.68	27.88±7.67
Chrysichthys	Bator	8	26.0- 58.0	40.3	68.38-121.39	94.82±21.02
Auratus						
Hydrocynus Sp	Bator	23	18.9-242.1	88.8	18.36-208.54	88.01±58.53
Labeo coubie	Bator	5	25.2-202.9	112.8	14.28-24.93	18.58±4.28
Synodontis	Bator	4	8.8-76.6	33.7	32.01-68.43	52.34±15.32
Oceillifer						
Mormyrus	Bator	4	9.5-14.6	11.8	45.5-62.04	56.07±7.45
Macrophthalmus						
Synodontis Sp	Bator	4	8.9-20.5	15.2	42.41-86.55	62.23±18.72

Species Name	Sampling Site	Sample Size (n)	Fresh weight Range (g)	Mean Weight (g)	Hg concentration Range (ng/g)	Mean Hg Concentration (ng/g)
Irvinea voltae	Bator	2	23.9-97.0	60.4	104.86-105.89	105.38±0.73
Mormyrus Sp	Bator	2	14.2-58.0	36.3	69.78-99.57	84.68±21.07
Barbus Sp	Bator	1	43.2	43.2	28.55	28.55
Marcusenius abadii	Bator	1	18.7	18.7	68.73	68.73
Hepsetus odoe	Bator	1	119.5	119.5	94.36	94.36
Schilbe Mandibularies	Dam site	20	17.4-26.9	21.3	45.00-226.37	119.82±48.11
Hydrocynus Forkali	Dam site	4	44.0-81.4	64.5	119.48-345.89	198.58±101.8
Brycinus Imberi	Dam site	4	76.0-110.4	95.6	35.02-140.59	70.84±48.62
Clarias sp	Dam site	1	454.0	454.0	66.80	66.80
Bagrus Sp	Dam site	1	866.3	866.3	109.61	109.61
Heterobrunchus bidorsalis	Dam site	1	87.4	87.4	191.04	191.04

s.d= standard deviation

Table 4.5 Mean (n=10) Hg concentration in sediment from Bator and Dam site

Month	Sampling site	Mean Hg Concentration (ng/g)
October	Bator	43.41
	Dam site	51.06
November	Bator	49.80
	Dam site	58.77
December	Bator	50.05
	Dam site	48.86
January	Bator	50.49
	Dam site	56.32
February	Bator	105.69
	Dam site	45.13
March	Bator	114.96
	Dam site	49.75

Table 4.6 Summary of mean Hg conc. in sediment and water at Bator and Dam site

Sample Tested	Sampling site	Mean Hg concentration ± s.d
Sediment	Bator	69.07 ± 32.20ng/g
	Dam site	51.65 ± 5.03ng/g
Water	Bator	0.06 ± 0.03ng/g
	Dam site	0.04 ± 0.02ng/g

s.d= standard deviation

Detailed results obtained including graphs showing relationships between total mercury concentration, total length, fresh weight in fish as well as mercury concentration in sediment and water are presented in the Appendix. The level of mercury in fish ranged between 14.28 to 208.54 ng/g wet weight as shown in Table 4.4. The highest level of 208.54 ng/g wet weight was found in *Hydrocynus sp* and the lowest level of 14.28 ng/g wet weight was found in *Labeo coubie*. Mercury levels obtained showed that within the same aquatic environment, fish of the same species have different levels of mercury. This could be due to size (age) since it is an important variable that determines level of mercury accumulation in fish according to Huckabee *et al.*, (1979).

The range of mercury concentration (ng/g wet weight) in the muscle tissue ranged from 59.25 to 181.29 (mean =127.03±42.49) for *Lates niloticus*, from 57.49 to 208.19 (mean = 137.76±49.11) for *Schilbe mystus*, from 19.97 to 27.04 (mean = 22.90±3.68) for *Alestes dentex*, from 14.39 to 39.68 (mean=27.88±7.67) for *Brycinus nurse*, from 68.38 to 121.39 (mean= 94.82) for *Chrysichthys auratus*, from 69.78 to 99.57 (mean = 84.68±21.07) for *Mormyrus sp*, from 14.28 to 24.93 (mean = 18.58±4.28) for *Labeo coubie*, from 18.36 to 208.54 (mean = 88.01±58.53) for *Hydrocynus sp*, from 32.01 to 68.43 (mean= 52.34±15.32) for *Synodontis oceillifer*, from 42.41 to 86.55 (mean= 61.23±18.72) for *Synodontis sp*, from 45.50 to 62.04 (mean = 56.07±7.45) for *Mormyrus macrophthalmus*, and from 104.86 to 105.89 (mean = 105.38±0.73) for *Irvinea voltae*.

A study conducted by Kidd *et al.*, (1995) on fish from Lango Manso, a reservoir in Brazil showed that mean mercury concentrations vary widely between species, which can be explained by trophic positions in the food web. Fish from top trophic levels

(carnivores) in a food chain usually have higher mercury concentrations than fish from lower trophic levels (herbivores) as a result of biomagnification. A similar trend was observed in some cases in this study. Hg concentrations increased with trophic levels. For example from Bator (Table 4.4) *Labeo coubie* species recorded Hg concentration range of 14.28 to 24.93 ng/g wet weight (mean = 18.58±4.28) and fresh weight range of 25.2 to 202.9g (mean =112.2±81.4) whereas *Lates niloticus* at a higher trophic level recorded Hg concentration range of 59.25 to 181.29 ng/g wet weight (mean = 127.03±42.49) and fresh weight range of 18.1 to 134.8g (mean=49.1±45.9). There was also, significant correlation between mercury concentration, fresh weight (r^2=0.505) and total length (r^2=0.604) for *Lates niloticus* (fig.4.1 and 4.2) whilst *Labeo coubie* (fig. 4.13 and 4.14) showed poor correlation between mercury concentration, fresh weight (r^2=0.095) and total length (r^2=0.176). This observation can be due to the feeding habits of the fish species as *Lates niloticus* is a carnivorous fish which feeds on other fishes whereas *Labeo coubie*, a detritivore feeds on vegetable debris. Work done by Lange *et al*., (1994) also showed that mercury concentration varies with fresh weight of fish and total length. Good correlation between mercury concentration and total length and fresh weight of fish are normally observed among carnivorous species whereas poor correlations are observed among herbivorous species. Among the fish species studied at Bator, there was a poor correlation between Hg concentration and total length of fish for *Schilbe mystus* (r^2 = 0.266, insectivore), *Alestes dentex* (r^2 = 0.402, pelagic omnivore), *Brycinus nurse* (r^2 = 0.110, pelagic omnivore), *Chrysichthys auratus* (r^2 = 0.113, benthic omnivore), *Hydrocynus sp* (r^2=0.344, piscivore), *Labeo coubie* (r^2 = 0.176, detritivore), *Mormyrus macrophthalmus* (r^2 = 0.084, bottom invertebrates). There was also a poor correlation between mercury concentration and total weight of fish for *Schilbe mystus* (r^2

= 0.049), *Alestes dentex* (r^2 = 0.399), *Brycinus nurse* (r^2 = 0.096), *Chrysichthys auratus* (r^2 = 0.045), *Hydrocynus sp* (r^2 = 0.113), *Labeo coubie* (r^2 = 0.095) and *Mormyrus macrophlthlmus* (r^2 = 0.208), whereas good correlation between Hg concentration and fresh weight of fish was observed for *Lates niloticus* (r^2 = 0.505) and an inverse correlation between Hg concentration and fresh weight of fish were observed for *Synodontis oceillifer* (r^2 = 0.836) and slight correlation for *Synodontis sp* (r^2 = 0.447). Similarly, there was also a good correlation between Hg concentration and total length of fish for *Lates niloticus* (r^2 = 0.604) and an inverse correlation between Hg concentrations and total length of fish for *Synodontis oceillifer* (r^2 = 0.911) and *Synodontis sp* (r^2 = 0.682). The inverse correlation between Hg concentrations, fresh weight and total length of fish observed for *Synodontis sp* could be due to changes in the metabolic rate of the fish as some studies have shown that smaller fish have higher uptake rates of mercury due to higher metabolic rate than larger individuals of the same species (Huckabee *et al.*, 1979).

Thompson (1985) observed lack of correlation between total length and mercury concentration for several fish species distributed along the Tasmanian continental shelf. Results of this study agree with this finding. Poor correlation exist between mercury concentration and total length of fish for *Schilbe mystus* (r^2=0.355), *Schilbe mandibularies* (r^2=0.049) and *Brycinus imberi* (r^2=0.233) whereas good correlation was observed between mercury concentration and total length of fish for *Hydrocynus forkali* (r^2=0.545). For instance, *Hydrocynus forkali* with length of 24.2cm and weight of 78.8g had Hg concentration of 345.89 ng/g while *Schilbe mandibularies* with length and weight of 14.2cm and 21.3g respectively had Hg concentration of 77.28 ng/g all from

the same sampling site. This finding agrees with that of Lathrop *et al.*, (1991) which reports that length and age of fish species have been shown to be important factors determining mercury concentration in fish. Apart from diet and trophic level, differences in longevity, growth rate and other physiological and ecological factors can also lead to differences in mercury concentrations between species (Huckabee *et al.*, 1979).

Mean mercury levels in sediment and water from Bator are 69.07±32.20 ng/g and 0.06±0.03 ng/L respectively (Table4.6). The high deviation (±32.2 ng/g) of mercury in sediment at Bator may be due to differences in organic matter content in the sediment as some studies suggest that, mercury has high affinity for particles and organic matter (Benoit *et al.*, 1998; Mason, 2001). However, such conclusion cannot be drawn since information about organic matter is lacking in this study.

In general, mercury concentration (ng/g wet weight) ranged from 35.02 to 345.89 for fish species sampled from Dam site. *Hydrocynus forkali* recorded the highest level of mercury of 345.89 whilst the lowest recorded Hg concentration of 35.02 was by *Brycinus imberi*. The highest level of mercury 345.89 ng/g wet weight recorded is less than the standard 500 ng/g set by the World Health organization (WHO) and hence does not pose health hazard. Hg concentration in the range of 49.94 to 279.44 (mean = 131.31±64.08) was obtained for *Schilbe mystus*, 45.0 to 226.37 (mean=119.82±48.11) for *Schilbe mandibularies*, 109.61 for *Bagrus sp*, 191.04 for *Heterobrunchus bidorsalis* and 66.80 for *Clarias sp*.

Although it has been reported that mercury concentrations in bed sediments are not necessarily correlated with concentration in fish tissue (Rose *et al.*, 1999) other studies carried out on mercury in fish showed that there is positive correlation between Hg in fish and sediment (Hunter *et al.*, 1987). In this study, correlation between Hg concentration in different fish species and sediments at Bator and Dam Site was determined and reported as correlation coefficient (r^2) in Table 4.7. *Synodontis Oceilifer* and *Hydrocynus sp* are the only species that showed positive correlation between their Hg concentration and river sediment at Bator. The r^2 value for *Synodontis Oceilifer* and *Hydrocynus sp* at Bator were 0.8237 and 0.6231. The rest of the fish species showed poor correlation between Hg concentration in fish and sediment at the two sites (Table 4.7). However, the correlation between Hg concentration in *Lates niloticus* from Bator, *Hydrocynus forkali* and *Brycinus imberi* from Dam site and sediment was relatively week (r^2=0.3188. 0.3258 and 0.2886 respectively). The poor correlation between Hg concentrations and river sediments in most of the fish species could be attributed to the fact that Hg content in the fish muscle is not related only to the Hg content in the sediments, but also to the diet composition of the fish and to the other chemical and biological characteristics of the aquatic ecosystem (Huckabee *et al.*, 1979) and this may be similar to that of the Black Volta river basin at Bui.

Table 4.7 Correlation (r2) between Hg concentration (ng) in fish and river sediments

Sampling Sites	Fish species and R^2 values												
	Lates niloticus	*Schilbe mystus*	*Brycinus nurse*	*Chrysichthys auratus*	*Hydrocynus sp*	*Labeo coubie*	*Synodontis ocellifer*	*Synodontis sp*	*Mormyrus macropthalmus*	*Alestes dentex*	*Schilbe mandibularies*	*Hydrocynus forkali*	*Brycinus imberi*
Bator	0.3188	0.0047	0.1795	0.0842	0.6231	0.0673	0.8237	0.0046	0.0103	0.226	-	-	-
Dam Site	-	0.0093	-	-	-	-	-	-	-	-	0.0429	0.3258	0.2886

The dash (-) suggest that such species were not obtained at those sampling sites.

Sorensen *et al*., (1990) found that Hg levels in northern pike from Minnesota lakes were correlated with Hg in water. The correlation between Hg concentration in different fish species and water at Bator and Dam Site was also determined and reported in Table 4.8. *Synodontis Oceilifer* with correlation coefficient of (0.6846) at Bator is the only species that showed consistency with the findings above. The rest of the fish species showed poor correlation between Hg concentration in fish and water at the various sites (Table 4.8). The correlation between Hg concentration in *Lates niloticus* from Bator. *Hydrocynus forkali* and *Brycinus imberi* from Dam site and water was relatively week with correlation values of r^2=0.3683, 0.3833 and 0.3148 respectively.

Table 4.8. Correlation (r2) between Hg concentration (ng) in fish and water

Sampling Sites	Fish species and R^2 values												
	Lates niloticus	*Schilbe mystus*	*Brycinus nurse*	*Chrysichthys auratus*	*Hydrocynus sp*	*Labeo coubie*	*Synodontis ocellifer*	*Synodontis sp*	*Mormyrus macrophalmus*	*Alestes dentex*	*Schilbe mandibularies*	*Hydrocynus forkali*	*Brycinus imberi*
Bator	0.3683	0.0237	0.0011	0.0676	0.0663	0.2706	0.6846	0.2008	0.0265	0.0469	-	-	-
Dam Site	-	0.0604	-	-	-	-	-	-	-	-	0.0765	0.3833	0.3148

The dash (-) suggest that such species were not obtained at those sampling sites.

The correlation between Hg concentration in sediment and water was determined at the two sampling sites and reported in Table 4.9.

Table 4.9. Correlation (r^2) between Hg concentration (ng/g) in sediment and water

Sampling Site	R^2 Values
Bator	0.003
Dam Site	0.365

No correlation was observed between the total Hg in the sediment and the water at the respective sites. This however depicts that the total Hg concentrations in the sediment and water are independent of each other. This finding agrees with the study of Harris and Bodaly, (1998) who reported that concentrations of total Hg or methylmercury (MeHg) in surface waters often do not correlate well with the Hg content of freshwater biota.

The low Hg concentrations in sediments from both sites as well as the small difference in Hg levels between the sites indicate that there has not been any local contaminating source in the studied area. This agrees with work done by Gilmour and Henry (1991) on non-contaminated sediments where low Hg levels were found. USEPA, (1997) reported that total Hg levels in lakes and streams are typically well under 20 ng/L, however, elevated levels may be found in lakes and streams thought to be impacted by anthropogenic Hg sources. This is consistent with this study, where at all the sampling

sites Hg concentration range of 0.04±0.02 to 0.06±0.03 ng/L obtained were below 20 ng/L and even the background level of 0.1µg/L.

An attempt was made to compare mercury levels in fish, sediment and water at the two sampling sites but same fish species were not found at both sampling sites. *Schilbe mystus* which is the only species that was found at both sampling sites had an average Hg level of 137.76ng/g at Bator and an average Hg level of 131.31ng/g from Dam site. When the results were subjected to t -Test, there was significant difference between the results ($p \leq 0.05$) at the 95% confidence level. This suggests that, factors responsible for the methylation of mercury are more pronounced at Bator than Dam Site. In addition, mercury level in water from Bator was 0.06±0.03ng/L while that of Dam site was 0.04±0.02ng/L. Similarly, when the results were again subjected to t-Test, there was significant difference between the results ($p \leq 0.05$) at the 95% confidence level. Mean Hg level in sediment from Bator was found to be 69.07±32.20 ng/g whilst that of Dam site was 51.65±5.03ng/g. There was also significant difference between the results when they were subjected to t -Test ($p \leq 0.05$) at the 95% confidence level.

Mercury content in fish is considered to be a good indicator of human exposure to organic or methylmercury. Hg enters the aquatic ecosystem as inorganic mercury which may be converted to methylmercury by microorganisms in the sediment and water and methylmercury may enter the aquatic food chain and work its way up to the larger fish, which eventually may be consumed by humans and other animals. Humans' health concerns arise when fish and wildlife from these ecosystems are consumed by humans since fish accumulate high concentrations of the most toxic form of mercury i. e.

methylmercury (Uchida *et al.*, 1961) which can affect the nervous system, cause blurred vision, coma and ultimately death (Harada, 1995; Takeuchi and Eto, 1999). Hg in fish appears to be predominantly in the form of methylmercury and this has been confirmed in many publications (Al-Majeed and Preston, 2000; Bloom, 1992). Therefore, diet consisting particularly of fish, could be the main source of human exposure to methylmercury.

The relatively low mercury levels found in the fish species studied from the Black Volta River could be attributed to the fact that, organic matter, pH, seasonal changes, regional variations and hydrologic conditions which are thought to be the most significant factors that control accumulation of mercury in fresh water fish as reported by Lindquist (1991) might be unfavourable. Atmospheric deposition of Hg which is another factor that increases Hg loads in freshwaters may be low. Agricultural activities that employ the use of mercurial compounds could be minimal. The evaporation of elemental mercury as a result of microbial activities in the river could also represent a significant pathway for the small amount of the metal in the aquatic system of the Black Volta River. However, these conditions could be favorable for methylation of Hg after the construction of the dam as has been reported elsewhere (Jackson, 1988; Stocks and Wren, 1987). For example, a study conducted in Canada revealed that, after the impoundment of the 13,670km^2 La Grande hydroelectric complex, level of Hg in fish increased 3 – 6 folds (Tremblay *et al.*, 1998; Schetagne *et al.*, 2000). At the same time, concentrations of organic carbon in the water increased and oxygen concentrations decreased, indicating increased decomposition and anoxia as contributing to the increased Hg concentrations in fish. Again, a similar work conducted (Lars *et al.*, 2006) at Rio Manso hydroelectric

power plant in western Brazil indicated that predatory fish had total Hg concentrations ranging between 70 and 210 ng g^{-1} fresh weight (fw) seven (7) years before flooding and between 72 and 755 ng g^{-1} f w during flooding, but increased to between 216 and 938 ng g^{-1} fw in the piscivorous and carnivorous species *Pseudoplatystoma fasciatum*, (cachara), and *Salminus brasiliensis*, (dourado) respectively, 3 years after flooding. Mercury concentration recorded in 230 fish samples from Amazona hydroelectric reservoir recorded levels higher than 0.5mg Hg/kg fresh weight (Rosenberg *et al.*, 1997). Rondón and Pérez, (1999) also reported increased bioaccumulation of Hg in fish in tropical reservoirs as a result of impoundments. Rondón and Peréz (1999) then concluded that the complexity of trophic relationships in tropical aquatic ecosystems is higher than in those observed in cold and temperate regions, whereas other studies suggest that bioaccumulation patterns are surprisingly similar (Meili *et al.*, 1996a and 1996b). Furthermore, an attempt to open commercial fisheries in Notigi Reservoir was immediately ceased when quiet high Hg concentrations were found in fish from this reservoir after impoundments (Bodaly *et al.*, 1984).

It is therefore important to find Hg levels now and after the construction of the dam to corroborate the suggestion that Hg levels in fish increase after dam construction.

CHAPTER FIVE

5. CONCLUSIONS AND RECOMMENDATIONS

5.1 CONCLUSIONS.

From the analysis carried out, the following conclusions may be deduced from the results obtained.

- This study reports mercury levels in fish species from Bator and Bui Dam site, all along the Black Volta River at Bui. The concentration of total mercury in the edible muscle tissue of all the fish analysed from Bator and Dam site ranged from 14.28 to 208.54 ng/g and 35.02 to 345.89 ng/g wet weight respectively.
- The study also reported Hg level in sediments from Bator and Bui Dam site. The range of Hg concentrations in the river sediments were 43.41 to 114.96 (mean= 69.07) and 45.13 to 58.77 ng/g (mean=51.65) respectively. Average Hg level in water from the two sites was 0.04 ng/L and 0.06 ng/L. Hg concentrations in fish are below the World Health organization (WHO) limit of 500ng/g. Hg concentrations in sediment are below the International Atomic Energy Agency (IAEA) limit of 810ng/g and Hg concentrations in water are below the WHO maximum contamination level of 1000 ng/L.
- The results obtained showed that fish species from the Black Volta River are unlikely to constitute a significant exposure of the public to Hg through consumption of fish.
- The low concentrations of mercury in sediments, fish species and water obtained in this study suggest a relatively clean aquatic environment.

5.2 RECOMMENDATIONS

This recommendation is made as a result of the outcome of this research.

- The same species of fishes tested in this study should be tested over a longer period after the impoundment to corroborate the suggestion that Hg levels increase after dam construction.

REFERENCES

Agorku, S. E., Voegborlo, R. B. and Adimado, A. A. (2008) Total mercury levels in nine species of freshwater fish from two hydroelectric reservoirs and a crater lake in Ghana. Environ Monit Assess, 153, 383-389.

Akagi, H. and Ikingura, J. R. (2003) Total mercury and methylmercury levels in fish from hydroelectric reservoirs in Tanzania. Sci Total Environ, 304, 355 – 68.

Akagi, H. and Nishimura, H. (1991) Speciation of mercury in the environment. In: T. Suzuki, N. Imura and T. W. Clarkson, (Eds), Avances in Mercury Toxicology, Plenun Press USA, 53-76.

Alfthan, G., Jarvinen, O., Pikkaraine N, J. and Verta, M. (1983) Mercury and artificial lakes in northern Finland. Possible ecological and health consequences. Nord. Counc. Arct. Med. Res. Rep., 35, 77-81.

Al-Majeed, N. B. and Preston, M. R. (2000) An assessment of the total and methylmercury content of zooplankton and fish tissue collected from Kuwait territorial waters. Marine Pollution Bulletin, 40, 298-307.

AMAP (Arctic Monitoring and Assessment Programme) (2000) Report on issues of concern: Updated information on human health, persistent organic pollutants, radioactivity, and mercury in the Arctic. AMAP Report 2000.4, found on www.amap.no, July 2001.

Amyot, M., Gill, G. A. and Morel, F. M. M. (1997) Production and loss of dissolved gaseous mercury in coastal seawater. Environmental Science and Technology, 31, 3606–3611.

Bailey, E. A., Gray, J. E. and Hines, M. E. (2001) Mercury transformations in soils near mercury mines in Alaska, Materials and Geoenvironment, 48, 1, 212-218.

Bailey, E. H., Clark, A. L. and Smith, R. M. (1973) Mercury. USA Geological Survey Prof. Pap., 821, 410-414.

Benoit, J. M., Gilmour, C. C., Mason, R. P., Riedel, G. S. and Riedel, G. F. (1998) Behavior of mercury in the Patuxent estuary. Biogeochem, 40, 249-265.

Bisogni, J. J. and Lawrence, A. W. (1973) Kinetics of microbially mediated methylation of mercury in aerobic and anaerobic aquatic environments. Ithaca, NY, Cornell University Water Resources and Marine Science Center (Technical Report No. 63).

Bloom, N. (1992) On the chemical form of mercury in edible fish and marine invertebrate tissue. Canadian Journal of Fisheries and Aquatic Science, 49, 1010-1017.

Bodaly, R. A. and Rosenberg, D. M. (1990) Retrospective Analysis of Predictions and Actual Impacts for the Churchill- Nelson Hydroelectric Development, Northern Manitoba, In: Joules in Water- Managing the Effects of Hydroelectric Development. Delisle C. E. and Bouchard M. A. (ed.), 221-242. Canadian Society of Environmental Biologists.

Bodaly, R. A., Hecky, R. E. and Fudge, R. J. P. (1984) Increases in fish mercury levels in lakes flooded by Churchill River diversion, Northern Manitoba. Canadian Journal of Fisheries and Aquatic Sciences, 41, 682 – 691.

Bodaly, R. A., St Louis, V. L., Paterson, M. J., Fude, R. J. P., Hall, B. D., Rosenberg, D. M. and Rujdd, J. W. M. (1997) Bioaccumulation of mercury in the aquatic food chain in newly flood areas. Water, Air and Soil Pollution, 100, 13-24.

Burger, J. (1997) Ecological effects of biomonitoring for mercury in tropical ecosystems. Water, Air and Soil Pollution, 97, 265-272.

Cabana, G., Tremblay, A., Kalff, J. and Rasmussen, J. B. (1994) Pelagic Food Chain Structurein Ontario Lakes: A Determinant of Mercury Levels in Lake Trout (*Salvelinus namaycush*).Canadian Journal of Fisheries and Aquatic Sciences, 51, 381-389.

Canada-Manitoba (1987) Summary Report. Canada-Manitoba Agreement on the study and monitoring of mercury in the Churchill River Diversion, Winnipeg, Manitoba. Environment and Workplace Safety and Health; Hull, Québec, Environment Canada, 77.

Clarkson, T. W. (1992) Mercury: Major issues in Environmental Health, Environmental Health Perspective, 100, 31-38.

Coolbaugh, M. F., Gustin, M. S. and Rytuba, J. J. (2002) Annual emissions on mercury to the atmosphere from three natural source areas in Nevada and California. Environmental Geology, 42, 338-349.

Coquery, M., Cossa D. and Martin J. M., (2001) The Distribution of Dissolved and Particulate Mercury in Three Siberian Estuaries and Adjacent Arctic Coastal Waters, Water Air Soil Pollution, 2491-2496.

Costa, M. and Liss, P. S. (1999) Photoreduction of Mercury in Sea Water and its Possible Implications for Hg^0 Air-sea Fluxes. Marine Chemistry, 68, 87-95.

Dennis, I. F., Clair, T. A., Driscoll, C. T., Kamman, N.C., Chalmers, A., Shanley, J.B., Norton, S. A. and Kahl, S. (2005) Distribution patterns of mercury in lakes and rivers of northeastern North America. Ecotoxicology, 14, 113–123.

ERM (Environmental Resources Management) (2006) Baseline and Scoping Report-Environmental and Social Impact Assessment of the Bui Hydropower Project, 8 -30.

FAO/WHO (1991) Codex Alimentarius: Guideline Levels for Mercury in Fish (CAC/GL 7-1991). Adopted by the Commission at its Nineteenth Session in Rome 1-10 July 1991.

Fitzgerald, W. F. (1986) Cycling of Mercury between the Atmosphere and Oceans. In: Buat-Ménard, P., ed., The role of Air-sea Exchange in Geochemical Cycling. R. Reidel Publishing Company, 363-408.

Gilmour, C. C. and Henry, E. A. (1991) Mercury methylation in aquatic systems affected by acid deposition, Environmental Pollution, 71, 131–169.

Gilmour, L. C., Henry, E. and Mitchell, R. (1992) Sulfate stimulation of mercury methylation on freshwater sediments. Environ. Sci. Technol., 26, 2281-2287.

Gremillion, P. T., Cizdziel, J. V. and Norman R. C. (2004) Caudal fin mercury as a non-lethal predicor of fish-muscle mercury, Environmental Chemistry, 2(2), 96 – 99.

Harada, M. (1995) Minamata disease, methylmercury poisoning in Japan caused by environmental pollution. CRC Critical Reviews in Toxicology, 1995, 25, 1-24.

Harris, R. C. and Bodaly, R. A. (1998) Temperature, growth, and dietary effects on fish mercury dynamics in two Ontario lakes. Biogeochemistry, 40, 175–187.

Harris, R. C. and Snodgrass, W. J. (1993) Bioenergitic Simulations of Mercury Uptake and Retention in Walleye (*Stizostedion vitreum*) and Yellow Perch(*Perca flavescens*), Water Pollut. Res. J. Can., 28, 217-236.

Heyes, A. and Gilmour, C. C. (1999) The biogeochemical controls on mercury methylation across ecosystems. ASLO Aquatic Science Meeting, Santa Fe, NM.

http://www.epa.gov/mercury/exposure.htm (accessed 2008 February 20).

http://www.irn.org/programs/threeg/resettle.html (accessed 2008 March 12).

http://www.waterencyclopedia.com/Re-St/Reservoirs-Multipurpose.html (accessed 2008 April 10).

Huckabee, J. W., Elwood, J. W. and Hildebrand, S. G. (1979) Accumulation of Mercury in Freshwater biota. In Nriagu, J. O. (editor). The biogeochemistry of Mercury in the Environment. Elsevier /North –Holland Biomedical Press, Amsterdam, 277-302.

Hunter, D. B., Leatherland, J. F. and Stokes P. M. (1987) The effects of polychlorinated biphenyls and methylmercury, single adding combination on mink II. Reproduction and kit development, Arch. Environ. Contam Toxicol, 16, 449 – 454.

IRN (International Rivers Network), (2001).
http://www.irn.org/programs/safrica/index.php?id=others.html>.

Irukayama, K. and Tsubaki, T. (1977) Minamata Disease, Methylmercury Poisoning in Minamata and Niigata, Kodansha Ltd., Tokyo, and Elsevier Sci. Publ. Co., Amsterdam, 2 – 56.

Iverfeldt, Å. (1991) Occurrence and turnover of atmospheric mercury over the Nordic countries. Water, Air and Soil Pollution, 56, 251-265.

Jackson, T. A. (1988) Mercury Problem in Recently Formed Reservoirs of Northern Manitoba (Canada): Effects of Impoundment and other Factors on the Production of Methylmercury by Microorganisms in Sediments. Canadian Journal of Fisheries and Aquatic Science, 45, 97 – 121.

Jackson, T. A. (1991) Biological and environmental control of mercury accumulation by fish in lakes and reservoirs of Northern Manitoba, Canada. Canadian Journal of Fisheries and Aquatic Sciences, 48, 2449-2470.

Jensen, S. and Jernelov, A. (1967) Biosynthesis of methylmercury. Biocidinformation, 10, 4.

Jensen, S. and Jernelov, A. (1969) Biological methylation of mercury in aquatic organisms. Nature (London) 223, 753-754.

Jonasson, I. R. and Boyle, R. W. (1971) Geochemistry of mercury. Spatial Symposium on Mercury in Man's Environment, Environment Canada, Ottawa 15-16 February 1971, Ottawa, Royal Society of Canada, 5-21.

Kelly, C. A., Rudd, J.W. M. and Bodaly, R. A. (1997) Increases in fluxes of greenhouse gases and methylmercury following flooding of an experimental reservoir. Environ. Sci. Technol., 31, 1334 – 1344.

Kidd, K. A., Hesslein, R. H., Fudge, R. J. P. and Hallard, K. A. (1995) The influence of Trophic Level as Measured by δ^{15} N on mercury concentrations in Freshwater Organisms. Water, Air and Soil Pollution, 80, 1011-1015.

Koeman, J. H, Vande Ven, W. S. M, de Goejj, J. J. M., Tjioe, P. S., and Van Haaften, J. L. (1975) Mercury and Selenium in mammals and birds. Sci. Total Environ, 3, 279 – 287.

Krabbenhoft, D. P., Wiener, J. G., Brumbaugh, W. G., Olson, M. L., DeWild, J. F., Sabin, T. J. (1999) A national pilot study of mercury contamination of aquatic ecosystems along multiple gradients. Morganwalp DW, Buxton HT. (eds.) US Geological Survey Toxic Substances Hydrology Program, 147–160.

Landis, M. S., Vette, A. F. and Keeler, G. J. (2002) Atmospheric Deposition to Lake Michigan during the Lake Michigan Mass Balance Study. Environ. Sci. Technol. 36(21), 4518-4524.

Lange, T. R., Royals, H. E. and Connor, L. L. (1994) Mercury accumulation in largemouth bass (*Micropterus salmoides*) in a Florida Lake, Florida Game and Freshwater Fish Commission, 32, 727.

Lars, D. H., Janina, G., Magdalena, T., Anna, V., Henriette, W., Edinaldo de Castro e Silva, Markus, M. and Lázaro, J. O. (2006) Mercury cycling in contaminated tropical non- marine ecosystems, Journal of Environmental Management, 81, Issue 2, 155-166.

Lathrop, R. C., Rasmussen, P. W. and Knauer D. R. (1991) Mercury Concentrations in Walleyes from Wisconsin (USA) Lakes. Water, Air, and Soil Pollution, 56, 295-307.

Lindberg, S. E. (1999) The Role of Mercury Air/Surface Exchange Process in the Global Biogeochemical Cycle. Proceeding of 5th International conference on Mercury as Global Pollutant,'' Rio de Janeiro.

Lindberg, S. E. and Stratton, W. J. (1998) Atmospheric Mercury Speciation: Concentration and Behaviour of Reactive Gaseous Mercury in Ambient Air. Env. Sci and Techn, 32, 40 – 57.

Lindberg, S. E., Wallschlager, D., Prestbo, E. M., Bloom, N. S., Price, J. and Reinhart, D., (2001) Methylated mercury species in municipal waste landfill gas sampled in Florida, USA, and Atmospheric Environment, 35, 4011-4015.

Linguist, O. (1991) Mercury in Swedish Environment, Water, Air and Soil Pollution, 55 1-261.

Lodenius, M., Seppänen. A. and Herranen, M. (1983) Accumulation of mercury in fish and man from reservoirs in Northern inland. Water, Air and Soil Pollution, 19, 237 – 246.

Marvin-Dipasquale, M., Agee, J., McGowan, C., Oremland, R., Thomas, M., Krabbenhoft, D. and Gilmour, C. C. (2000) Methyl-mercury Degradation Pathways: A Comparison among Three Mercury Impacted Ecosystems, Environmental Science and Technology 2000, 34, 4908-4916.

Mason, R. P. (2001) The bioaccumulation of mercury, methylmercury and other toxic elements into pelagic and benthic organisms. In: M. C. Newman, M. H. Robert, and R. C. Hale [eds.], Coastal and Estuarine Risk Assessment, CRC/Lewis Publ., 127-149.

Mason, R. P. and Fitzgerald, W. F. (1996) Sources, sinks and biochemical cycling of mercury in the ocean. In: Baeyens, W., Ebinghaus, R. and Valiliev, O., (eds.): Global and regional mercury cycles: Sources, fluxes and mass balances. NATO ASI Series, 2. Environment, 21. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Mason, R. P. and Fitzgerald, W. F. (1997) Biogeochemical cycling of mercury in the marine environment. In: Sigel, A., and Sigel, H., Metal ions in biological systems. Marcel Dekker, Inc. 34, 53-111.

Mason, R. P., Lawson, N. M., Lawrence, A. L., Lee, J. G., Leaner, J. J. and Sheu, G. R. (1999) Mercury in the Chesapeake Bay. Mar. Chem., 65, 77-96.

Meech, J. A., Veiga, M. M., Tromans, D. (1998) Reactivity of Mercury from Gold Mining Activities in Darkwater Ecosystems. Ambio, 27, 92-98.

Meili, M., Malm, O., Guimarãs, J. R. D., Forsberg, B. R., Padovanic, C. R., Viana, J. P. , and Silveira, E. G., (1996a) Mercury concentrations in tropical (Amazon) and boreal freshwater fish: natural spatial variability and pollution effects. In: Proceedings of the Fourth International Conference on Mercury as a Global Pollutant, Hamburg, Germany, 4 – 8 August, 1996.

Meili, M., Malm, O., Guimarães, J. R. D., Padovani, C., Forsberg, B. R., Viana J. P. and Silveira, E. G. (1996b) Bioaccumulation of mercury in tropical river food webs (Amazon):Similar patterns as in boreal lakes. In: Proceedings of the Fourth International Conference on Mercury as a Global pollutant., Hamburg, Germany, 4-8 August,1996.

Morel, F. M. M., Kraepiel, A. M. L., and Amyot, M. (1998) The chemical cycle and bio-accumulation of mercury. Annual Review of Ecology and Systematics, 29, 543–566.

Munthe, J. and Kindbom, K. (1997) Mercury in products, a source of transboundary pollutant transport. KEMI Report No. 10/97, The Swedish National Chemicals Inspectorate.

Munthe, J., Hultberg, H., Lee, Y. H., Parkman, H., Iverfeldt, Å. and Renberg, I. (1995) Trends of mercury and methylmercury in deposition, run-off water and sediments in relation to experimental manipulations and acidification. Water, Air and Soil Pollution, 85(2), 743-748.

Munthe, J., Wängberg, I., Iverfeldt, Å., Petersen, G., Ebinghaus, R., Schmolke, S., Bahlmann, E., Lindquist, O., Strömberg, D., Sommar J., Gårdfeldt K., Feng X., Larjava K. and Siemens V. (2001), Mercury species over Europe (MOE). Relative importance of depositional methylmercury fluxes to various ecosystems. Final report for the European Commission, Directorate General XII. September 2001.

Novak, P., Moffat, A. I. B. and Nalluri, C. (2007) Hydraulic Structures, Taylor & Francis Group, Fouth Edition, 4-78.

NRC (2000) Toxicological effects of methylmercury, Committee on the toxicological effects of methylmercury, Board on Environmental Studies and Toxicology, Commission of Life Sciences, National Research Council, National Academy Press, Washington DC.

Pacyna, J. M. and Pacyna, E. G. (2000) Assessment of emissions/discharges of mercury reaching the Arctic environment. The Norwegian Institute for Air Research, NILU Report OR 7/2000, Kjeller, Norway.

Pirrone, N., Munthe, J., Barregård, L., Ehrlich, H.C., Petersen, G., Fernandez, R., Hansen, J.C., Grandjean, P., Horvat, M., Steinnes, E., Ahrens, R., Pacyna, J.M., Borowiak, A., Boffetta, P. and Wichmann-Fiebig, M. (2001) EU Ambient Air Pollution by Mercury (Hg) - Position Paper. Office for Official Publications of the European Communities, 2001.

Porvari, P. (1998) Development of fish mercury concentrations in Finnish reservoirs from 1979 to 1994. Science of the Total Environment, 213, 279 – 290.

Ramsey, D. J. and Ramlal, P. S. (1986) Measurements of Mercury Methylation Balance in Relation to Concentration of the Total Mercury in Northern Manitoba Reservoirs and their use in Predicting the Duration of the Fish Mercury Problem in new Reservoirs. In: Technical Appendices to the summary Report of the Canada – Manitoba Agreement on the study and Monitoring of Mercury in the Churchill River Diversion, 1, 55.

Rea, A. W., Lindberg, S. E. and Keeler, G. J. (2000) Assessment of dry deposition and foliar leaching of mercury and selected trace elements based on washed foliar and surrogate surfaces. Environmental Science and Technology, 34, 2418-2425.

Rea, A. W., Lindberg, S. E., and Keeler, G. J., (2001) Dry deposition and foliar leaching of mercury and selected trace elements in deciduous forest throughfall. Atmospheric Environment, 35, 3453-3462.

Rissanen, T., Voutilainen, S., Nyyssönen, K., Lakka, T. A. and Salonen, J. T. (2000) Fish-oil derived fatty acids, docosahexaenoic acid, and the risk of acute coronary events. Circulation 2000, 102, 2677-2679.

Rodgers, D.W., Dickman, M. and Han, X. (1995) Stories from old reservoirs: Sediment Hg and Hg methylation in Ontario Hydroelectric developments. Water, Air and Soil Pollution, 80, 829-839.

Rondón, A. and Pérez, L. E. (1999) Mercury bioaccumulation in fifteen dams of Bolivar State (Venezuela), estimated with the indicator fish *Hoplias malabaricus*. In: Barbosa, J., Melamed, R., Villas Bôas, R., (Eds.) Mercury as a Global Pollutant – Fifth International Conference, May 23 -27, 1999, Rio de Janeiro, Brazil. CETEM. Center for Mineral Technology. Rio de Janeiro, Brazil, 277, 592.

Rose, R., Hutcheson, M. S., West, C. R., Pancorbo, O., Hulme, K., Cooperman, A., DeCesare, G., Isaac, R. and Screpetis A. (1999) Fish mercury distribution in

Massachusetts, USA lakes. Environmental Toxicology and Chemistry, 18(7), 1370-1379.

Rosenberg, D. M., Berkes, F., Bodaly, R. A., Hecky, R. E., Kelly, C. A. and Rudd, J. W. M. (1997) Large scale Impacts of Hydroelectric Development, Environmental Rev., 5, 27-54.

Salonen, J.T., Seppanen, K., Nyyssonen, K., Korpela, H., Kauhanen, J., Kantola, M., Tuomilehto, J., Esterbauer, H., Tatzber, F. and Salonen, R. (1995) Intake of mercury from fish, lipid peroxidation, and the risk of myocardial infarction and coronary, cardiovascular, and any death in eastern Finnish men. Circulation 1995, 91, 645-55.

Schetagne, R. and Plante, M. (2006) Mercury in Hydroelectric Reservoirs. Hydro-Québec Production. Fact Sheet, 1, 2-5.

Schetagne, R., Doyon, J. F. and Fournier, J. J. (2000) Export of Mercury Downstream from Reservoirs. The Science of the Total Environment, 260,135-145.

Shanley, J. B., Kamman, N. C., Clair, T. A., Chalmers, A. (2005). Physical controls on total and methylmercury concentrations in streams and lakes of the northeastern USA. Ecotoxicology, 14, 125–134.

Simola, H. and Lodenius, M. (1982) Recent increase in mercury sedimentation in a forest lake attributable to peatland drainage. Bull. Environ. Contam. Toxicol., 29, 298-305.

Sjoblom, A., Meili, M. and Sundbom, M. (2000) The influence of Humic Substances on the speciation and Bioavailability of Dissolved Mercury and Methylmercury, measured as uptake by Chaoborus Larvae and Loss by Volatilization. The science of the Total Environment, 261, 115-124.

Sommar, J., Feng, X. and Lindqvist, O. (1999) Speciation of volatile mercury species present in digester and deposit gases. Applied Organometallic Chemistry, 13, 441-447.

Sorensen, J. A., Glass, G. E., Schmidt, K. W., Huber, J. K. and Rapp Jr, G. R. (1990) Airborne mercury deposition and watershed characteristics in relation to mercury concentrations in water, sediments, plankton and fish of eighty northern Minnesota lakes. Environmental Science and Technology, 24(11), 1716-1727.

Stokes, P. M. and Wren, C. D. (1987) Bioaccumulation of Mercury by Aquatic Biota in Hydroelectric Reservoirs. Chapter 16. SCOPE 31. John Wiley New York, 225 – 277.

Takeuchi, T. and Eto, K. (1999) The Pathology of Minamata Disease. A Tragic Story of Water Pollution. Fukuoka, Kyushu University Press, 1999.

Thompson, J. D. (1985) Mercury concentrations of the axial muscle tissue of some marine fishes of the continental shelf adjacent to Tasmania. Australian J. Marine and Freshwater Research, 36, (4), 509-517.

Tremblay, A., Cloutier, L. and Lucotte, M. (1998) Total Mercury and Methylemercury Fluxes via Emerging Insects in Recently Flooded Hydroelectric Reservoirs and a Natural Lake. The Science of the Total Environment, 219, 209-221.

Tropp, M. (2000) Mechanisms for and Effects of Elevated Mercury Levels in Fish after Flooding of Hydroelectric Reservoirs, a Literature Study and Baseline Data for APM-Manso, Brazil. Department of Limnology, Evolutionary Biology Centre, Uppsala University, Uppsala, Sweden.

Uchida, M., Hirakawa, Y. and Inoue, T. (1961) Biochemical studies on Minamata disease IV. Isolation and chemical identification of the mercury compound in the toxic shellfish with special refrence to the causal agent of the disease. Kumamoto Medical Journal, 14, 181-184.

Ullrich, S. M., Tanton, T. W. and Abdrashitova, S. A. (2001) Mercury in the Aquatic Environment. A Review of Factors affecting Methylation. Critical Reviews in Environmental Science and Technology, 31, 241-293.

UNEP (2002) Global Mercury Assessment Report, 1-258.

UNEP (1999) Global 500 Forum. Courting Mega-disaster: Bui Dam May Cause Havoc Stories. <http://www.global500.org/feature _4.html>

US EPA (1997) Mercury study report to congress. USEPA, Dec. 1997. http://www.epa.gov/airprogm/oar/mercury.html, January 2001.

US EPA (2001) Mercury update: Impact on fish advisories. EPA Fact sheet, June 2001. Found on http://www.epa.gov/ost/fish, June 2001.

Verdon, R., Brouard, D., Demers, C., Lalumiere, R., Laperle, M. and Schetagne, R. (1991) Mercury Evolution (1978-1988) in Fishes of La Grande Hydroelectric Complex, Québec, Canada. Water, Air, and Soil Pollution, 56, 405-417.

Verta, M. (1984) The Mercury Cycles in Lakes; Some New Hypotheses. Aqua Fennica, 14, 215-221.

Vette, A. F., Landis, M. S. and Keeler, G. J. (2002) Deposition and Emission of Gaseous Mercury to and from Lake Michigan during the Lake Michigan Mass Balance Study (July, 1994 - October, 1995), submitted to Environmental Science and Technology.

Wasserman, J. C., Hacon, S. and Wasserman, M. A. (2003) Biogeochemistry of mercury in the Amazonian environment. Ambio, 32, 336 – 342.

WHO (1976) Environmental Health Criteria 1: Mercury, Geneva, World Health Organization, 131.

WHO/IPCS (1990) Methylmercury. Environmental Health Criteria No 101, World Health Organisation, International Programme on Chemical Safety (IPCS), Geneva, Switzerland, 1990.

WRM (World Rainforest Movement) (2006) Bulletin #102, Ghana: A dam at the cost of the trees. http://www.wrm.org.uy/bulletin/102/Ghana.html>.

Yoshino, Y., Mozai, T. and Nakao, K. (1966) Biochemical changes in the brain poisoned with an alkyl mercury compound, with special reference to the inhibition of protein synthesis in the brain cortex slices. J., Neurochem, 13, 1223 – 1230.

APPENDIX

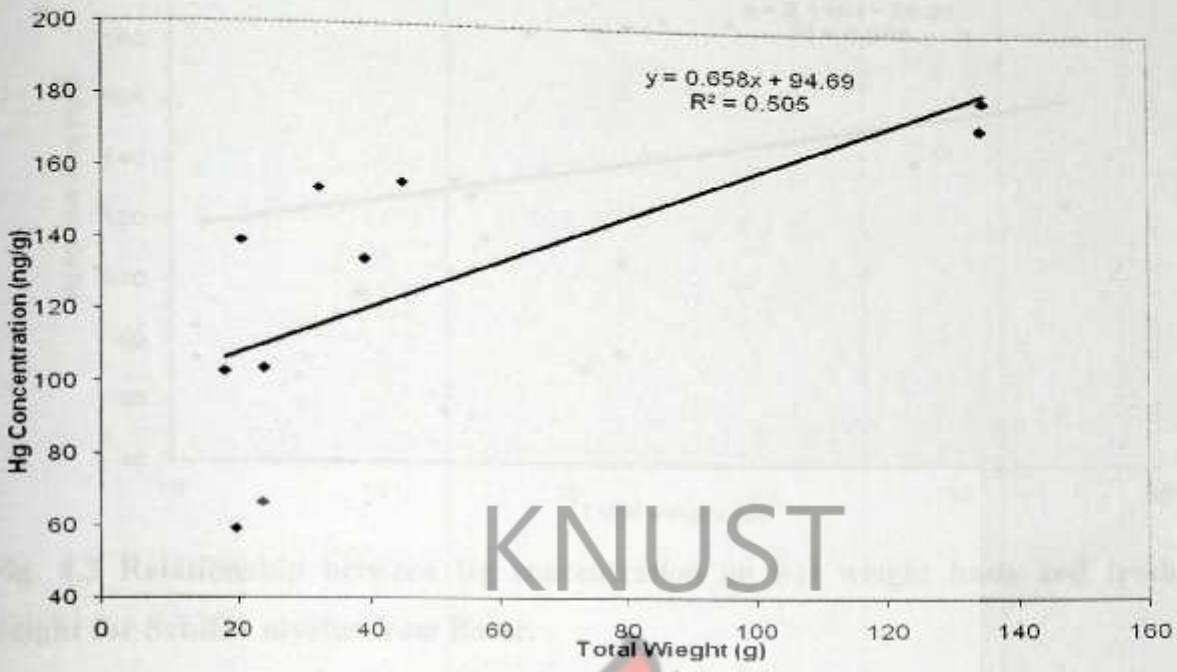


Fig. 4.1 Relationship between Hg concentration on wet weight basis and fresh weight for Lates niloticus from Bator.

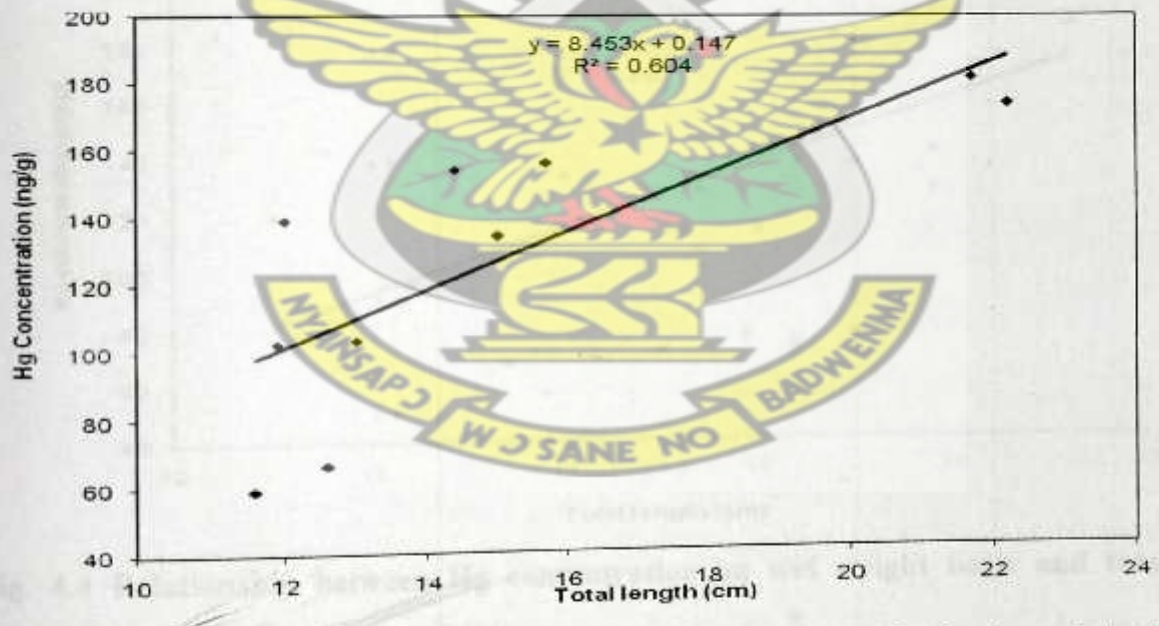


Fig. 4.2 Relationship between Hg concentration on wet weight basis and total length for Lates niloticus from Bator.

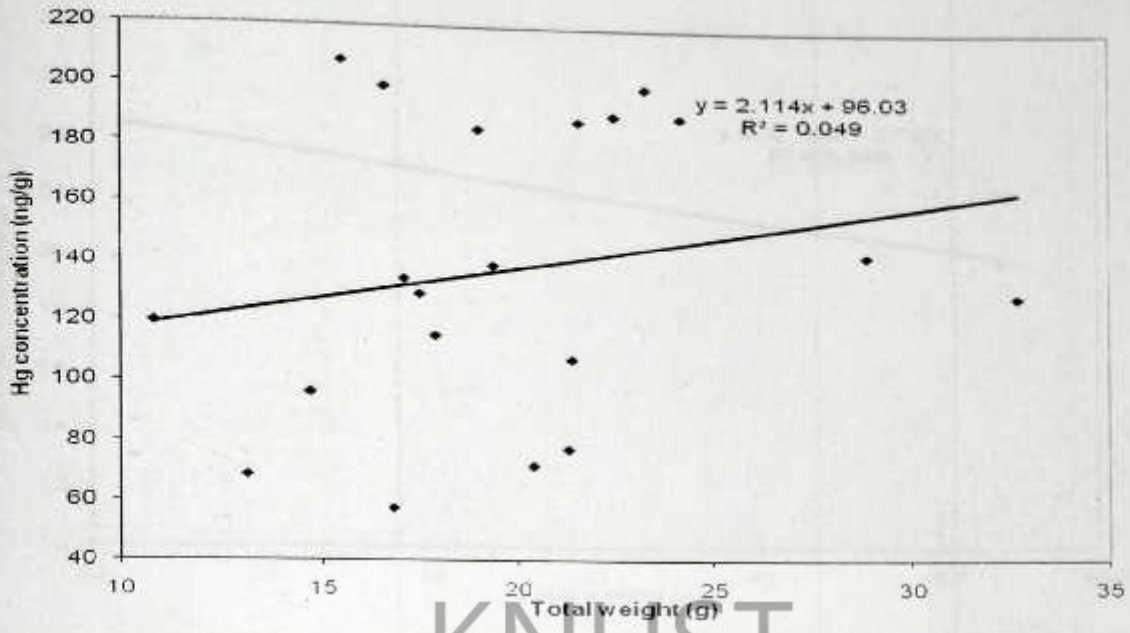


Fig. 4.3 Relationship between Hg concentration on wet weight basis and fresh weight for Schilbe mystus from Bator.

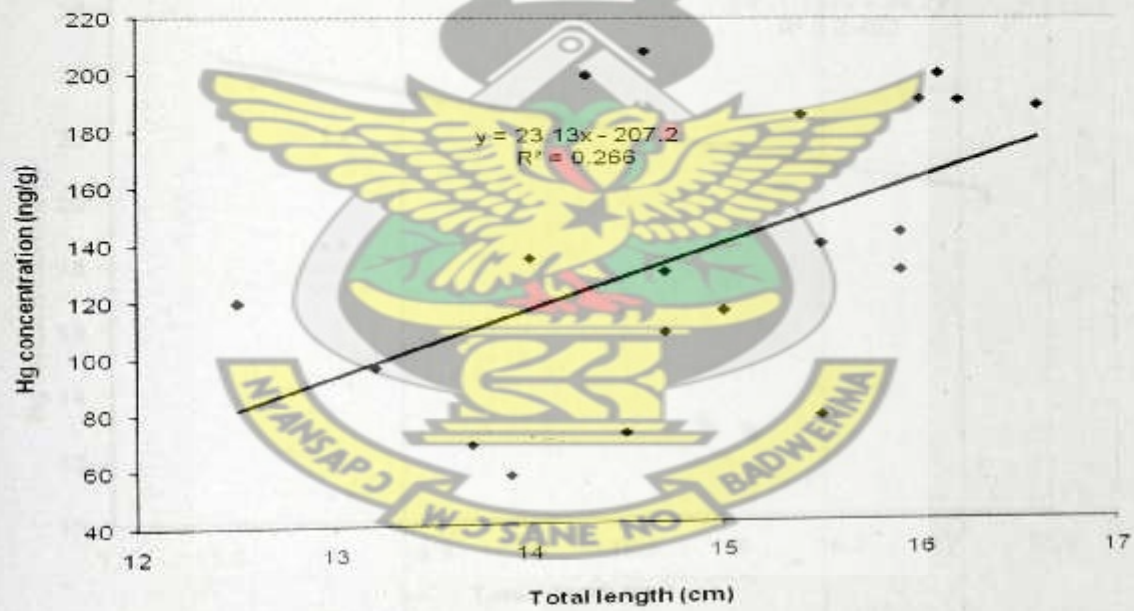


Fig. 4.4 Relationship between Hg concentration on wet weight basis and total length for Schilbe mystus from Bator.

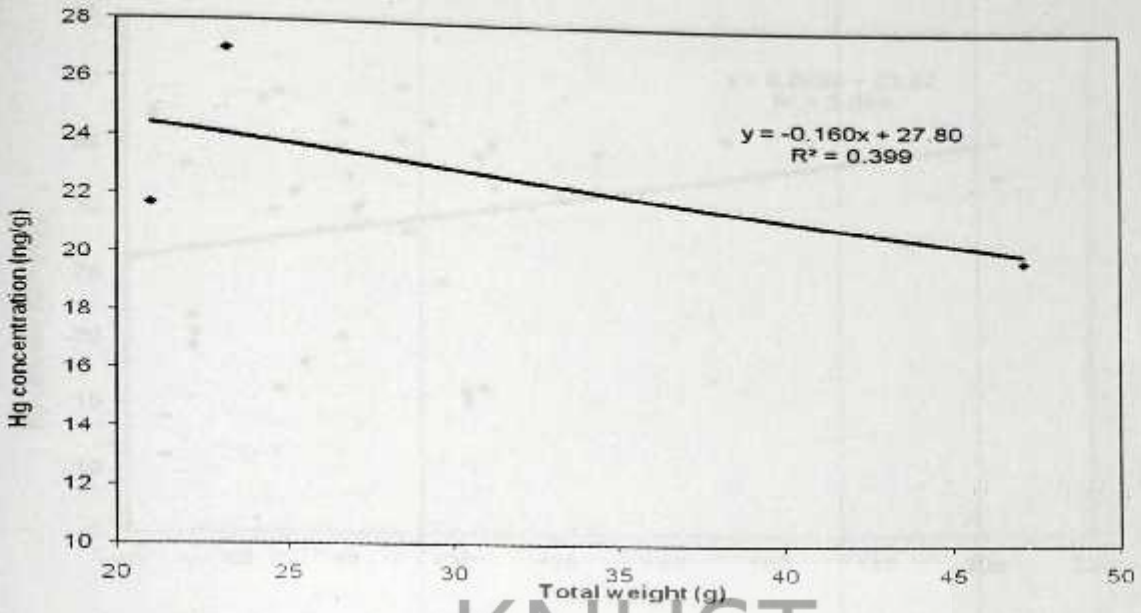


Fig. 4.5 Relationship between Hg concentration on wet weight basis and fresh weight for Alestes dentex from Bator.

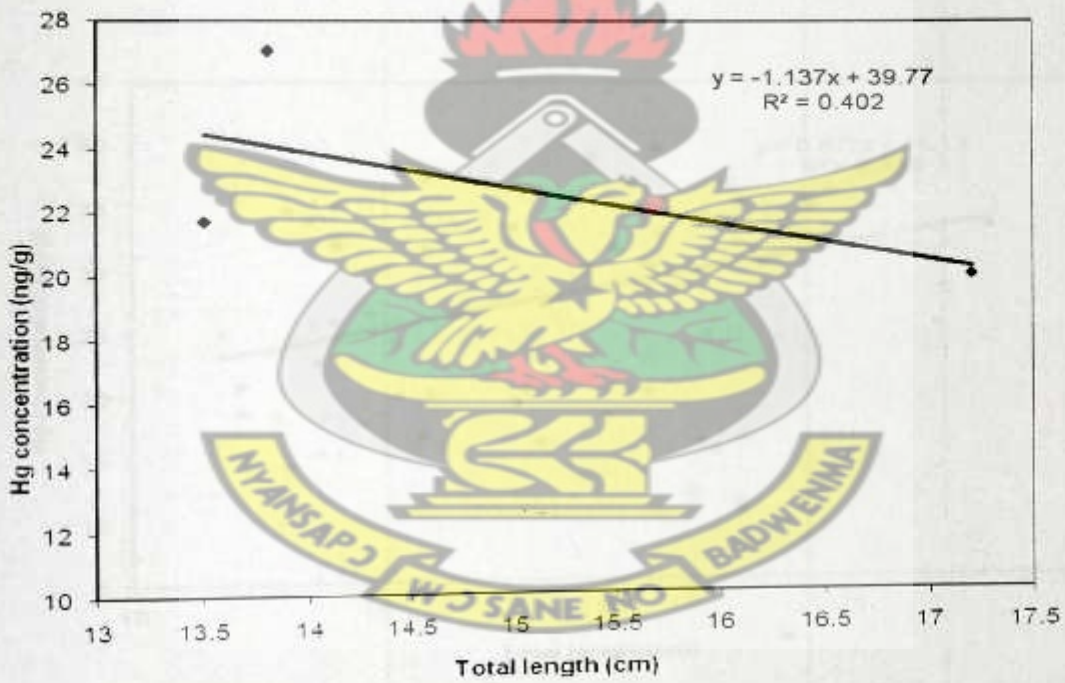


Fig. 4.6 Relationship between Hg concentration on wet weight basis and total length for Alestes dentex from Bator.

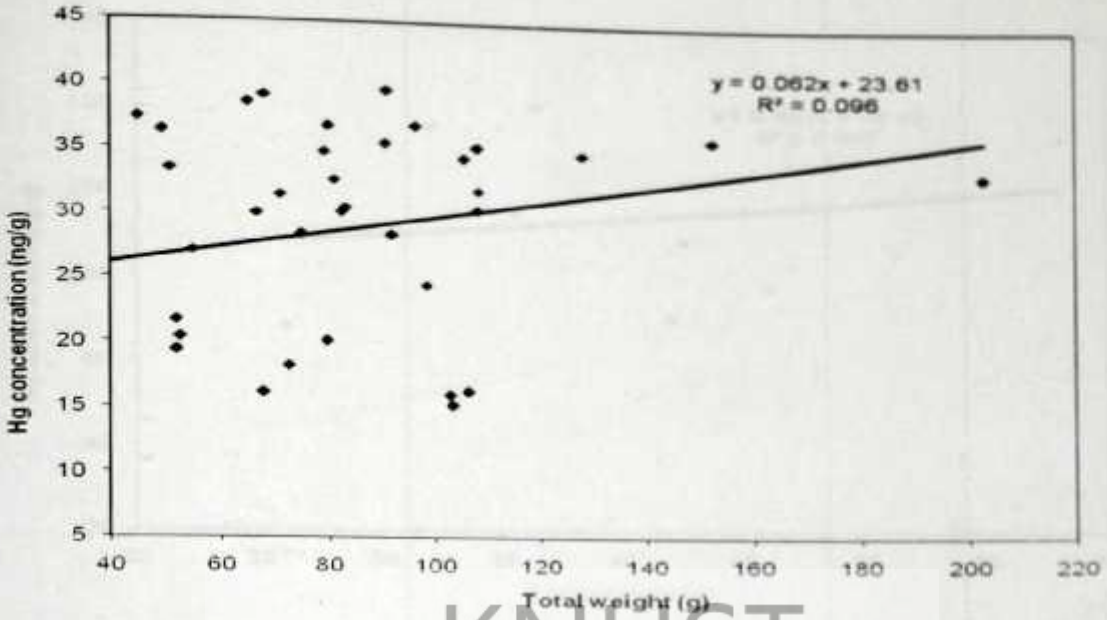


Fig. 4.7 Relationship between Hg concentration on wet weight basis and fresh weight for Brycinus nurse from Bator.

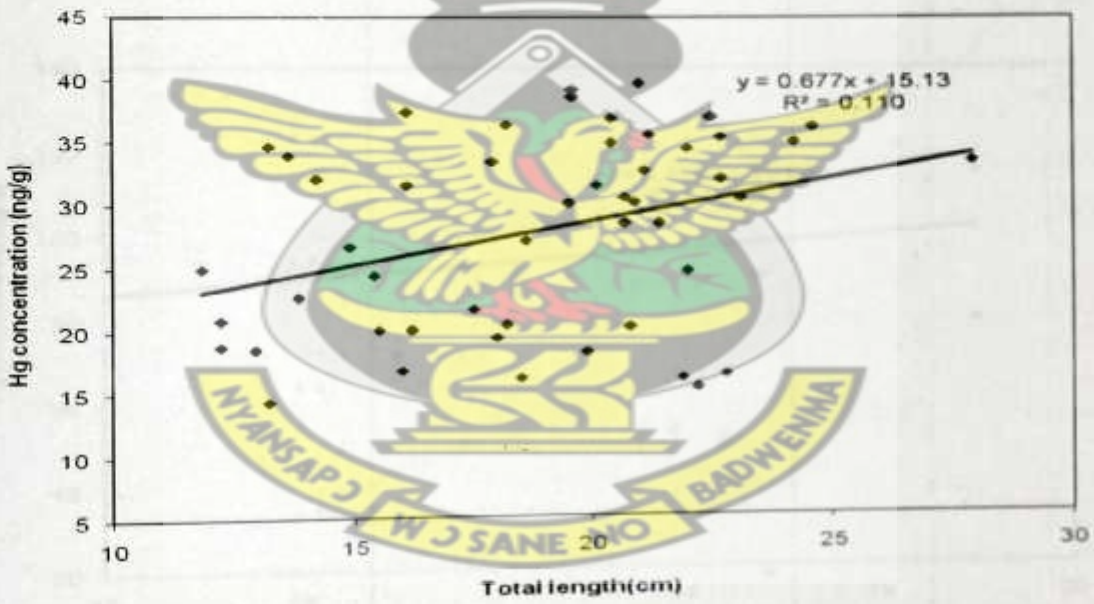


Fig. 4.8 Relationship between Hg concentration on wet weight basis and total length for Brycinus nurse from Bator.

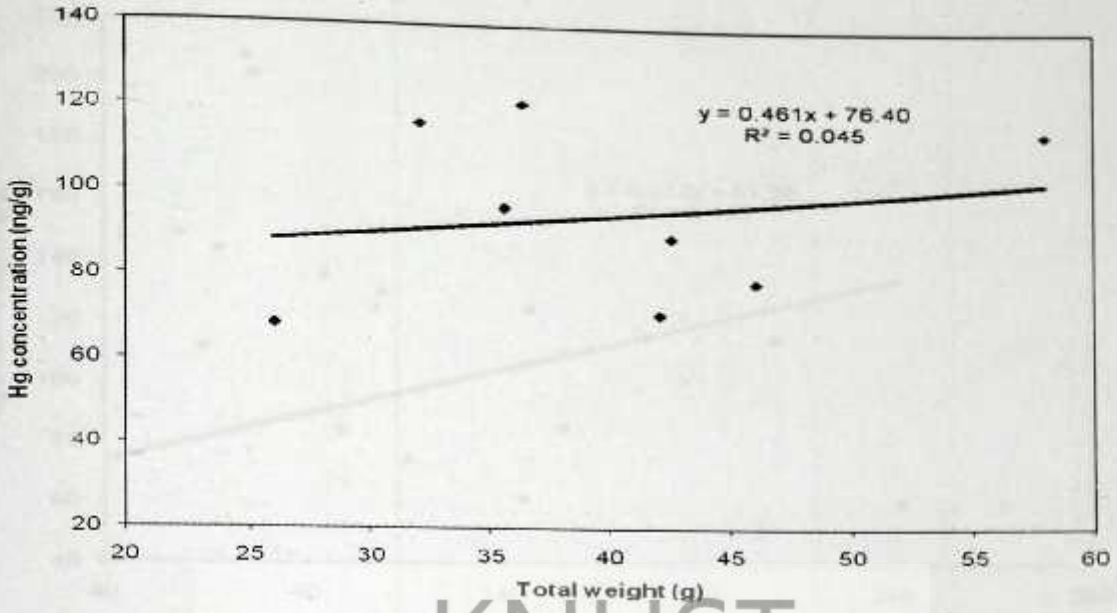


Fig. 4.9 Relationship between Hg concentration on wet weight basis and fresh weight for Chysichthys auratus from Bator.

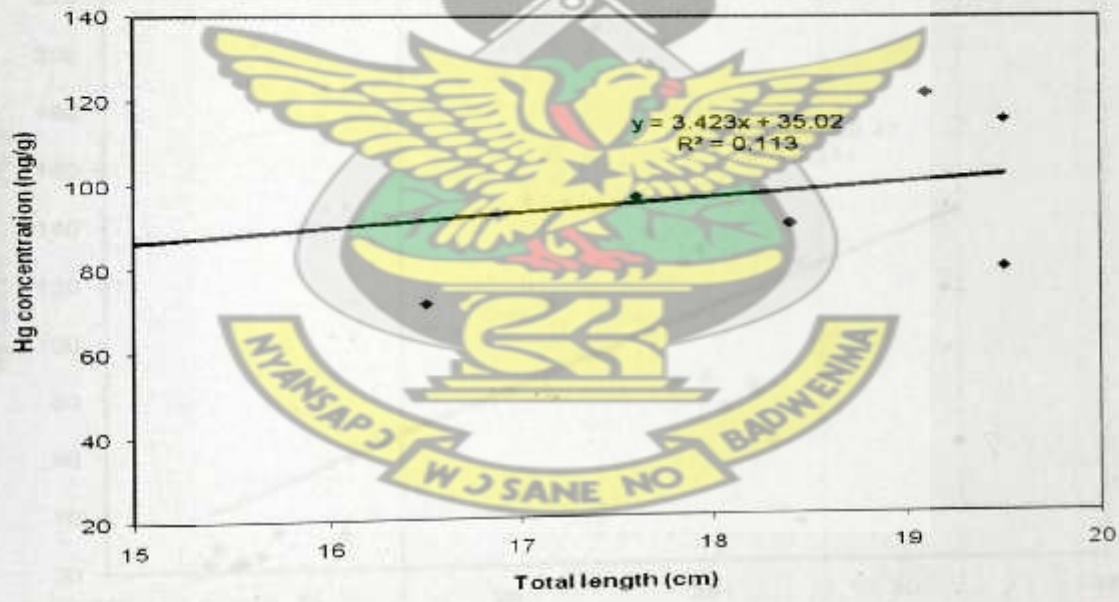


Fig. 4.10 Relationship between Hg concentration on wet weight basis and total length for Chysichthys auratus from Bator.

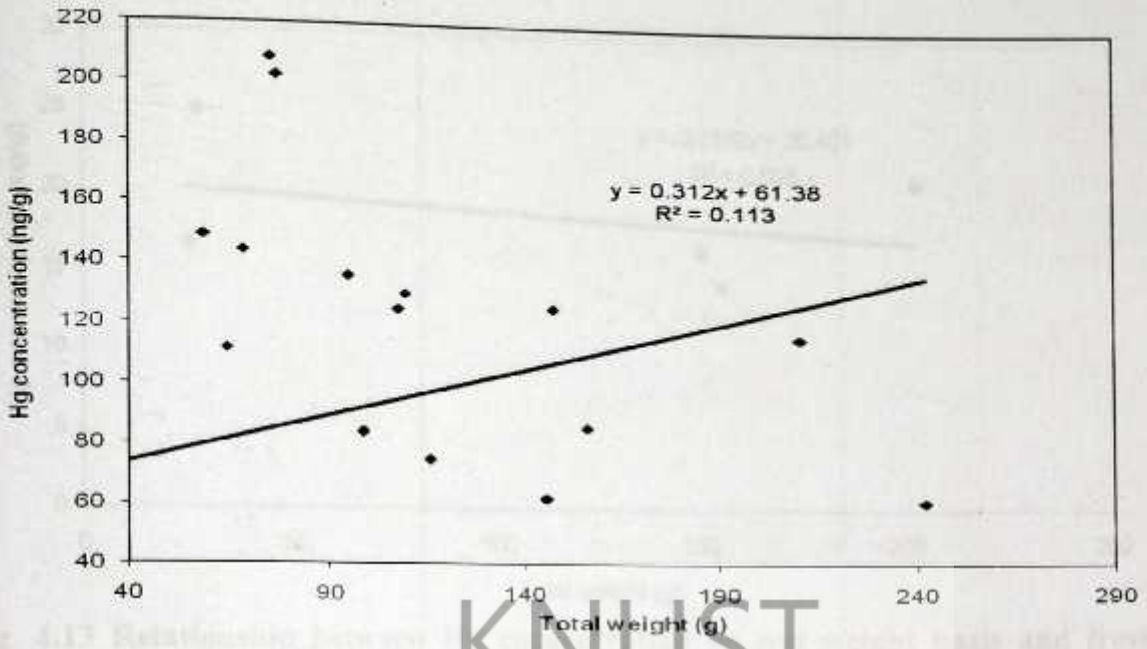


Fig. 4.11 Relationship between Hg concentration on wet weight basis and fresh weight for Hydrocynus sp from Bator.

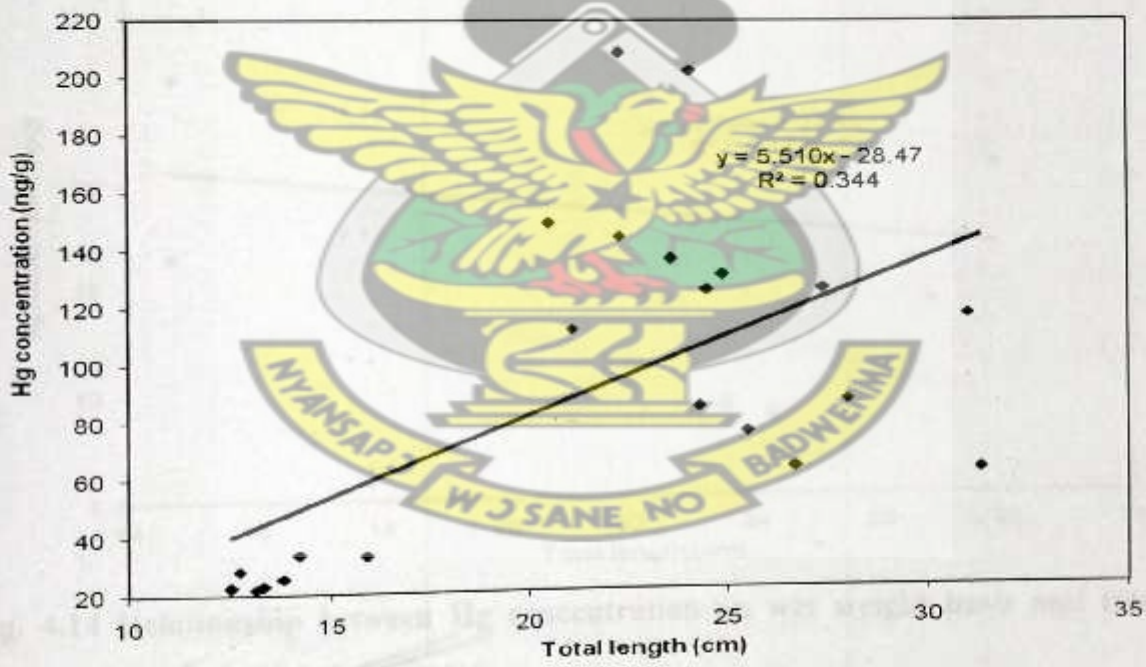


Fig. 4.12 Relationship between Hg concentration on wet weight basis and total length for Hydrocynus sp from Bator.

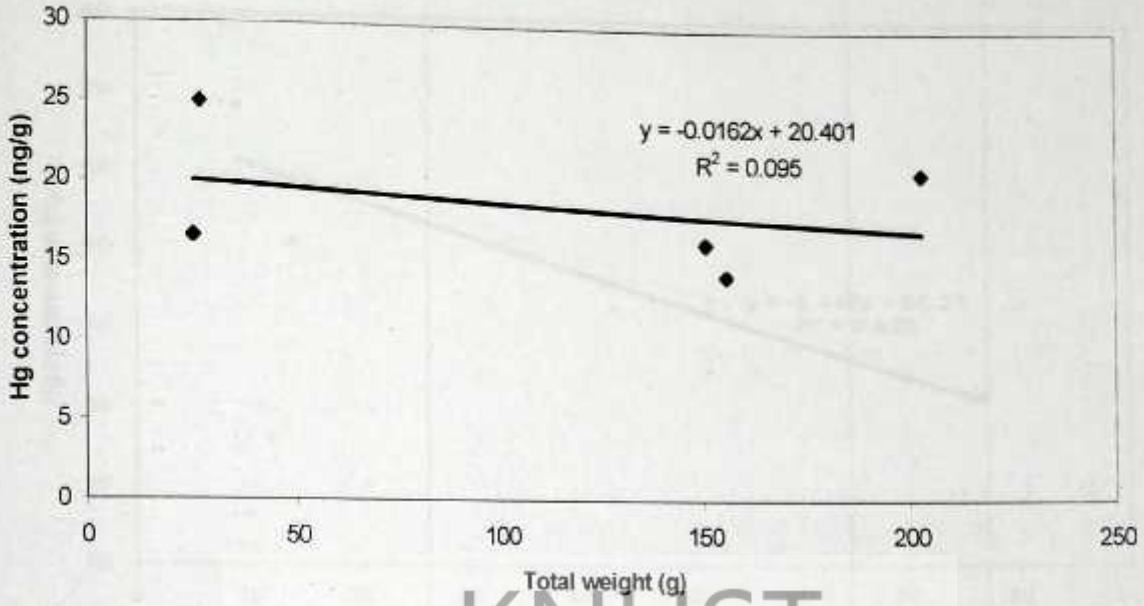


Fig. 4.13 Relationship between Hg concentration on wet weight basis and fresh weight for Labeo coubie from Bator.

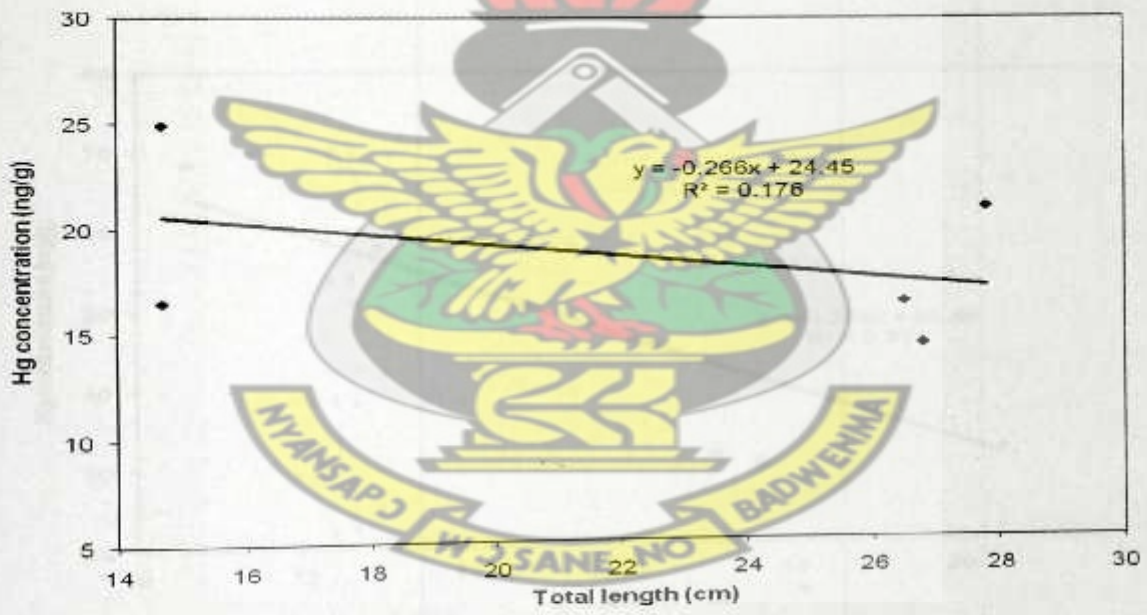


Fig. 4.14 Relationship between Hg concentration on wet weight basis and total length for Labeo coubie from Bator.

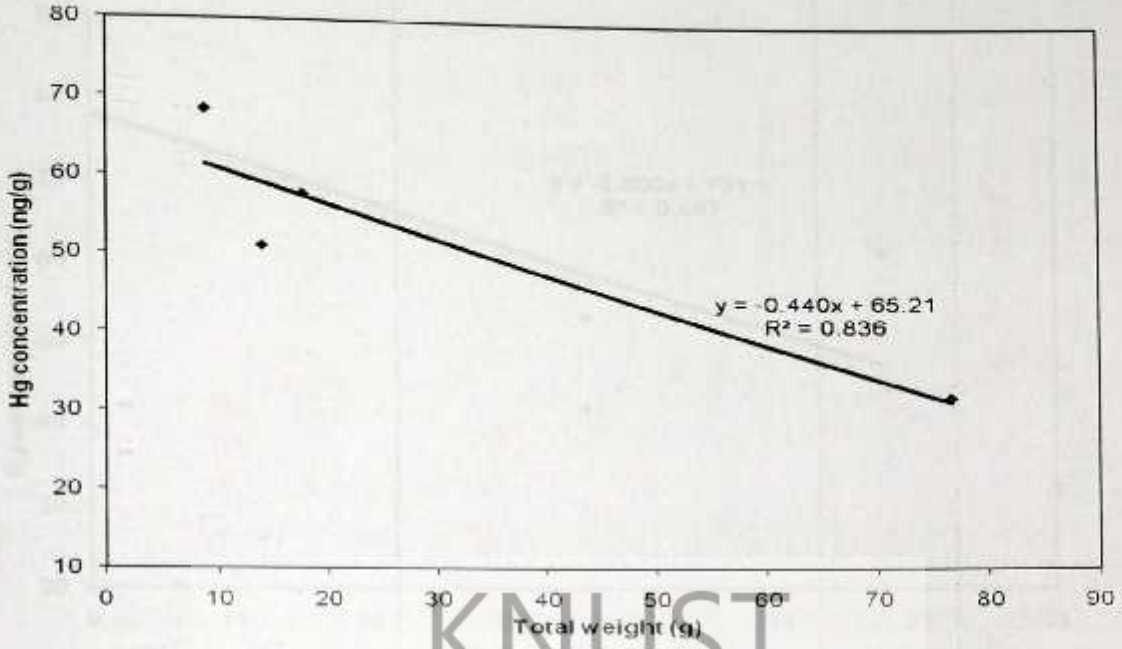


Fig. 4.15 Relationship between Hg concentration on wet weight basis and fresh weight for Synodontis oceillifer from Bator.

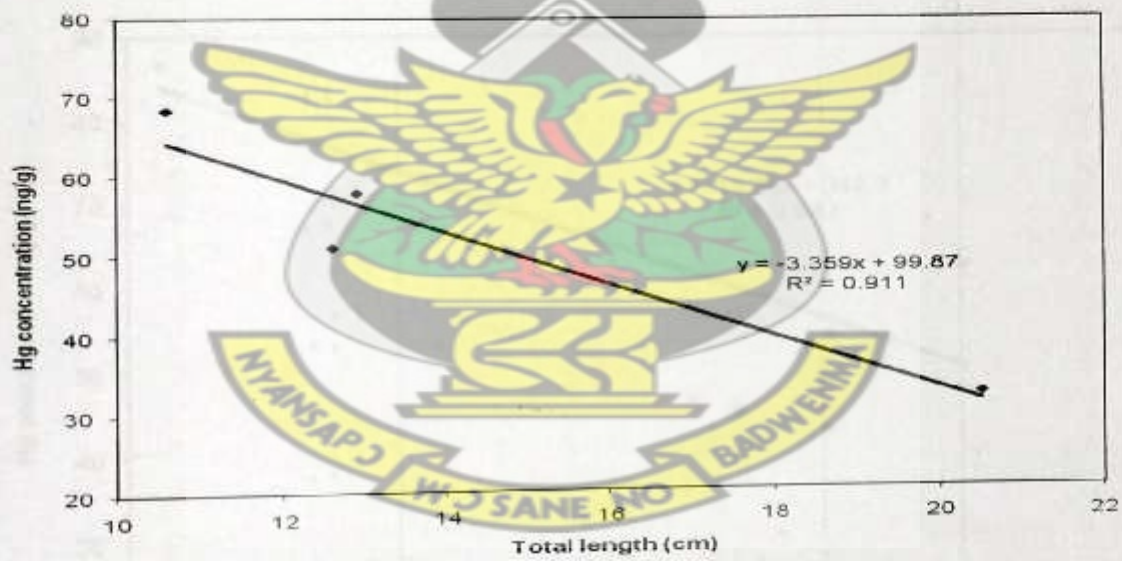


Fig. 4.16 Relationship between Hg concentration on wet weight basis and total length for Synodontis oceillifer from Bator.

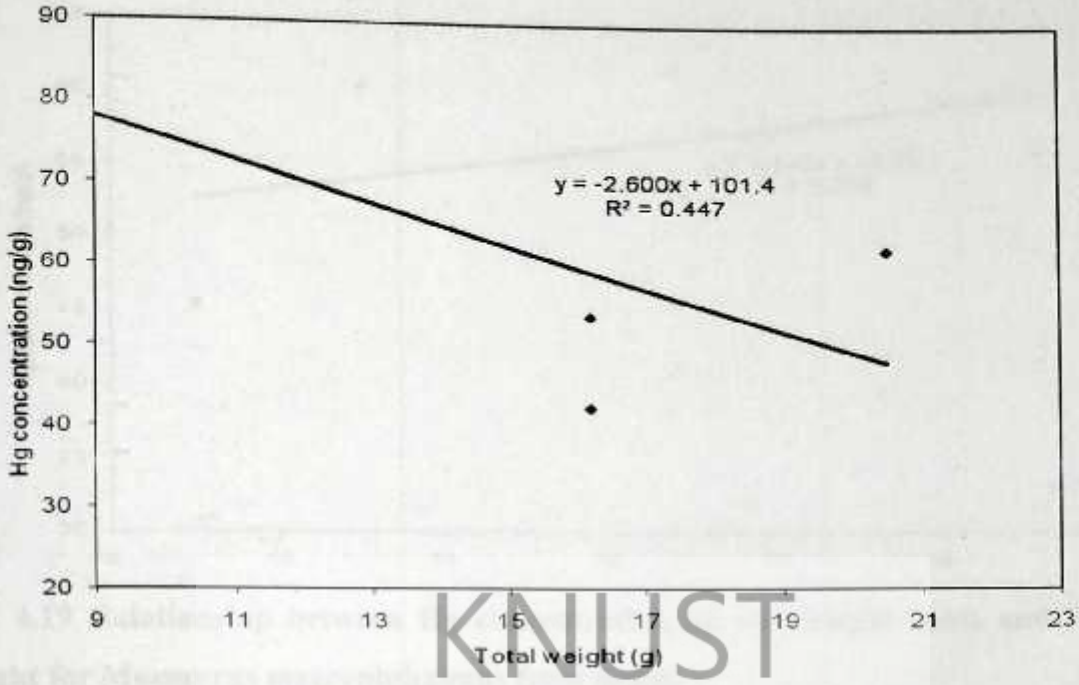


Fig. 4.17 Relationship between Hg concentration on wet weight basis and fresh weight for Synodontis sp from Bator

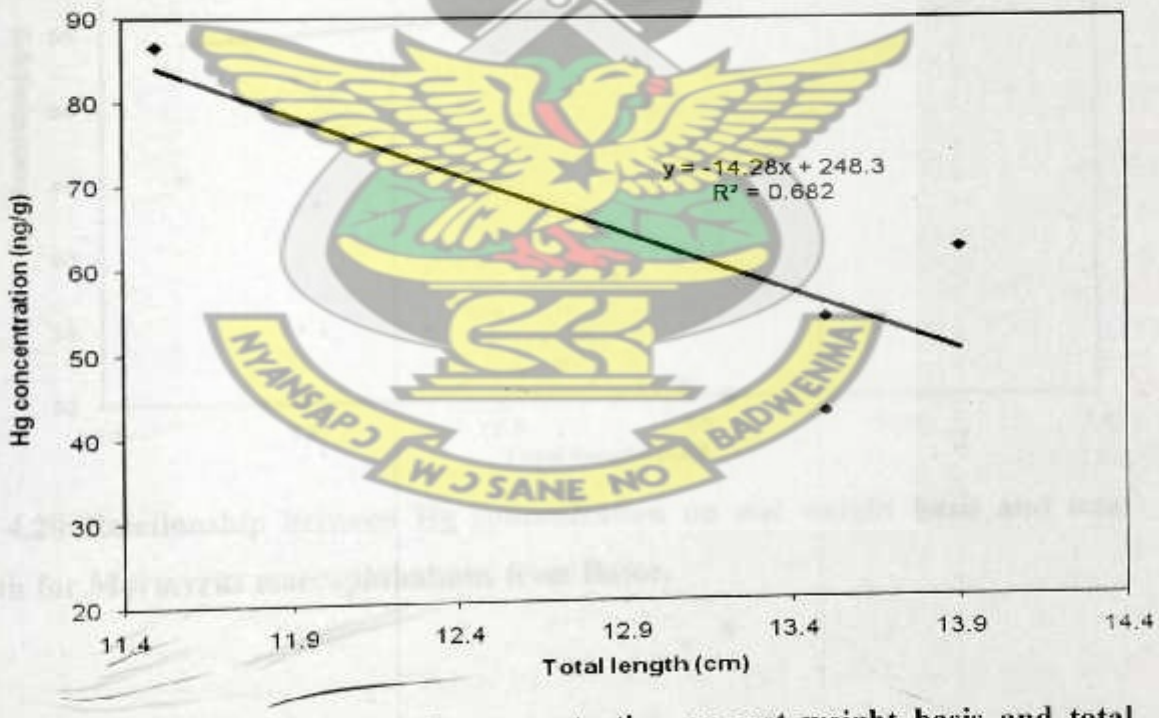


Fig. 4.18 Relationship between Hg concentration on wet weight basis and total length for Synodontis sp from Bator.

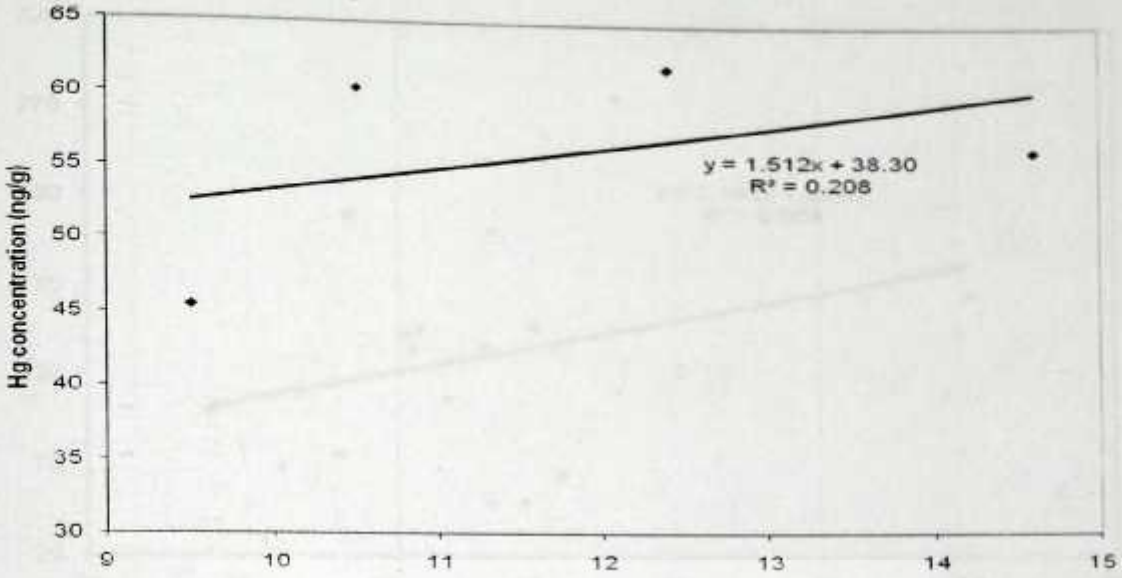


Fig. 4.19 Relationship between Hg concentration on wet weight basis and fresh weight for Mormyrus macrophthalmus from Bator.

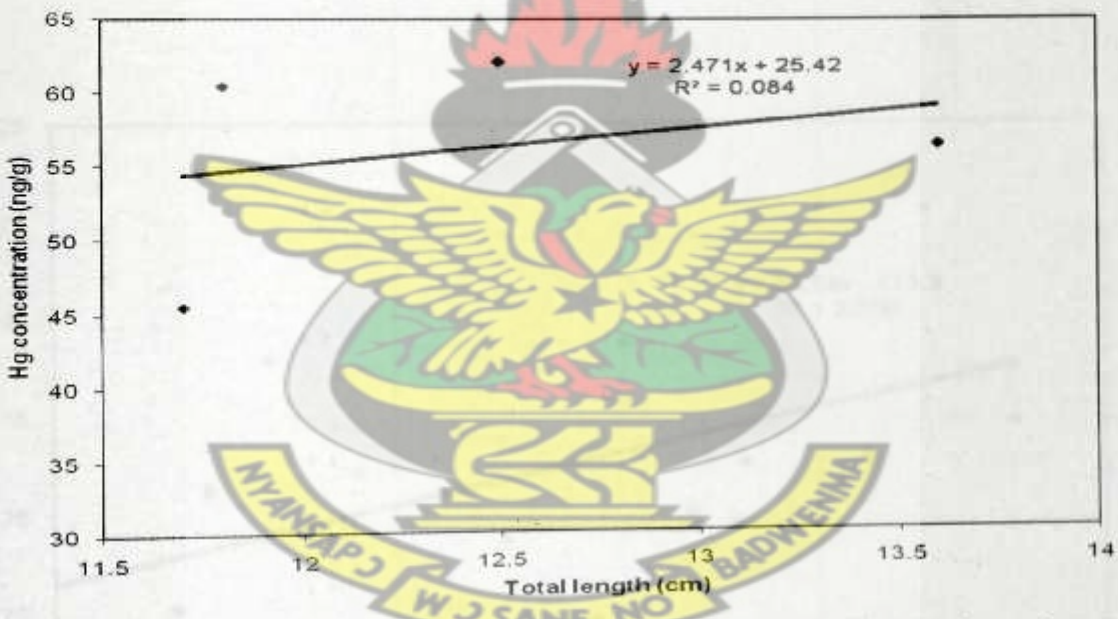


Fig. 4.20 Relationship between Hg concentration on wet weight basis and total length for Mormyrus macrophthalmus from Bator.

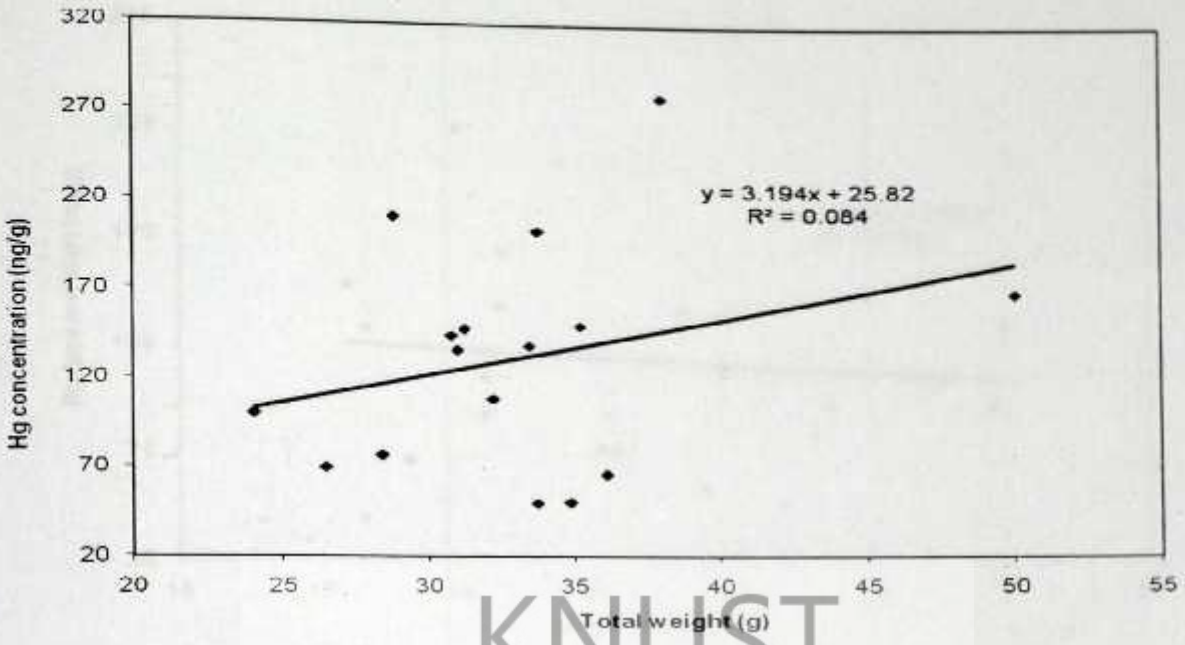


Fig. 4.21 Relationship between Hg concentration on wet weight basis and fresh weight for Schilbe mystus from Dam site.

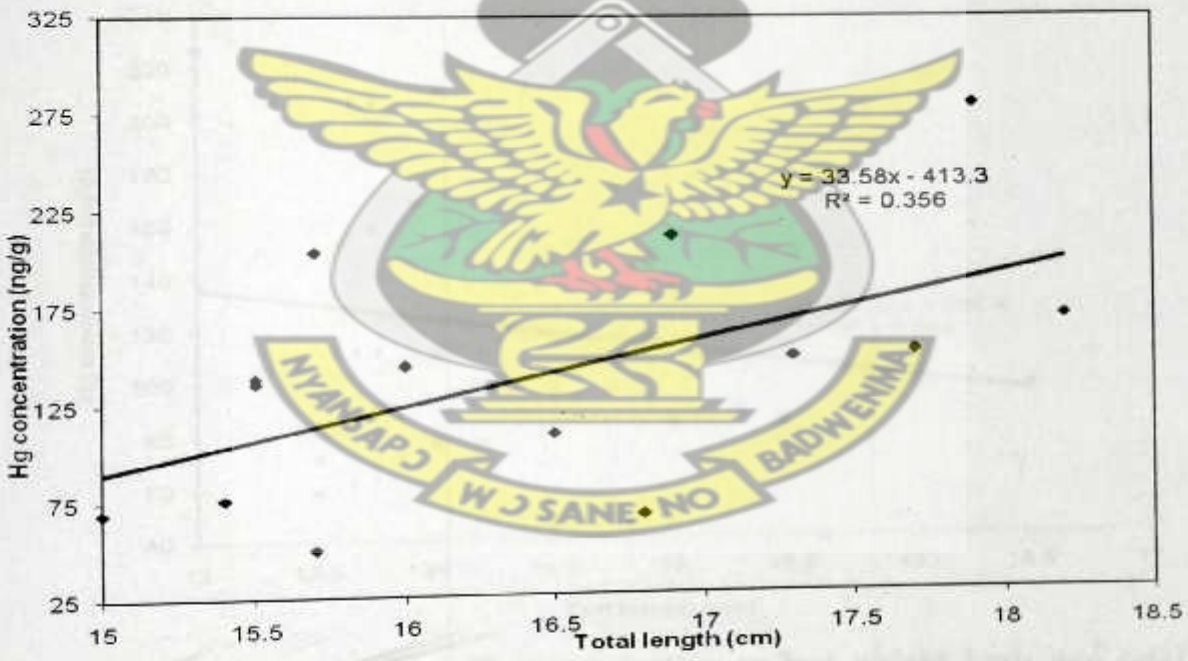


Fig. 4.22 Relationship between Hg concentration on wet weight basis and total length for Schilbe mystus from Dam site.

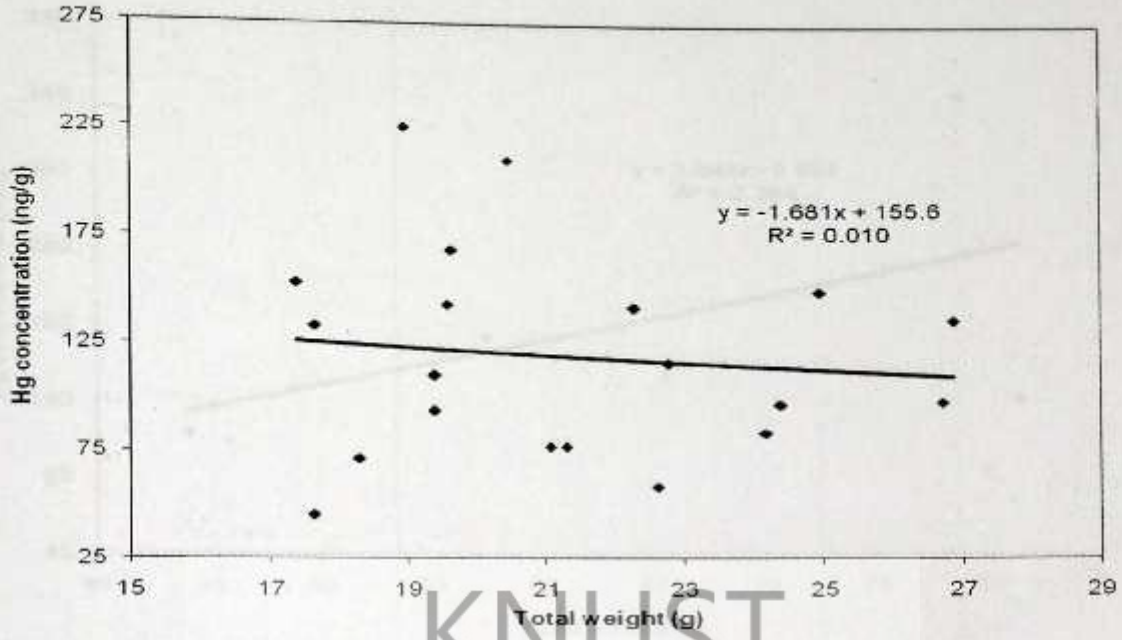


Fig. 4.23 Relationship between Hg concentration on wet weight basis and fresh weight for Schilbe mandibularies from Dam site.

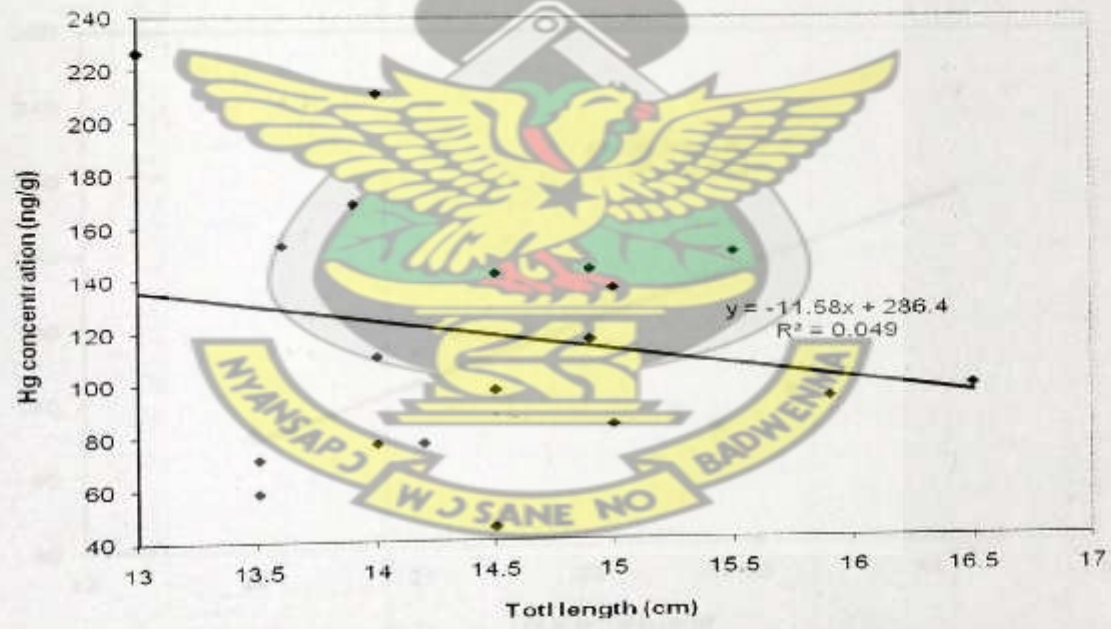


Fig. 4.24 Relationship between Hg concentration on wet weight basis and total length for Schilbe mandibularies from Dam site.

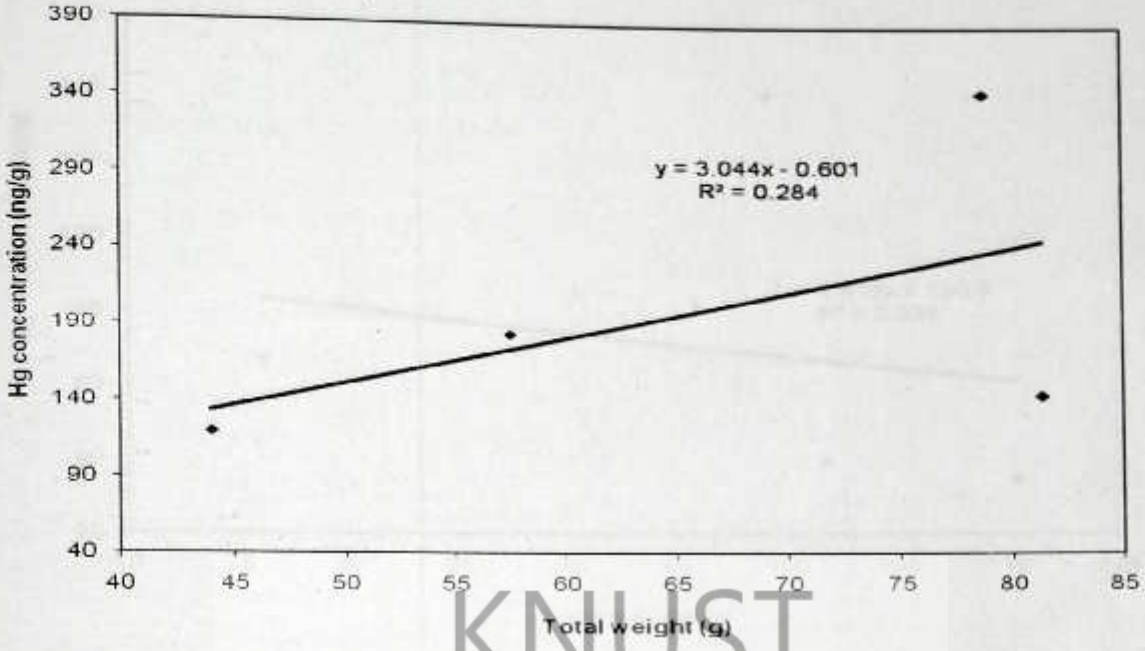


Fig. 4.25 Relationship between Hg concentration on wet weight basis and fresh weight for Hydrocynus forkali from Dam site.

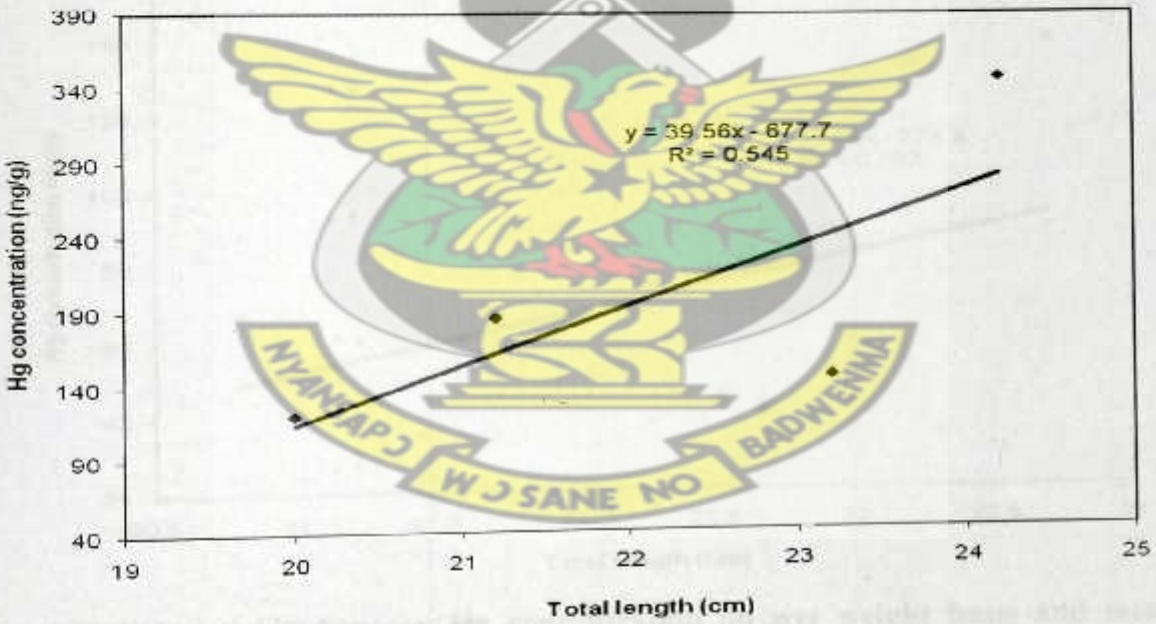


Fig. 4.26 Relationship between Hg concentration on wet weight basis and total length for Hydrocynus forkali from Dam site.

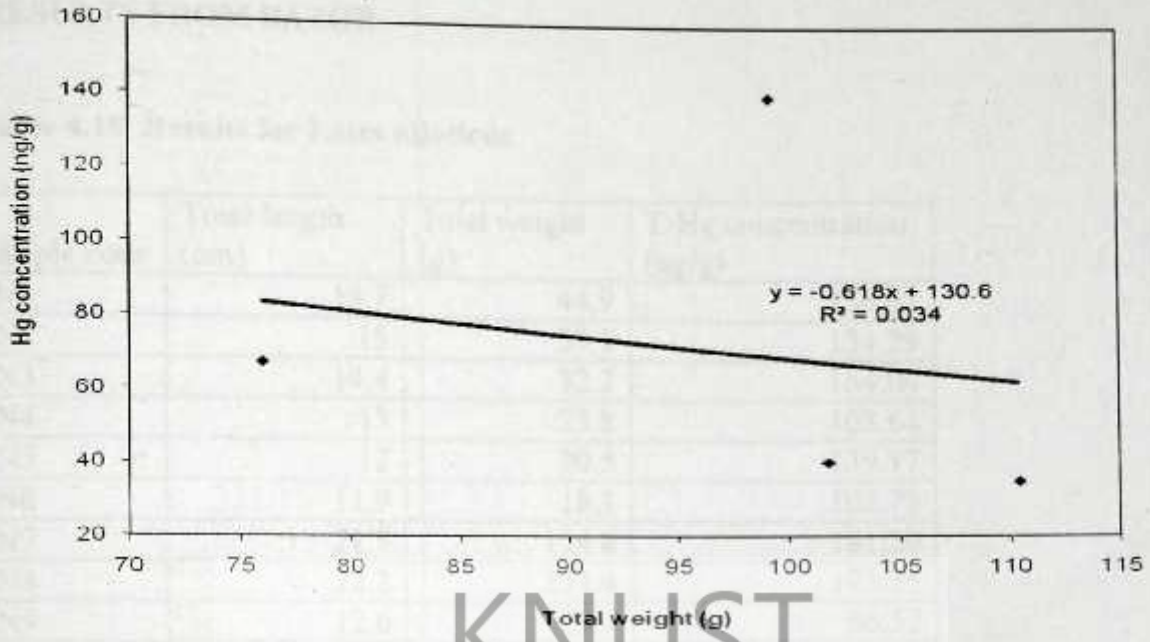


Fig. 4.27 Relationship between Hg concentration on wet weight basis and fresh weight for Brycinus imberi from Dam site.

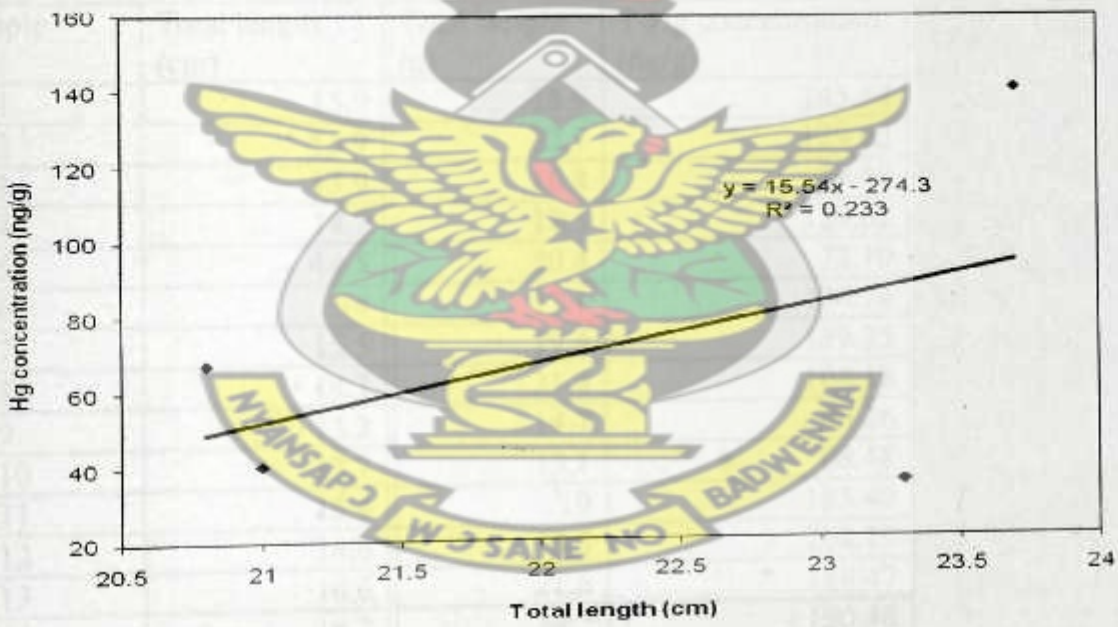


Fig. 4.28 Relationship between Hg concentration on wet weight basis and total length for Brycinus imberi from Dam site.

RESULTS FROM BATOR

Table 4.10 Results for Lates niloticus

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
LN1	15.7	44.9	156.02
LN2	15	39.1	134.29
LN3	14.4	32.2	154.04
LN4	13	23.8	103.64
LN5	12	20.5	139.17
LN6	11.9	18.1	102.75
LN7	21.7	134.8	181.29
LN8	22.2	134.4	173.35
LN9	12.6	23.7	66.52
LN10	11.6	19.7	59.25

LN=Lates niloticus

Table 4.11 Results for Schilbe mystus

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
SM1	15.9	28.9	143.54
SM2	15.9	32.7	129.62
SM3	13.9	16.8	57.49
SM4	14.7	17.5	129.62
SM5	14.5	20.4	72.10
SM6	14	17.1	134.74
SM7	15.5	19.4	139.35
SM8	14.7	21.4	108.18
SM9	13.2	14.7	96.26
SM10	13.7	13.1	68.48
SM11	15.4	19	185.40
SM12	14.6	15.5	208.19
SM13	16.6	21.6	188.47
SM14	16.2	24.2	190.48
SM15	15.5	21.3	77.74
SM16	16	22.5	190.63
SM17	12.5	10.8	119.66
SM18	15	17.9	115.63
SM19	16.1	23.3	200.02
SM20	14.3	16.6	199.60

SM=Schilbe mystus

Table 4.12 Results for Alestes dentex

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
AD1	17.2	47.1	19.97
AD2	13.8	23.2	27.04
AD3	13.5	21	21.70

AD=Alestes dentex

Table 4.13 Results for Brycinus nurse

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
BN1	20.4	79.1	34.88
BN2	23.1	108	30.48
BN3	27.9	202.9	33.42
BN4	22.7	108.2	32.02
BN5	21.2	90.6	35.59
BN6	20.7	74.7	28.45
BN7	24.6	152.6	36.17
BN8	22	98.3	24.61
BN9	24.2	128	34.95
BN10	17.9	50.7	33.45
BN11	18.2	49.3	36.37
BN12	19.5	66.6	30.08
BN13	20.9	82.4	30.19
BN14	21.4	91.8	28.45
BN15	18.6	54.7	27.14
BN16	22.7	108.1	35.37
BN17	19.6	64.8	38.61
BN18	20.7	83.1	30.57
BN19	21	90.9	39.68
BN20	21.1	80.9	32.71
BN21	20.1	70.9	31.50
BN22	22	105.5	34.48
BN23	20.4	79.8	36.86
BN24	22.5	96.4	36.93
BN25	16.1	44.9	37.38
BN26	13.2	24.8	34.64
BN27	19.6	67.9	39.22
BN28	22.8	106.1	16.35

BN29	20.8	79.5	20.17
BN30	18.2	52.6	20.40
BN31	18	51.8	19.39
BN32	18.5	67.7	16.13
BN33	22.2	103.2	15.28
BN34	19.9	72.4	18.22
BN35	21.9	102.7	16.07
BN36	16	35.9	16.81
BN37	14.9	30.1	26.73
BN38	17.5	51.7	21.68
BN39	15.5	31.5	20.05
BN40	13.2	26.7	14.39
BN41	15.4	31.9	24.39
BN42	16.2	33.3	20.01
BN43	13.8	26.2	22.68
BN44	14.2	29	32.08
BN45	16.1	34.9	31.60
BN46	12.9	23.4	18.54
BN47	13.6	23.5	33.96
BN48	12.2	17.1	18.80
BN49	12.2	16	20.84
BN50	11.8	16	25.01

BN=Brycinus nurse

Table 4.14 Results for Chrysichthys auratus

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
CA1	19.5	58	115.10
CA2	16.5	42.1	71.40
CA3	14.9	32.1	116.32
CA4	18.4	42.6	90.02
CA5	14.2	26	68.38
CA6	17.6	35.6	96.55
CA7	19.1	36.4	121.39
CA8	19.5	46.1	79.37

CA=Chrysichthys auratus

Table 4.15 Results for Hydrocynus sp.

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
H1	25.5	115.6	75.04
H2	31	210.7	116.31
H3	26.7	145.7	62.25
H4	31.3	242.1	61.34
H5	23.6	94.7	136.03
H6	22.3	68.2	144.11
H7	24.5	107.7	125.17
H8	22.3	75.1	208.54
H9	24.3	98.4	83.96
H10	21.1	64.1	111.55
H11	27.4	147.4	125.60
H12	24.9	109.4	130.31
H13	28	156.1	86.18
H14	20.5	58.4	148.90
H15	24.1	76.7	202.20
H16	14.2	30.7	33.77
H17	13.8	26.2	25.58
H18	13.2	22.8	18.36
H19	13.1	26.3	21.93
H20	12.5	20	22.53
H21	13.3	23.2	22.97
H22	15.9	20.3	33.02
H23	12.7	18.9	28.50

H=Hydrocynus sp.

Table 4.16 Results for Labeo coubie

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
LC1	27.8	202.9	20.86
LC2	26.5	150.7	16.32
LC3	26.8	155.7	14.28
LC4	14.7	26.7	24.93
LC5	14.7	25.2	16.51

LC=Labeo coubie

Table 4.17 Results for Synodontis oceillifer

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
SC1	20.5	76.6	32.01
SC2	12.9	17.6	57.84
SC3	12.6	13.9	51.06
SC4	10.6	8.8	68.43

SC=Synodontis oceillifer

Table 4.18 Results for Synodontis Sp

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
S1	13.9	20.5	62.16
S2	13.5	16.2	53.78
S3	11.5	8.9	86.55
S4	13.5	16.2	42.41

S=Synodontis Sp

Table 4.19 Results for Mormyrus macrophthalmus

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
MM1	13.6	14.6	56.32
MM2	12.5	12.4	62.04
MM3	11.7	9.5	45.50
MM4	11.8	10.5	60.43

MM=Mormyrus macrophthalmus

Table 4.20 Results for Mormyrus Sp

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
M1	20	58	99.57
M2	14	14.6	69.78

M=Mormyrus

Table 4.21 Results for Irvinea Voltae

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
IV1	22.1	97	104.86
IV2	14.4	23.9	105.89

IV=Irvinea Voltae

Table 4.22 Results for Barbus Sp

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
BA1	17.4	43.2	28.55

BA=Barbus Sp

Table 4.23 Results for Marcusenus abadii

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
MA1	13.6	18.7	68.73

MA=Marcusenus abadii

Table 4.24 Results for Hepsetus odoe

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
HP1	25.2	119.5	94.36

HP=Hepsetus odoe

RESULTS FROM DAM SITE

Table 4.25 Results for Schilbe mystus

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
SM1	15.7	33.73	49.94
SM2	17.7	35.23	150.38
SM3	17.3	31.25	148.09
SM4	17.9	38.05	279.44
SM5	15.4	28.4	76.37
SM6	14.9	34.88	50.50
SM7	15.5	30.99	136.08
SM8	15.7	33.79	203.67
SM9	15	26.48	69.20
SM10	16	30.8	144.56
SM11	18.2	50.02	168.57
SM12	16.5	32.23	108.68
SM13	16.9	28.8	210.91
SM14	14.5	24.03	99.72
SM15	15.5	33.47	138.87
SM16	16.8	36.12	66.05

SM=Schilbe mystus

Table 4.26 Results for Schilbe mandibularies

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
SD1	15.9	19.38	93.88
SD2	14	20.47	210.92
SD3	13.9	19.64	169.07
SD4	14.5	24.39	97.39
SD5	14	21.07	77.28
SD6	13.6	17.38	153.39
SD7	15	26.87	136.53
SD8	16.5	26.71	98.30
SD9	14.2	21.3	77.28
SD10	14.9	22.78	116.50
SD11	15.5	24.96	150.09
SD12	14.5	22.29	142.41
SD13	13.5	22.62	58.59

SD14	14.9	19.59	143.63
SD15	13	18.96	226.37
SD16	15	24.18	83.98
SD17	14	19.39	110.60
SD18	13.5	18.29	71.59
SD19	14.5	17.63	45.00
SD20	12.9	17.65	133.54

SD=Schilbe mandibularies

Table 4.27 Results for Hydrocynus forkali

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
HF1	23.2	81.4	144.37
HF2	21.2	57.5	184.57
HF3	20	44	119.48
HF4	24.2	78.8	345.89

HF=Hydrocynus forkali

Table 4.28 Results for Brycinus imberi

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
AM1	20.8	76	67.40
AM2	23.3	110.4	35.02
AM3	21	101.8	40.33
AM4	23.7	99.2	140.59

AM=Alestes imberi

Table 4.29 Results for Bagrus Sp

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
BS1	45.2	866.3	109.61

BS=Bagrus Sp

Table 4.30 Results from Heterobrunchus bidorsalis

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
HB1	17	87.4	191.04

HB=Heterobrunchus bidorsalis

Table 4.31 Results for Clarias sp

Sample code	Total length (cm)	Total weight (g)	T-Hg concentration (ng/g)
MF1	38.8	454	66.80

MF=Mud fish

Table 4. 32 Mercury concentration in Water from Bator and Dam site

Sampling Site	Month	Mercury concentration (ng/L)
Bator	October	0.0891
	November	0.0739
	December	0.0576
	January	0.0027
	February	0.0467
	March	0.0630
		Average = 0.06±0.03
Dam Site	October	0.0522
	November	0.0467
	December	0.0035
	January	0.0565
	February	0.0304
	March	0.0326
		Average = 0.04±0.02

Table 4.33 Results for Hg Concentration in sediment from Bator and Dam site at their respective sampling points

Sampling site	Months					
	October	November	December	January	February	March
Bator	27.70	60.73	51.10	92.88	126.32	102.20
	41.69	51.60	60.73	41.69	123.84	92.88
	41.69	30.66	40.49	21.93	80.97	83.38
	40.49	49.65	40.88	40.49	115.79	84.21
	48.73	52.11	60.15	47.85	97.47	93.80
	47.85	38.63	41.69	20.84	97.47	91.09
	31.58	40.49	39.72	62.53	114.64	98.38
	50.13	49.65	51.10	62.53	114.64	93.80
	51.60	63.16	52.11	51.60	91.98	291.82
	52.63	61.32	62.53	62.53	93.80	118.05
Average Hg Concentration (ng/g)	43.41	49.80	50.05	50.49	105.69	114.96
Dam site	49.65	73.68	48.73	47.42	52.63	39.72
	52.63	61.32	50.13	55.89	60.73	39.35
	51.10	61.32	50.13	51.60	52.63	59.58
	72.95	66.99	51.60	61.32	52.11	40.49
	63.16	41.69	41.69	61.32	40.88	73.68
	39.72	51.60	42.11	52.63	52.11	41.69
	51.10	49.19	52.11	62.53	0.00	39.35
	41.28	62.44	48.73	55.89	47.85	51.10
	49.65	62.53	40.88	52.63	41.28	50.61
	39.35	56.90	62.53	61.92	51.10	61.92
Average Hg Concentration (ng/g)	51.06	58.77	48.86	56.32	45.13	49.75

Table 4.34 Results for t- Test analysis of mercury concentration in Schilbe mystus from Bator and Dam site

One-Sample Statistics

	N	Mean	Std. Deviation	Std. Error Mean
Hg concentration in ng/g of fish from Bator	20	137.7600	49.11070	10.98149
Hg concentration in ng/g of fish from Dam site	16	131.3144	64.08325	16.02081

One-Sample Test

	Test Value = 5					
	t	df	Sig. (2-tailed)	Mean Difference	95% Confidence Interval of the Difference	
					Lower	Upper
Hg concentration in ng/g of fish from Bator	12.089	19	.000	132.76000	109.7755	155.7445
Hg concentration in ng/g of fish from Dam site	7.884	15	.000	126.31438	92.1668	160.4619

Table 4.35 Results for t- Test analysis of mercury concentration in sediment from Bator and Dam site

One-Sample Statistics

	N	Mean	Std. Deviation	Std. Error Mean
Hg Concentration in ng/g of Sediment from Bator	60	69.0665	40.35056	5.20924
Hg concentration in ng/g of Sediment from Dam site	60	51.6475	11.27451	1.45553

One-Sample Test

	Test Value = 0					
	t	df	Sig. (2-tailed)	Mean Difference	95% Confidence Interval of the Difference	
					Lower	Upper
Hg Concentration in ng/g of Sediment from Bator	13.258	59	.000	69.06650	58.6428	79.4902
Hg concentration in ng/g of Sediment from Dam site	35.484	59	.000	51.64750	48.7350	54.5600

Table 4.36 Results for t- Test analysis of mercury concentration in water from Bator and Dam site

One-Sample Statistics

	N	Mean	Std. Deviation	Std. Error Mean
Hg concentration in ng/L of water from Bator	6	.0550	.03017	.01232
Hg concentration in ng/L of water from Dam site	6	.0367	.02160	.00882

One-Sample Test

	Test Value = 5					
	t	df	Sig. (2-tailed)	Mean Difference	95% Confidence Interval of the Difference	
					Lower	Upper
Hg concentration in ng/L of water from Bator	-401.533	5	.000	-4.94500	-4.9767	-4.9133
Hg concentration in ng/L of water from Dam site	-562.789	5	.000	-4.96333	-4.9860	-4.9407