RIVERINE NUTRIENT INPUTS TO LAKE KIVU

BY

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Declaration

I, **Muvundja Amisi**, hereby declare that this thesis is my own findings and has never been submitted for any other award in this or any other institution of higher learning.

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Dedication

To my lovely wife Justine Balimbirire and my dear children Eloge Mwirango, Emmanuel Mungubigaba and Edith Munyiragi for their patience and understanding in favour of this work,

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List of Acronyms

APHA	:	American Public Health Association
Bio-P	:	Bio-available Phosphorus
CRSN	:	Centre des Recherches en sciences Naturelles de Lwiro
DIN	:	Dissolved Inorganic Nitrogen
DO	:	Dissolved Oxygen
D.R. Congo	:	Democratic Republic of Congo
Eawag	:	Swiss federal Institute of Aquatic Research and Technology
EPP	:	Eawag Partnership Program for Developing Countries
FAO	:	Food and Agriculture Organization of the United Nations
FUNDP	:	Facultés Universitaires Notre Dame de la Paix de Namur
GIS	:	Geographical Information System
GPS	:	Global Positioning System
INEAC	:	Institut National d'Etudes Agronomiques
ISP	:	Institut Supérieur Pédagogique
ISRIC-WISE	:	International Soil Reference and Information Centre-World Inventory of Soil Emission Potentials
NUR	:	National University of Rwanda
OVG	:	Observatoire Volcanologique de Goma
SDC	:	Swiss Agency for Development and Cooperation
SNEL	:	Société Nationale d'Electricité
SNSF	:	Swiss National science Foundation
SRP	:	Soluble Reactive Phosphorus
SRSi	:	Soluble Reactive Silica
SWAT	:	Soil and Water Assessment tool

TDS	:	Total Dissolved Solids
TP	:	Total Phosphorus
TSS	:	Total Suspended Solids
UERHA	:	Unité d’Enseignement et de Recherché en Hydrobiologie Appliquée
UV/VIS	:	Ultraviolet/Visible
WHO	:	World Health Organization

Abstract

Riverine nutrient inputs of Lake Kivu were estimated by measuring the nutrient concentrations and water discharges of 21 rivers (13 on the Congolese side and 8 on the Rwandan side of the lake) in five sub-basins representing 1550 km^2 (30% of the river active area). The Congolese river basin was sampled by me and the Rwandan side by my colleague Jean-Népomuscène Namugize of the National University of Rwanda under the same project. Data on Water quality in the sampled rivers in relation to physico-chemical and nutrient concentrations are presented.

Mean values ranging from 6.4 to 7.4 for pH and 144 to 638 $\mu S.cm^{-1}$ for Electrical conductivity were measured in the sampled rivers. Mean temperatures varied from 18.0 to 23.1°C. Discharge and total suspended solids from 0.21 to 7.30 $m.s^{-1}$ and 2 to 561 $mg.L^{-1}$ respectively. The catchment sizes of Congolese river basins of the lake were comprised between 11 and 296 km^2. Cropland-dryland was the first most dominant land use in the catchment (57%) and the second was Evergreen broadleaf forest (25%). Shrubland occupied 16% and Cropland-Woodland mosaic 1.4%.

The inputs of soluble reactive phosphorus (SRP), dissolved inorganic nitrogen (DIN) and soluble reactive silica (SRSi) are compared to other nutrient sources (through atmospheric deposition and internal loading) into the epilimnion.

The input of SRP adds up to 111 t yr^{-1}, which was 15 times less than total phosphorus (TP). SRP input represents 48% of the external nutrient fluxes (230 t P yr^{-1}). Total phosphorus inputs from rivers and all external inputs were 1,650 and 4,600 t P yr^{-1} respectively.

Dissolved inorganic nitrogen input consisted mainly of ammonia (370 t N yr^{-1}) and nitrate (1,550 t N yr^{-1}) in river inputs but 2,220 and 1,230 for overall external inputs respectively.

A positive dependence was found between riverine phosphorus loads and (1) human population in the catchment and (2) the catchment size.

Erosion and land degradation caused by agricultural activities leading to deforestation and pollution caused by human and domestic wastes in urban areas can be considered as the current major constraints of water quality in Lake Kivu river basin.

Both ammonia and nitrate contributed to 35 % of the external nitrogen inputs to the epilimnion. Dissolved inorganic nitrogen (DIN) and total phosphorus (TP) loads were estimated to 1,920 in the riverine inputs and 5,400 t N yr^{-1} in the overall external inputs. For total phosphorus, estimates are 1,650 and 4,600 t P yr^{-1} in river fluxes and overall external fluxes respectively. Lake Kivu tributaries constitute the most important supplier of dissolved silica (23,300 t Si yr^{-1}) to the epilimnion (95% of the external sources and 43% of both external and internal sources). Rivers load silica at the rate of 23,300 t Si yr^{-1} and the overall external sources at 24,600 t yr^{-1}.

Considering phosphorus as the production limiting nutrient, it is estimated that the nutrient inputs of 111 t P yr^{-1} from rivers were likely to induce a new primary production of 5 g C m^{-2} yr^{-1} which represents ~2% of the overall production.

CHAPTER ONE: INTRODUCTION

1.1 Background

Lake Kivu is one of the African great lakes of the Albertine Rift. It is a transboundary lake situated between 1°34’30’’and 2°30’ S and between 28°50’ and 29°23’ E on the border between the Democratic Republic of Congo and Rwanda. The Ruzizi River is the outflow (~3.6 km^3 yr^{-1}) which feeds Lake Tanganyika from Lake Kivu. Lake Kivu is different from other African lakes by its volcanic origin, altitude, morphology and its strong and permanent stratification due to its water physico-chemical properties (Damas 1937; Degens et al. 1973). It is located at the highest altitude (1463 m) compared to other African rift lakes and lies between two mountain chains: the Mitumba Mountain in the west and the Rwandan dorsal in the eastern part. Lake Kivu is believed to have been strongly affected by the Virunga volcanoes, whose Pleistocene lavas dammed its former outflow to the Nile and thus formed its current only one outflow river, the River Ruzizi which links it to Lake Tanganyika (Haberyan and Hecky 1987). The geology of the region is dominated by rocks of Precambrian aeon (Schist, granulites and mica schist) and Cenozoic aeon (volcanic rocks, lateritic and recent alluvium) (Samir et al. 1981).

Damas (1937) distinguished 5 great basins and the bay of Kabuno-Kashanga. Hills surrounding the lake are composed of an old geomorphologic structure which is currently undergoing a serious change due to soil erosion, landslides and seismicity (Mavonga 2007). Most of their alluviums are deposited in the lake. Since small occasional rivers flow into the lake during the wet season throughout the town of Bukavu; it creates an unusual hydrography (Samir et al. 1981). Subaquatic water springs are an important contributor to the lake’s water budget (Degens et al. 1973; Schmid et al. 2005; Bergonzini 1998).

The climate of the Kivu basin is tropical with a long wet season from September to June and a dry season from July to August. The rainiest month is April with ~200 mm of precipitation on the Lake surface and ~190 mm on its watershed (Bergonzini 1998). There is a short dry season in February with precipitations of 130-140 mm per month. In the dry season, precipitations decrease to 27 mm on the Lake Surface and 25 mm on its watershed (Bergonzini 1998) and strong monsoon wind blows from the South towards the North mixing the surface water in the epilimnion down to 40-60 m. The total rainfall is estimated between 1300-1500 mm per year (Bergonzini 1998). The hydrology of Lake Kivu is mainly led by the precipitation given that the catchment area is formed by 127 small mountainous tributaries (Bergonzini 1998; Marlier 1954).

Like other African great lakes, Lake Kivu is considered vulnerable to human activities (Bootsma and Hecky 1993), similar to Lake Victoria, where agriculture, deforestation, and urbanization caused an increase in nutrient inputs and eutrophication (Hecky 1993). For Lake Malawi, Hecky et al. (2003) found that the nutrient loading might have increased by 50% due to agricultural development and growing population.

Land use is dominated by subsistence agriculture and farming, implying that only manure is used for fertilization and hardly any use for agro-chemicals. Nutrient load from mining is limited, as only the extractions of Colombo-tantalite and cassiterite in the Kalehe and Goma regions have substantially increased in the last few years (Martineau 2003). Industrial activities in the catchment comprise only two breweries and processing of some agricultural products (such as tea, coffee, quinine). Deforestation in the catchment is a major concern, given the increasing need for firewood and the lack of alternative energy for households. This region has been experiencing large population movements, refugees or displaced people, for more than a decade to now, due to political instability in the region leading to forest clearing. In Rwanda, annual deforestation rates of up to 4% yr^{-1} have been reported (Jones 2003). As a result of deforestation, soil erosion and landslides (Moeyersons et al. 2004), and the lack of sewage treatment plants, nutrient fluxes into the waterbodies might have increased. In areas dominated by agricultural land use, rivers are often visibly “brownish” turbid.

Compared to the biodiversity of other East African lakes, the fauna, especially ichthyologic community, in Lake Kivu is poor, probably because of gas outburst from volcanic activity during the late Pleistocene and Holocene and also because it is still a young lake (~15,000 years; Haberyan and Hecky 1987). Lake Kivu supports obviously the poorest ichtyological fauna compared to other lakes of the region with only 29 ichthyologic species (Snoecks et al. 1997; Froese et al. 2008). One of the 29 ichthyologic species and the most dominant, the *Limnothrissa miodon*, is a Lake Tanganyika Sardine which was introduced in this lake in 1959 to fill an empty niche made by the pelagic zone and enhance food supply to the riparian populations.

Lake Kivu has been classified as a “sodium-potassium-magnesium-bicarbonate” lake (Damas, 1937). A Large amount of organic material is definitely deposited by sedimentation from the oxic zone to the hypolimnion in the deep waters (Botz et al. 1988) where it is degraded into methane by methanogenic bacteria (Schoell et al. 1988). The surface water temperature is surprisingly constant with an average of 23.5°C (Sarmento et al. 2006). Lake Kivu experiences several temperature gradients according to which, below 100 m depth, the temperature increases with depth due to sub-aquatic flows (Degens et al. 1973). This volcanic lake is permanently stratified without oxygen in its deeper waters (Damas 1937; Lorke et al. 2004).

A comparison of chemical analyses made by Schmid et al.(2005) to those found by Degens et al. (1973) showed that silica concentrations have significantly decreased (from 230 to130 μmol l^{-1}) in surface waters but have increased (from 1050 to 1400 μmol l^{-1} i.e. 30-

35%) at 350 m depth within 30 years. Significant increase in P concentrations and in the ratio of Ca: Na was reported by Schmid et al. (2005) in the deep waters, which indicate that more sedimentation has occurred.

The recent increase in methane formation within the lake (Schmid et al. 2005) can be assessed as a function of nutrient availability. Nutrient cycling in the lake may be governed either by an increasing nutrient load due to fast growing human activities in the watershed and/or the high rate of sedimentation. Increased sedimentation rates are caused by algal blooms due to the decline in the zooplankton biomass, and the disappearance of the most efficient phytoplankton grazer species (*Daphnia curvirostris*) (Isumbisho et al. 2004). The decline of the zooplankton biomass resulted from the introduction of the Tanganyika sardine, *Limnothrissa miodon* (Dumont 1986; Villanueva et al. 2008). The growing population in the catchment may also lead to physical changes resulting into a larger nutrient input to rivers and runoff due to environmental degradation by agriculture and farming, deforestation and biomass burning, erosion, etc. (Bugenyi and Magumba 1996; Hecky et al. 2003). Moreover industrial and domestic waste is directly loaded to the lake without any prior treatment. Bootsma and Hecky (1993) noticed a growing concern about the risk of African Great Lakes to be disturbed from the activities of the rapidly increasing human populations in their watersheds. Thus, Hecky et al. (2003) found that increasing populations and associated agricultural development might have increased nutrient loading to Lake Malawi by 50% as forests had been increasingly cleared for agriculture.

According to Hecky et al. (1996), the most important question concerning sediment formation is what controls the primary production and the algal species composition of the phytoplankton communities of African Great Lakes as Lake Kivu. The organic sediments of the African Greats Lakes and the aquatic microfossil record they contain owe their origins to the nutrient fluxes entering these lakes and to the internal cycling of nutrients in these ecosystems (Hecky et al. 1996). According to them, understanding and managing the productivity of these ecosystems and interpreting their fossil records require quantification and modelling of the external and internal processes which determine nutrient availability for algal growth and fish production.

In Lake Kivu, methane is produced by bacterial decomposition of organic material such as dead algae. The small oxic zone in the lake (~40 m, Isumbisho et al. 2006) contributes to the large loss of nutrients by sedimentation. The higher sedimentation rate enhances the availability of organic matters which are degraded by microorganisms to produce more methane in the water/sediment interface (Schoell et al.1988; Tietze et al.1980).

Limnologists and oceanographers are both concerned with relationship of nutrient (carbone, C, Nitrogen, N and Phosphorus, P) and biological production (Guildford and Hecky 2000). Marine and freshwater studies have emphasized inorganic nutrient concentrations for modelling phytoplankton growth (Kilham and Hecky 1988) and for tracing geochemical fluxes of the nutrient fluxes to the productive surface waters (Smith et al. 1986). Healy (1975) discussed the nutrient deficiency of algal production. Their findings indicated the following degrees of nutrient deficiency regarding to their elemental molar ratios: N extreme deficiency if $C:N > 14.6$ and P extreme deficiency if $C:P = 258$; N moderate deficiency if $8.3 < C:N < 14.6$ and $129 < C:P < 258$ for P moderate deficiency whereas No deficiency was found when $C:N < 8.3$ and $C:P < 129$ for N and P respectively. In a study on nutrient limitation of the productivity of freshwater systems (Guildford and Hecky 2000), P control of algal biomass (as indicated by Chla) and algal growth rate (as indicated by nutrient status) were evident in most of them, especially when P was less than $0.5 \mu mol.L^{-1}$. In terms of TN:TP molar ratios, Guildford and Hecky (2000) observed a N-deficiency for $TN:TP > 20$ and P-deficient growth at $TN:TP > 50$ while for intermediate values, either N or P could become deficient. Although silica is most abundant nutrient, some studies have emphasized also the possibility of silica limitation for diatom growth in some lakes during rare periods such as Lakes Albert, Malawi and Tanganyika which experience some periods of extremely low silica concentrations (Kilham et al. 1986 and references therein).

1.2. Research problem statement

Only a few studies have been done on Lake Kivu and its tributaries, and available are not sufficient to allow its exploitation (Kling et al. 2006). River inflows of this lake have limited hydrobiological resource importance due to their small size, morphological aspects and the poor ichthyologic fauna (Marlier 1954). Therefore, they attracted very few researchers and consequently knowledge on them remains poor despite their overall hydrological importance to the water balance of the lake (~50% of the total water input of the lake, Bergonzini 1998). There is a lack of knowledge about their water quality and their contribution to the chemistry of the lake for a sustainable management of the Kivu system.

This study which was conducted under the project: “Nutrient cycling and methane production in Lake Kivu” and funded by the Swiss National Science Foundation (SNSF), was one of a series of studies which aimed at quantifying the nutrient inputs from external and internal nutrient sources and their contribution to the nutrient balance of the lake in order to understand the causes of the recent increase in methane concentrations within the lake. Data collected under this project on the Congolese side (13 rivers) were combined with that collected in the Rwandan side (8 rivers) by Namugize (2009) to provide more representative estimates of nutrient inputs. In order to overcome the uncertainties yielded by the extrapolation of the nutrient inputs from 21 rivers to the entire lake with 127 rivers, data provided by a similar study (Rinta, 2009) were also used, which aimed at quantifying the riverine nutrient inputs using a modelling approach, and compared it to our own findings.

The sampled rivers were selected regarding their catchment size, the related land use and the geographical location to be representative of the whole catchment features and patterns.

1.3. Objectives

1.3.1 General objective

The purpose of this study on Lake Kivu is to assess the nutrient fluxes from the Lake tributaries.

1.3.2 Specific objectives

The specific objectives of this study are:

(i) To determine the physico-chemical and nutrient concentrations in the 13 selected rivers;

(ii) To estimate the nutrient inputs of the river basin to Lake Kivu and evaluate its relative contribution to the lake nutrient budget;

(iii) To evaluate the contribution of riverine nutrient inputs to the current primary productivity of the lake.

1.4 Research questions

1. What is the current state of Water quality of the Kivu tributaries?
2. How much does the riverine nutrient loading to Lake Kivu contribute to the nutrient budget of the lake?
3. Do nutrient loads from rivers to the lake dependent on river catchment and land use?

1.5 Justification

This study will lead to the establishment of the relative contribution to the riverine inputs into the nutrient balance of the lake and consequently to primary production. It will also provide some information on the current water quality of the Lake Kivu tributaries.

Data provided by this study together with previous and ongoing studies will be useful to predict scenarios for further development of Lake Kivu methane reservoir as well as fisheries. The results will also be helpful in predicting the different strategies to limit eutrophication for sustainable lake management.

CHAPTER TWO: LITERATURE REVIEW

2.0 Introduction

The fast growing populations especially in developing countries has led to higher demand and depletion of natural resources. This global problem has many environmental consequences such as deforestation, overgrazing of grassland, overcultivation, erosion, siltation and salinization, waste discharge due to unplanned urbanization, etc. All of these anthropogenic activities are leading to poor water quality in the catchment. Such problems have been experienced abroad by several lakes such as Great Lakes of Northern America: Superior, Michigan, Huron, Erie and Ontario (EPA, 2008), Lake Léman, and in this region by Lake Victoria (Hecky and Bugenyi 1992, Hecky 1993) and Lake Malawi (Hecky et al. 2003).

Lake Kivu river basin has also experienced a fast growing population since the 1960s leading to some physical changes due to increasing need of new agricultural lands, deforestation and urbanization (Muvundja et al. 2009). These changes must have some impacts on water quality by enhancing the water turbidity, nutrient loads, etc. as well as living organisms.

2.1. Water pollution sources

Current global water quality issues are mainly due to pollution caused by environmental impacts of human activities. Sources of pollutants are either point sources such as industrial and sewage or non point such agro-chemical diffuse. These pollutants are released either directly to the waterbody or via land points and transported to the waterbody by runoff and groundwater. Gas pollutants released to the atmosphere are also deposited by dry and wet atmospheric deposition processes. Direct discharges to waterways are known as point sources. Because such sources have specific owners and can be easily sampled, regulatory programs have resulted in a high degree of control. Non-point sources include urban and agricultural runoff, airborne deposition of pollutants from automobiles and commercial activities, and contaminated sediments and contaminated groundwater. Control of non-point sources is made difficult by their diffuse nature, episodic release and lack of institutional arrangements to support their control (EPA 2008). However, in some regions, natural outburst of unstable environments such as volcanoes or "killer lakes", such as the Kivu region, can also constitute an important pollutant source.

Water issues are not only a matter of quantity or availability but also a matter of quality. Poor quality water is recognized as one of the causes of aquatic biodiversity loss due to habitat degradation and low standards in abiotic factors, waterborne diseases, unaesthetic conditions leading to lack of benefit from it as resource providing environmental and socio-economic services. Odada et al. (2004) argued that for Lake Victoria the most important pollution issues are microbiological, eutrophication, chemical, and suspended solids.

2.2. Water pollution categories

Water pollution can be categorized into microbiological, chemical and eutrophication (Odada et al. 2004). In Lake Victoria basin four immediate causes were identified in microbiological pollution, namely, municipal untreated sewage, runoff, and storm water, animal waste, and maritime transport waste (Odada et al. 2004). Of these immediate causes, the two most important are municipal untreated sewage, and runoff, and storm water. Municipal untreated sewage, runoff, and storm water: Direct discharge of municipal untreated effluent into rivers and the lake directly contribute to microbiological pollution. These have contributed to the degradation of river and lake-water quality for habitats and drinking water use (Ntiba et al. 2001, Wadinga et al. 1987). The low standards of health in the region are caused by a general lack of awareness of good hygiene practices, direct contamination of beach waters through bathing and washing, and uncontrolled waste disposal around the shore line (Karanja 2002). Reduction of the biological oxygen demand (BOD) load of such effluent can significantly reduce the occurrence of waterborne diseases such as typhoid and cholera which are common in the region (Odada et al. 2004 and references therein). Runoff and storm water collect a lot of animal, plant, and human waste from point and non-point sources and channel these to rivers and the lake, creating an environment that supports microbiological pathogens. Harbor and bilge discharges compound the microbiological pollution problem.

Odada et al. (2004) identified the main causes of chemical pollution of Lake Victoria. These are enhanced effluent discharge, enhanced discharge of solids, runoff and storm water, and atmospheric deposition. The main sources of chemical pollutants are wastes from agricultural activities (manure and agrochemicals such as fertilizers and pesticides), industrial, domestic and mining activities.

2.3. Nutrient cycling in aquatic systems

Biogeochemical cycles of nutrient elements include the environmental processes which support and sustain life in all ecosystems, and therefore determine the well functioning as well as good environmental health of an aquatic ecosystem.

2.4.1. Nitrogen cycle

Nitrogen is essential for living organisms as an important constituent of proteins, including genetic material. Plants and micro-organisms convert inorganic nitrogen to organic forms. In the environment, inorganic nitrogen occurs in a range of oxidation states as nitrate (NO_3^-) and nitrite (NO_2^-); the ammonium ion (NH_4^+) and molecular nitrogen (N_2) (Bartram and Ballance 1996). It undergoes biological and non-biological transformations in the environment as part of the nitrogen cycle. The major non-biological processes involve phase transformations such as volatilisation, sorption and sedimentation. The biological transformations consist of: (a) assimilation of inorganic forms (ammonia and nitrate) by plants and micro-organisms to form organic nitrogen e.g. amino acids, (b) reduction of nitrogen gas to ammonia and organic nitrogen by micro-organisms, (c) complex heterotrophic conversions from one organism to another, (d) oxidation of ammonia to nitrate and nitrite (nitrification), (e) ammonification of organic nitrogen to produce ammonia during the decomposition of organic matter, and (f) bacterial reduction of nitrate to nitrous oxide (N_2O) and molecular nitrogen (N_2) under anoxic conditions (denitrification).

Nitrogen in all inorganic forms and dissolved organic one are easily assimilated by all phytoplankton species to stimulate algal growth whereas a few number of species only including blue-green algae are susceptible of N_2- fixation (Hecky 1993). In some conditions, Nitrogen can become a limiting nutrient for primary productivity (Guildford and Hecky 2000).

2.4.2. Phosphorus cycle

Phosphorus is an essential nutrient for living organisms and exists in water bodies as both dissolved and particulate species. It is generally the limiting nutrient for algal growth and, therefore, controls the primary productivity of a water body (Hecky et al. 1996). Artificial increases in concentrations due to human activities are the principal cause of eutrophication. In natural waters and in wastewaters, phosphorus occurs mostly as dissolved orthophosphates and polyphosphates, and organically bound phosphates. Changes between these forms occur continuously due to decomposition and synthesis of organically bound forms and oxidised inorganic forms (Gächter et al. 2004).

Natural sources of phosphorus are mainly the weathering of phosphorus-bearing rocks and the decomposition of organic matter (Gächter et al. 2004). Domestic waste-waters (particularly those containing detergents), industrial effluents and fertiliser run-off contribute to elevated levels in surface waters. Phosphorus associated with organic and mineral constituents of sediments in water bodies can also be mobilised by bacteria and released to the water column. In most of freshwaters, particularly during mixing periods, Phosphorus constitutes the limiting nutrient for algal growth (Guildford and Hecky 2000, Kilham and Kilham 1990).

2.4.3. Silica cycle

Silica is widespread and always present in surface and groundwaters. It exists in water in dissolved, suspended and colloidal states. Dissolved forms are represented mostly by silicic acid, products of its dissociation and association, and organosilicon compounds. Reactive silicon (principally silicic acid but usually recorded as dissolved silica, SiO_2) or sometimes as silicate (H_4SiO_4) mainly arises from chemical weathering of siliceous minerals (Bartram and Ballance 1996).

Silica may be discharged into water bodies with wastewaters from industries using siliceous compounds in their processes such as potteries, glass works and abrasive manufacture. Silica is also an essential element for certain aquatic plants, principally diatoms (Hecky et al. 1996). It is taken up during cell growth and released during decomposition and decay giving rise to seasonal fluctuations in concentrations, particularly in lakes (Bugenyi and Magumba 1996; Hecky et al. 1996).

2.5. Eutrophication

2.5.1. Origin and Manifestations

Lakes can be characterized by their biological productivity, that is, the amount of living material supported within them, primarily in the form of algae. The least productive lakes are called oligotrophic; those with intermediate productivity are mesotrophic; and the most productive are eutrophic. The variables that determine productivity are temperature, light, depth and volume, and the amount of nutrients received from the environment (EPA 2008).

In Nothern America, the Great Lakes were 'oligotrophic' before European settlement and industrialization except in shallow bays and shoreline marshes (EPA 2008). Their size, depth and the climate kept them continuously cool and clear. The lakes received small amounts of fertilizers such as phosphorus and nitrogen from decomposing organic material in runoff from forested lands. Small amounts of nitrogen and phosphorus also came from the atmosphere. Nowadays, these conditions have changed. Temperatures of many tributaries have been increased by removal of vegetative shade cover and some by thermal pollution. But, more importantly, the amount of nutrients and organic material entering the lakes has increased with intensified urbanization and agriculture. Nutrient loading increased with the advent of phosphate detergents and inorganic fertilizers. Although controlled in most jurisdictions bordering the Great Lakes, phosphates in detergents continue to be a problem where they are not regulated (EPA 2008).

Before controls could be developed, it was necessary to determine which nutrients were most important in causing eutrophication in previously mesotrophic or oligotrophic waters (EPA 2008). By the late 1960s, the scientific consensus was that phosphorus was the key nutrient in the Great Lakes and that controlling the input of phosphorus could reduce eutrophication (EPA 2008., Hecky and Kilham 1988). Lakes are susceptible to depletion of oxygen in waters near the bottom because they stratify in summer, forming a relatively thin layer of cool water, the hypolimnion, which is isolated from oxygen-rich surface waters. Oxygen is rapidly depleted from this thin layer as a result of decomposition of organic matter. When dissolved oxygen levels reach zero, the waters are considered to be anoxic. With anoxia, many chemical processes change and previously oxidized pollutants may be altered to forms that are more readily available for uptake by the water. By contrast, in case of strong winds, lakes especially shallow lakes, do not stratify and generally are not susceptible to anoxia because the wind keeps the shallow basin well mixed, preventing complete stratification.

In Lake Erie during eutrophic conditions Cladophora, a filamentous alga that thrives under eutrophic conditions, became the dominant nearshore species covering beaches in green, slimy, rotting masses (EPA 2008). In eutrophic Lake Victoria, blue-green algae (Cyanobacteria) are prolific leading to toxic and unhygienic waters (Gorham and Carmichael 1980, Helmer and Hespanhol 1997). Increased turbidity caused the lake to appear greenish-brown and murky (Hecky 1993).

2.5.2. Effects of pollution on living organisms

Increased nutrients in the lakes stimulate the growth of green plants, including algae (EPA 2008; Hecky 1993). The amount of plant growth increases rapidly in the same way that applying lawn fertilizers (nitrogen, phosphorus and potassium) results in rapid, green growth and proliferation of toxic algae (Gorham and Carmichael 1980). In the aquatic system the increased plant life eventually dies, settles to the bottom and decomposes. During decomposition, the organisms that break down the plants use up oxygen dissolved in the water near the bottom and lead to more acidic conditions. With more growth there is more material to be decomposed, and more consumption of oxygen. Under normal conditions, when nutrient loadings are low, dissolved oxygen levels are maintained by the diffusion of oxygen into water, mixing by currents and wave action, and by the oxygen production of photosynthesizing plants.

Depletion of oxygen through decomposition of organic material is known as biochemical oxygen demand (BOD), which is generated from two different sources. In tributaries and harbors oxygen depletion is often caused by materials contained in the discharges from treatment plants (Gächter et al., 2004; Schärer et al. 2006). The other principal cause is decaying algae. In large embayments and open lake areas such as the central basin of Lake Erie, algal BOD is the primary problem. As the BOD load increases and as oxygen levels drop, certain species of fish can be killed and pollution-tolerant species that require less oxygen, such as sludge worms and carp, replace the original species. Changes in species of algae, bottom-dwelling organisms (or benthos) and fish are therefore biological indicators of oxygen depletion (EPA 2008).

Turbidity in the water as well as an increase in chlorophyll also accompany accelerated algal growth and indicate increased eutrophication (Hecky 1993). Lake Erie was the first of the Great Lakes to demonstrate a serious problem of eutrophication because it is the shallowest, warmest and naturally most productive (EPA 2008). Lake Erie also experienced early and intense development of its lands for agricultural and urban uses. About one-third of the total Great Lakes basin population lives within its drainage area and surpasses all other lakes in the receipt of effluent from sewage treatment plants. Oxygen depletion in the shallow central basin of Lake Erie was first reported in the late 1920s. Studies showed that the area of oxygen depletion grew larger with time, although the extent varied from year to year owing, at least in part, to weather conditions. Eutrophication was believed to be the primary cause (EPA 2008).

Nutrient-enriched natural waters also experience the proliferation of aquatic weeds or macrophytes such as water hyacinth in the case of Lake Victoria (Ntiba et al. 2001, Njiru et al. 2008). These weeds invaded the lake under eutrophic conditions causing more environmental stress to aquatic animals (fish hybridization, oxygen depletion, light limitation to algal productivity, etc.) as well as on navigation and fishing activities (Njiru et al. 2008).

Toxic pollutants include human-made organic chemicals and heavy metals that can be acutely toxic in relatively small amounts and injurious through long-term (chronic) exposure in minute concentrations. Many of the contaminants that are present in the environment have the potential to increase the risk of cancer, birth defects and genetic mutations through long-term, low-level exposure (Gorham and Carmichael 1980). Many toxic substances tend to bioaccumulate as they pass up the food chain in the aquatic ecosystem. While the concentrations in water of chemicals such as PCBs may be so low that they are almost undetectable, biomagnification through the food chain can increase levels in predator fish such as large trout and salmon by a million times (EPA 2008). Still further biomagnification occurs in birds and other animals that eat fish. There is little doubt that bioaccumulative toxic substances continue to affect aquatic organisms in the lakes and birds and animals that eat them. Public health and environmental agencies in the Great Lakes states and the Province of Ontario warn against human consumption of certain fish. Some fish cannot be sold commercially because of high levels of PCBs, mercury or other substances (EPA 2008).

Inadequate water supply and sanitation are largely responsible for the high levels of waterborne diseases in Africa. More than 75% of people in DR Congo lives in rural areas and do not have appropriate sanitation systems. Not surprisingly, infectious waterborne diseases such as diarrhoea diseases, dysentery, cholera and hepatitis are almost endemic in rural areas. Most of the faeces are deposited on land, providing an easy pathway for pathogens to enter surface and groundwater, and hence to the local population via contaminated drinking water. Bacteria are naturally occurring in water. However, the presence of faecal coliform bacteria indicates the presence of human or animal wastes.

If a dangerous pathogen, such as *Vibrio cholerae*, is introduced into a community with poor sanitation, poor water supply and poor food safety, an epidemic may ensue. Cholera and typhoid are now endemic within the riparian population of the Kivu region.

2.5.3. Mitigation measures

In response to public concern of Northern America Great lakes eutrophication, new pollution control laws were adopted in both countries USA and Canada to deal with water quality problems, including the reduction of phosphorus loadings to the lakes. In 1972, Canada and the United States signed the Great Lakes Water Quality Agreement to begin a binational Great Lakes cleanup that emphasized the reduction of phosphorus entering the lakes (EPA 2008). Studies were conducted to determine the maximum concentrations of phosphorus that could be tolerated by the lakes without producing nuisance conditions or disturbing the integrity of the aquatic community. Mathematical models were then developed to predict the maximum annual loads of phosphorus that could be assimilated by the lakes without exceeding the desired phosphorus concentrations (Helmer and Hespanhol 1997). These maximum amounts were then included in the Great Lakes Water Quality Agreement (EPA 2008). Following the progress made through waste treatment and detergent phosphate controls, it has been determined that control of phosphorus from land runoff is also necessary in controlling eutrophication. If a high degree of control of point sources is attained through regulations and adequate policies, and target levels can be met through additional progress in voluntary control of nonpoint sources (EPA 2008). Therefore, the control of phosphorus and associated eutrophication in the Great Lakes represents an unprecedented success in producing environmental results through international cooperation. The return to lower amounts of phosphorus do not only result in reducing excess growth of algae, but has also changes the composition of the algal population. Nuisance algal species might give way to more desirable and historically prevalent species, such as diatoms, thereby eliminating nuisance conditions and improving the quality of the food chain for other organisms (EPA 2008).

Other means of lake restoration include wetland restoration, reforestation of the catchment, sustainable use of agrochemicals, sewage treatment and waste reduction through civic education and public awareness as well as policy and regulations reinforcement, etc.(Hecky and Bugenyi 1992, Odada et al. 2004, Helmer and Hespanhol 1997) to avoid a sort of “tragedy of the commons”.

2.6. Siltation and Salinization

There is an increasing concern about the silt and salinity trends of many water bodies due to erosion, agro-chemical waste inputs, disposal of industrial and domestic wastes in effluents, mining activities, etc (Hecky and Bugenyi 1992). Freshwaters loose their properties when they receive much silt and salts (minerals) by pollution and freshwater fishes and other living organisms can be seriously affected. These minerals are mainly formed of soluble salts to form ions in water such as chloride, sulfate, sodium, potassium, etc. Silt may also deposit onto the water in form of dust from eroded and degraded soils.

Increases in the salinity of inland waters are an emerging problem in southern Africa where certain human activities are increasing the total dissolved solids content of water bodies. Irrigation is a major culprit, and the Vaal-Hartz irrigation scheme in South Africa is responsible for a three-fold increase in the concentration of total dissolved solids in the Hartz River (SARDC 1998 and references therein). Disposal of domestic and industrial effluent has led to Salinization of some water bodies. The levels of solids in Lake Chivero have risen from 7 mg/l in the 1950s to 37 mg/l in 1991 because of sewage effluent discharged into the lake (SARDC 1998 and references therein).

Siltation and salinization are also sources of sediment contamination and water turbidity leading to unfair conditions for light availability in the water column. Lower light penetration and transparencies affect the primary productivity of the water body and the life style of aquatic animals. Some species can be vulnerable to predation or predators can miss their prey, etc. Other species such as tilapiines can mate irregularly and lead to extinction due to hybridization (Njiru et al. 2008).

2.7. Water quality assessment and standards

2.7.1. Determination of Hydrological variables

Determining the hydrological regime of a water body is an important aspect of water quality assessment. Discharge measurements, for example, are necessary for mass flow or mass balance calculations and as inputs for water quality models.

(i) Velocity

The velocity (sometimes referred to as the flow rate) of a water body can significantly affect its ability to assimilate and transport pollutants. The measurement of velocity is extremely important in any assessment programme. It enables the prediction of the movement of materials (particularly pollutants) within water bodies. For example, knowledge of water velocity enables the prediction of the time of arrival downstream, of a contaminant accidentally discharged upstream. Water velocity can vary within a day, as well as from day to day and season to season, depending on hydrometeorological influences and the nature of the catchment area (Bartram and Ballance 1996).

(ii) Discharge

The discharge as an important hydrological parameter which helps to estimate the rate at which the materials are transported by the river, and consequently determining their fluxes. The amount of suspended and dissolved matter in a water body depends on the discharge and is a product of the concentration and the discharge (Moosman et al. 2005). Natural substances arising from erosion (suspended matter) normally increase in concentration rapidly with increased discharge (Moosman et al. 2005). Substances introduced artificially into a water body, such as trace elements and organic matters; tend to occur at decreasing concentrations with increasing river discharge (Moosman et al. 2005). If a pollutant is introduced into a river at a constant rate, the concentration in the receiving water can be estimated from the quantity input divided by the river discharge but sedimentation and re-suspension can, however, affect this simple relationship (Chapman and Kimstach 1996). Discharge can be estimated from the product of the velocity and the cross-sectional area of the river (Harrelson et al. 1994). It should be measured at the time of sampling and preferably at the same position as water samples are taken. As cross-sectional area varies with different discharges, a series of cross-section area measurements are needed in relation to the different discharges (Bartram and Ballance 1996; Harrelson et al. 1994). Measurements of depth across the water body can be used to obtain an approximate cross-sectional area. Specific methods for calculating discharge are available in Bartram and Ballance (1996).

2.7.2 Physico-chemical parameters

The main physico-chemical features of a water body are temperature, suspended solids conductivity, dissolved oxygen and pH.

(i) Temperature

Water bodies undergo temperature variations along with normal climatic fluctuations. These variations occur on a daily and seasonally basis. The temperature of surface waters is influenced by latitude, altitude, and season, time of day, air circulation, cloud cover and the flow and depth of the water body (Chapman and Kimstach 1996). In turn, temperature affects physical, chemical and biological processes in water bodies and, therefore, the concentration of many variables. As water temperature increases, the rate of chemical reactions generally increases together with the evaporation and volatilisation of substances from the water. Increased temperature also decreases the solubility of gases in water, such as O_2, CO_2, N_2, CH_4 and others. The metabolic rate of aquatic organisms is also related to temperature, and in warm waters, respiration rates increase leading to increased oxygen consumption and increased decomposition of organic matter. Growth rates also increase (this is most noticeable for bacteria and phytoplankton which double their populations in very short time periods) leading to increased water turbidity, macrophyte growth and algal blooms, when nutrient conditions are suitable (Bartram and Ballance 1996). Temperatures fluctuate seasonally with minima occurring during wet periods, and maxima during dry seasons, particularly in shallow waters (Chapman and Kimstach 1996).

(ii) Total suspended solids

Total suspended solid (TSS) or sesidue content of a water body play an important ecological role as it affects the water transparency and light limitation. Higher concentrations of TSS in rivers indicate water turbidity and sometimes high salinity depending on the geological features of the river basin rocks due to higher soil erosion or waste deposition. Therefore, siltation and salinization can match with higher total suspended solids.

(iii) Conductivity

Conductivity, or specific conductance, is sensitive to variations in dissolved solids, mostly mineral salts. The degree to which these dissociate into ions, the amount of electrical charge on each ion, ion mobility and the temperature of the solution all have an influence on conductivity (Chapman and Kimstach 1996). The conductivity of most freshwaters ranges from 10 to 1,000 $\mu S\ cm^{-1}$ but may exceed 1,000 $\mu S\ cm^{-1}$, especially in polluted waters or those receiving large quantities of land run-off (Bartram and Ballance 1996). In addition to being a rough indicator of mineral content when other methods cannot easily be used, conductivity can be measured to establish a pollution zone, e.g. around an effluent discharge, or the extent of influence of run-off waters. It is usually measured *in situ* with a conductivity meter, and may be continuously measured and recorded. Such continuous measurements are particularly useful in rivers for the management of temporal variations in TDS and major ions.

(iv) pH

The pH is an important variable in water quality assessment as it influences many biological and chemical processes within a water body and all processes associated with water supply and treatment. When measuring the effects of an effluent discharge, it can be used to help determine the extent of the effluent plume in the water body (Chapman and Kimstach 1996). The natural acid-base balance of a water body can be affected by industrial effluents and atmospheric deposition of acid-forming substances. Changes in pH can indicate the presence of certain effluents, particularly when continuously measured and recorded, together with the conductivity of a water body (Bartram and Ballance 1996). Diel variations in pH can be caused by the photosynthesis and respiration cycles of algae in eutrophic waters. The pH of most natural waters is between 6.0 and 8.5, although lower values can occur in dilute waters high in organic content, and higher values in eutrophic waters, groundwater brines and salt lakes (Chapman and Kimstach 1996). Ideally, pH should be determined *in situ*, or immediately after the sample is taken, as so many natural factors can influence it.

(v) Dissolved oxygen

Oxygen is essential to all forms of aquatic life, including those organisms responsible for the self-purification processes in natural waters. The oxygen content of natural waters varies with temperature, salinity, turbulence, the photosynthetic activity of algae and plants, and atmospheric pressure. The solubility of oxygen decreases as temperature and salinity increase. In freshwaters the solubility of oxygen (DO) at sea level ranges from 15 mg l^{-1} at 0° C to 8 mg l^{-1} at 25° C (Bartram and Ballance 1996).

Variations in DO can occur seasonally, or even over 24 hour periods, in relation to temperature and biological activity (i.e. photosynthesis and respiration) (Bartram and Ballance 1996). Biological respiration, including that related to decomposition processes, reduces DO concentrations. Waste discharges high in organic matter and nutrients can lead to decreases in DO concentrations as a result of the increased microbial activity (respiration) occurring during the degradation of the organic matter (EPA 2008). In severe cases of reduced oxygen concentrations (whether natural or man-made), anaerobic conditions can occur (i.e. 0 mg l^{-1} of oxygen), particularly close to the sediment-water interface as a result of decaying, sinking materials.

Determination of DO concentrations is a fundamental part of a water quality assessment since oxygen is involved in, or influences, nearly all chemical and biological processes within water bodies. Concentrations below 5 mg l^{-1} may adversely affect the functioning and survival of biological communities and below 2 mg l^{-1} may lead to the death of most fish (Bartram and Ballance 1996). The measurement of DO can be used to indicate the degree of pollution by organic matter, the destruction of organic substances and the level of self-purification of the water.

2.7.3 Nutrients

(i) Nitrogen forms

Ammonia occurs naturally in water bodies arising from the breakdown of nitrogenous organic and inorganic matter in soil and water, excretion by biota, reduction of the nitrogen gas in water by micro-organisms and from gas exchange with the atmosphere. It is also discharged into water bodies by some industrial processes (e.g. ammonia-based pulp and paper production) and also as a component of municipal or community waste. At certain pH levels, high concentrations of ammonia (NH_3) are toxic to aquatic life and, therefore, detrimental to the ecological balance of water bodies (Bartram and Ballance 1996). In aqueous solution, un-ionised ammonia exists in equilibrium with the ammonium ion. Total ammonia is the sum of these two forms. Ammonia also forms complexes with several metal ions and may be adsorbed onto colloidal particles, suspended sediments and bed sediments (Helmer and Hespanhol 1997). The concentration of un-ionised ammonia is dependent on the temperature, pH and total ammonia concentration.

Unpolluted waters contain small amounts of ammonia and ammonia compounds, usually less than 0.1 mg l^{-1} as nitrogen (Bartram and Ballance 1996). Total ammonia concentrations measured in surface waters are typically less than 0.2 mg N l^{-1} but may reach 2-3 mg N l^{-1}. Higher concentrations could be an indication of organic pollution such as from domestic sewage, industrial waste and fertiliser run-off. Ammonia is, therefore, a useful indicator of organic pollution. Natural seasonal fluctuations also occur as a result of the death and decay of aquatic organisms, particularly phytoplankton and bacteria in nutritionally rich waters (Bartram and Ballance 1996). High ammonia concentrations may also be found in the bottom waters of lakes which have become anoxic.

The nitrate ion (NO_3^-) is the common form of combined nitrogen found in natural waters. It may be biochemically reduced to nitrite (NO_2^-) by denitrification processes, usually under anaerobic conditions (Gächter et al. 2004). Under oxic conditions, the nitrite ion is rapidly oxidised to nitrate. Natural sources of nitrate to surface waters include igneous rocks, land drainage and plant and animal debris (Gächter et al. 2004). Nitrate is an essential nutrient for aquatic plants and seasonal fluctuations can be caused by plant growth and decay. Natural concentrations, which seldom exceed 0.1 mg l^{-1} N-NO_3, may be enhanced by municipal and industrial waste-waters, including leachates from waste disposal sites and sanitary landfills (Bartram and Ballance 1996). In rural and suburban areas, the use of inorganic nitrate fertilisers can be a significant source. When influenced by human activities, surface waters can have nitrate concentrations up to 5 mg l^{-1} N-NO_3, but often less than 1 mg l^{-1} N-NO_3 (Meybeck et al. 1996). Concentrations in excess of 5 mg l^{-1} N-NO_3 usually indicate pollution by human or animal waste, or fertiliser run-off. In cases of extreme pollution, concentrations may reach 200 mg l^{-1} N-NO_3. The World Health Organization (WHO) recommended maximum limit for NO_3^- in drinking water is 50 mg l^{-1} (or 11.3 mg l^{-1} as N-NO_3), and waters with higher concentrations can represent a significant health risk (Meybeck et al. 1996). In lakes, concentrations of nitrate in excess of 0.2 mg l^{-1} N-NO_3 tend to stimulate algal growth and indicate possible eutrophic conditions (Bartram and Ballance 1996).

Nitrite concentrations in freshwaters are usually very low, 0.001 mg l^{-1} N-NO_2, and rarely higher than 1 mg l^{-1}N- NO_2 (Meybeck et al. 1996). High nitrite concentrations are generally indicative of industrial effluents and are often associated with unsatisfactory microbiological quality of water. Determination of nitrate plus nitrite in surface waters gives a general indication of the nutrient status and level of organic pollution (Meybeck et al. 1996).

(ii) Phosphorus compounds

Phosphorus is rarely found in high concentrations in freshwaters as it is actively taken up by plants (Meybeck et al. 1996). As a result there can be considerable seasonal fluctuations in concentrations in surface waters. In most natural surface waters, phosphorus ranges from 0.005 to 0.020 mg l^{-1} P-PO_4 (Bartram and Ballance 1996). High concentrations of phosphates can indicate the presence of pollution and are largely responsible for eutrophic conditions (Hecky 1993, Helmer and Hespanhol 1997).

Phosphorus concentrations are usually determined as orthophosphates (soluble reactive phosphorus: PO_4^{3-}, $HPO_4^{2-,}$ $H_2PO_4^{-}$), total dissolved phosphate (dissolved organic and inorganic phosphate) or total phosphorus (organically combined phosphorus and all phosphates). The main sources of orthophosphate for aquatic ecosystems are nutrient cycling, soil leaching of soluble rocks. In fertilized agricultural lands, anthropogenic activities may generate considerable inputs of Phosphorus which may cause eutrophication.

(iii) Silica

The silica content of rivers and lakes usually varies within the range 1-30 mg l^{-1} (Bartram and Ballance 1996). Concentrations in ground and volcanic waters are higher, and thermal waters may reach concentrations up to 1 g l^{-1} or more (Bartram and Ballance, 1996). The origin of silica for aquatic systems is mainly soil leaching and erosion from the catchment.

2.8. Water quality management

According to Schindler (2006), changes in aquatic communities contribute to eutrophication via the trophic cascade, nutrient stoichiometry, and transport of nutrients from benthic to pelagic regions. Overexploitation of piscivorous fishes appears to be a particularly common amplifier of eutrophication. Internal nutrient loading can be controlled by reducing external loading, although the full response of lakes may take decades. Modern understanding of eutrophication and its management has evolved from simple control of nutrient sources to recognition that it is often a cumulative effects problem that will require protection and restoration of many features of a lake's community and its catchment Schindler (2006).

The standards-based approach to managing water quality relies heavily on environmental policies and regulations as the basis for achieving pollution control as well as institutional capacity building. Lacks of governance (weak and inadequate), transparency and incentives, research and education capacity as well as lack of public participation and awareness including the private sector involvement are among the causes of the failure to implement sustainable management frameworks needed to promote healthier ecosystems. Thus aquatic ecosystem management requires an integrated water resource management approach.

2.9 Description of the Lake Kivu river basin

Some morphometric and hydrologic parameters of Lake Kivu are given in Table 1. According to Marlier (1954), the water physico-chemistry of the inflow rivers of Lake Kivu is mostly led by the local climate (dry and wet seasons), the insolubility of the rocks and land use in the drainage basin . Water temperatures range from 12° and 15.5°C with an annual mean value of 14°C were measured for River Mushuva (Marlier 1954. Temperatures of other rivers seem to be similar. Marlier (1954) also found that the water conductivity of the rivers of the southern basin was very low due to the insolubility of the rocks in the drainage and the landscape dominated by natural forests. This implies low quantities in dissolved materials.

Alkalinity values of 0.42; 0.1 and 0.18 eq. l^{-1} were for Mushuva, Mpungwe and Lwiro respectively. The water content in bicarbonates revealed to be low in this region (6 mg L^{-1} and 11 mg. l^{-1} for Mpungwe and Lwiro respectively). The pH values of the waters measured were approaching neutrality (6.8 for Mushuva and 6.6 for Lwiro) (Marlier 1954). Water concentrations of calcium and magnesium ions seemed to be also quite low for some rivers but medium for others like Mushuva which had concentrations of 5.7 mg Ca^{2+}. l^{-1} and 3.5 mg Mg^{2+}. l^{-1} (Marlier 1954).

2.10. Plankton diversity and primary production in Lake Kivu

2.10.1. Plankton diversity

Modern Plankton diversity and abundance as well as primary production of Lake Kivu was documented and discussed in detail by Sarmento et al. (2006, 2007, 2008a&b).

The most common algal species in Lake Kivu are the pennate diatoms *Nitzshia bacteria* and *Fragilaria danica* and the Cyanobacteria, *Planktonlyngbya limnetica* and *Synechococcus sp*. (Sarmento et al., 2007& 2008a). A very high abundance of the centric diatom *Urosolenia sp*. and the cyanobacterium *Microcystis sp*. near the surface under stratification conditions has also been reported by Sarmento et al. (2008a). In total 42 taxa of pelagic algae were recorded: 14 cyanophyceae, 3 Cryptophyceae, 3 Dinophyceae, 7 Bacillariophyceae, 1 Chryophyceae, 7 Chlorophyceae, 3 Trebouxiophyceae and 4 Charophyceae (Sarmento et al., 2007& 2008a). Vertical stratification seemed to be the most important factor of diversification. These observations strengthen the statement that has always been made in the literature by many authors that Lake Kivu has a poor biodiversity (poor flora and poor fauna, with only 29 ichtyological species, Froese and Pauly, 2008).

In Lake Kivu, heterotrophic bacteria (HB) and photosynthetic picoplankton (PPP) cell numbers were found to be always high in the mixilimnion (top 60 m layer) but PPP concentrations (10^5 cell.mL^{-1}) were higher than HB (Sarmento et al., 2006). Three populations of picocyanobacteria were identified by Sarmento et al. (2006) namely: *Synechococcus* and two other categories corresponding to two and four colonies of the same taxon. The *Synechococcus* biomass in the euphotic zone (15-18 m) was estimated to 24.7 mg C. m^{-3} corresponding to 0.42 g C m^{-2} for this layer whereas the mean HB biomass in the same zone was 31.5 mg C m^{-3} corresponding to 1.42 g C m^{-2}, integrated for the whole mixolimnion (Sarmento et al. 2006 &2008b).

Present mesozooplankton is dominated by cyclopoid copepods (*Thermocyclops consimilis, Mesocyclops aequatorialis and Tropocyclops confinis*) but cladocerans and rotifers are also present (Isumbisho et al., 2006). The dependence of zooplankton on phytoplankton resource suggests that mesozooplankton dynamics in Lake Kivu is mainly bottom-up controlled, contrary to expectations from the food structure (Isumbisho et al., 2006).

2.10.2. Primary production, Elemental ratios and nutrient limitation in Lake Kivu

In Lake Kivu, the upper 60 m of the water column, which is also the maximal mixolimnion depth, Chl a values range between 0.63 and 3 mg.m^{-3} with an average of 1.37 mg. m^{-3}, with diatoms as the most dominant group of phytoplankton in the lake particularly during the dry season episodes of deeper mixing (Sarmento et al. 2008b). Mean daily primary production was found in the range of 0.19-1.09 g C.

m^{-2}. d^{-1} and and the annual average was estimated to 260 g C. m^{-2}. yr^{-1} using the ^{14}C radioactivity method (Sarmento et al.2006, 2008b).

The sestonic ratios are 256.3; 9.6 and 26.8 for C:P; C:N and N:P respectively (Sarmento et al. 2008b). These elemental ratios were interpreted as indicating moderate and infrequent N-limitation whereas they highlight a severe and frequent P-limitation especially during the rainy season for Lake Kivu. The higher elemental ratios for Lake Kivu, compared to other African Great lakes, can be interpreted as a higher nutrient limitation for Lake Kivu.

CHAPTER THREE: MATERIAL AND METHODS

3.1 Study Site

Lake Kivu is one of the lakes of the East African rift valley. It is located at 1463 m above the sea level, between 1° 34.5’ and 2° 30’ S and between 28° 50’ and 29° 23’ E, and is bordered by the Mitumba Mountains (D.R. Congo) in the west and the Rwandan Dorsal in the east. The geographical setting and catchment structure are shown in Figure 1, whereas the morphological and hydrological characteristics of the lake are summarized in Table 1.

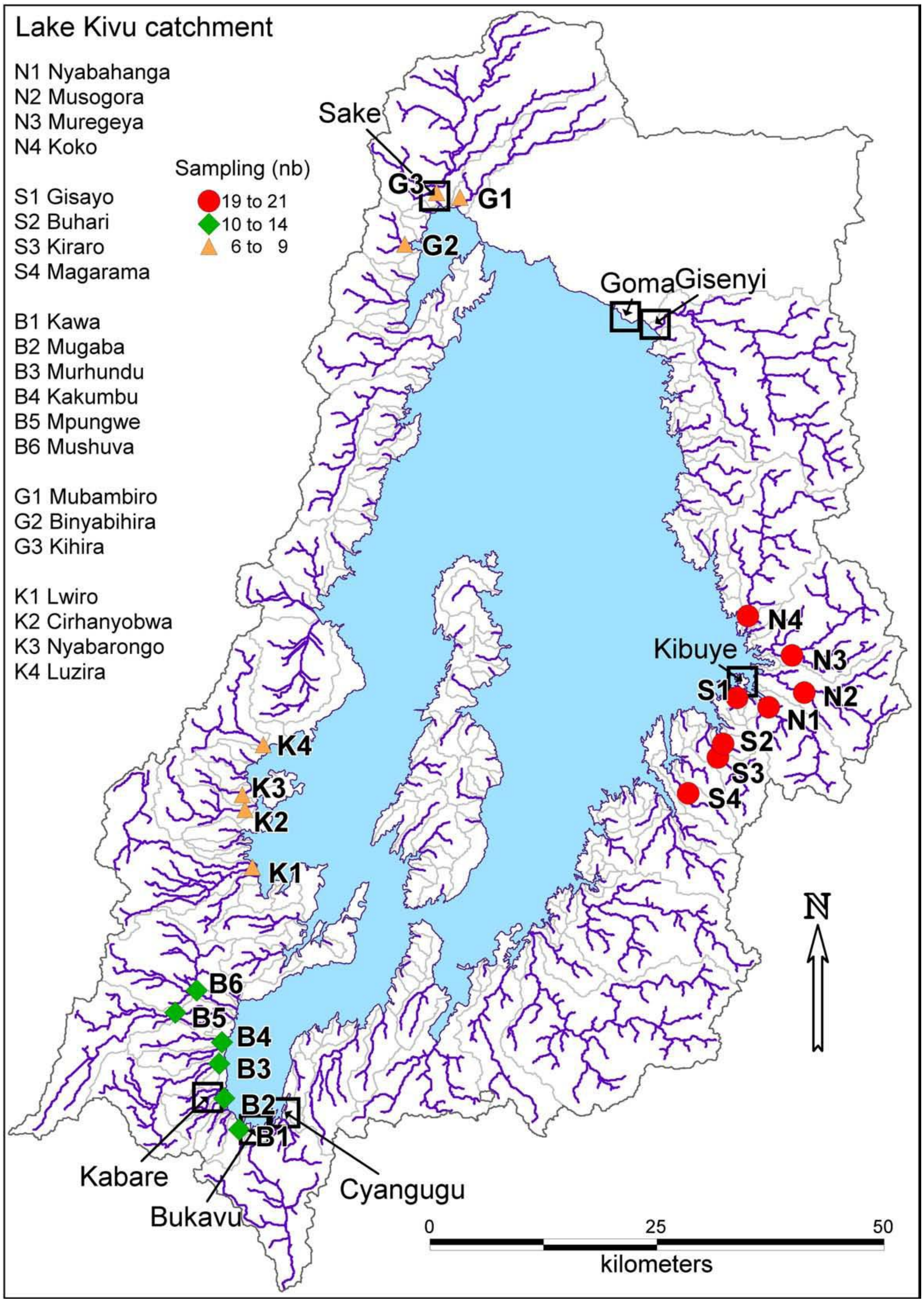


Figure 1. Geographic locations of the 21 sampled rivers in the five different regions (of a total of 127 rivers shown). Eight rivers are located on the Rwanda side (symbols N and S, Namugize 2009) and 13 rivers on the Congolese side (this study), including three near Goma (G), four near Kalehe (K) and six near Bukavu (B). The number of samples collected on each river is listed in Table 2 (circles of different sizes). Another 20 rivers have been sampled only 1 to 4 times (small grey circles) and are not included in this analysis**.**

Table 1: Morphometric and hydrological parameters of Lake Kivu

Parameter	Value
Volume[1]	580 km^3
Maximum depth[2]	485 m
Mean depth (volume/lake area)	245 m
Lake area[1]	2370 km^2
Catchment area (excluding lake)[3]	5097 km^2
Precipitation over lake surface[4]	3.3 km^3 yr^{-1}
Inflow from surface tributaries[3]	2.4 km^3 yr^{-1}
Specific inflow[5]	17 L s^{-1} km^{-2}
Internal sub-aquatic sources[6]	1.3 km^3 yr^{-1}
Evaporation lake surface[7]	3.4 km^3 yr^{-1}
Outflow[8]	3.6 km^3 yr^{-1}
Mean annual lake level fluctuation [8]	0.44 m
Flushing time (volume/outflow)	160 yr
Residence time (volume/(precipitation + inflow))	100 yr

(1) Stoffers and Hecky (1978)

(2) Beadle (1981)

(3) This study (specific inflow times river-active area of 4274 km^2)

(4) Adapted from Bergonzini (1998); precipitation = 1404 mm yr^{-1}

(5) Inflow from surface tributaries divided by the river-active area of 4274 km^2 (Muvundja et al. 2009)

(6) Schmid et al. (2005)

(7) Bultot (1954, 1977)

(8) Data from Ruzizi I Hydropower Plant (1941 to 2005).

Besides the nutrient input from external sources, the upwelling of deep-water during the season of convective cooling (internal loading) is another source of nutrients for Lake Kivu productivity. Over the centuries, the meromictic and anoxic deep-water layers have accumulated enormous amounts of gases (~300 km^3 of CO_2 and ~60 km^3 of CH_4; Schoell et al. 1988; Halbwachs et al. 2002; Schmid et al. 2004 &2005) as well as nutrients. Although the nutrient reservoir is gigantic, its availability is limited to approximately the upwelling flux caused by deep-water inflow times the nutrient concentration at the base of the seasonal mixed layer. As the water flows of the sub-aquatic sources are quite uncertain (Schmid et al. 2005), the lake internal nutrient fluxes carry a substantial uncertainty (Pasche et al. 2009). In this study, one component (river source) of nutrient fluxes into the surface layer (epilimnion) of Lake Kivu is quantified.

3.2 River Sampling

Thirteen rivers in the Congo side of the lake (Fig. 1) were sampled from Oct 2006 to Jul 2008: on a bi-weekly schedule in the basin of Bukavu (B, 6 rivers), and a monthly schedule in the other basins of Kalehe (K, 4 rivers) and Goma (G, 3 rivers). These rivers were selected according to their size, their geographical location and prevailing land use, after a first lake-wide survey in Oct 2006. The sampled rivers drain 1017 km^2, representing ~24% of the river-active area catchment of 4274 km^2 (the total catchment is 5097 km^2, Table 1). Most rivers were sampled at a distance of ~150 m from the mouth except for a few with difficult near-shore accessibility. Seven to fifteen samples were collected on the Congolese side of the catchment.

Water samples were collected in plastic bottles from the middle of the river without contacting the river bed. After rinsing twice, two plastic bottles of 0.25 or 1 l (depending on turbidity) were filled. Samples were stored in a cool-box within ice for sample preservation while transported to the laboratory for filtration and analysis. In Goma, sampling sites far from the laboratory, samples were also filtered immediately after the field and thereafter both sets of filtered and unfiltered were frozen at -20 °C before they were taken to Bukavu for analysis. Lake surface outflow concentrations of nutrients are available from the regularly sampled waters 30 km northward at Ishungu (Sarmento et al. 2006). No measurements were performed on the Ruzizi outflow. The water quality of the outflow was considered the same as the lake surface water (Langenberg et al. 2003) as confirmed by Muvundja et al. (2010).

Discharge was measured simultaneously with sampling by determining the velocity of a float and the total cross-sectional area of the river following the Float Method procedure (Harrelson et al. 1994; Bartram and Ballance 1996). The representative velocity for the entire cross-section was calculated as 0.85 times the observed velocity in the middle of the river surface (Harrelson et al. 1994).

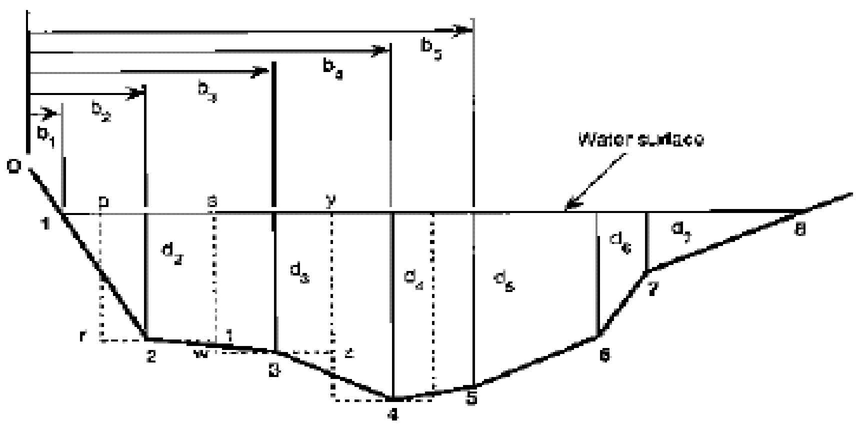


Figure 2: Cross-section of a stream divided into vertical sections for measurement of discharge (Bartram and Ballance 1996).

The river width was then divided into sub-sections for which the depths were determined according to the following chart of Fig.2. Six to seven vertical sub-sections were selected along the cross-section of the river depending on the width and shape at the site of each river. A 50 m-long tape measure was used to measure the distance (in metres) of a float in a given period of time (in seconds) measured using a chronometer. A stick-meter allowed us to determine the width of each vertical segment of the cross-section.

The computation for discharge is based on the assumption that the average velocity measured on a vertical line is valid for a rectangle that extends half of the distance to the verticals on each side of it, as well as throughout the depth at the vertical (Bartram and Balance 1996; Harrelson et al. 1994). In Fig. 2, the mean velocity (v_n) applies to each rectangle bounded by the given dashed line. The area (a_n) of each rectangle is given in function of the width (b_n) by:

$\mathbf{a_n = d_n * [b_{n+1} - b_{n-1}]/2}$ **;** and the discharge (Q_n) through it will be:

$\mathbf{Q_n = a_n * v_n}$

Therefore, the total discharge (Q_T) across the whole cross-section was given by:

$\boldsymbol{Q_T = Q_1 + Q_2 + Q_3 \ldots Q_{(n-r)} + Q_n}$

The geographical coordinates of the river sampling locations (Fig. 1) were recorded using a Garmin Olathe 72 GPS (Garmin, USA).

3.3 Physico-chemical parameter measurements

In the same cross-section where discharge was measured, some water physico-chemical parameters were determined in situ. The pH and temperature were measured using a pH-meter (pH/temperature WTW pH 330/340, Germany) and the electrical conductivity was measured by a Conductivity/TDS meter (Cyberscan CON 400/410&TDS 400, Singapore). Total suspended solids were measured using gravimetric methods on dry filters after 12 hours of drying in an oven at 80 °C.

3.4. Nutrient Analyses and Load estimation

(a) Chemical analysis

Nutrient concentrations were analyzed applying standard spectrophotometric techniques

(APHA 1992&2005) with a UV/VIS spectrophotometer (Spectronic ®20 Genesys,USA). Samples were filtered using Whatman GF/C filters (1.2 µm, 47 mm) conditioned by heating for 2 h at 400 °C, and with cellulose acetate filters OE 67 (0.45 µm, 47 mm) using a vacuum pump (Gelman, USA). Total Phosphorus (TP) was measured in unfiltered water after digestion using persulphate in an autoclave (Wolf Samoclav KL 12-2, Germany) for 2 h at 120 °C. Soluble Reactive Phosphorus (SRP) was determined using the molybdenum blue method. The nitrogen species were analyzed as follows: Ammonia (NH_4^+) by the dichloroisocyanurate-salicylate method, nitrate (NO_3^-) by the cadmium reduction method, and nitrite (NO_2^-) with azo-dye complex formation. Soluble Reactive silica (SRSi) was analyzed with the molybdate complex method (APHA 1992&2005).

b) Nutrient specific load estimation

Instantaneous transport of a nutrient [mg d^{-1}] is given by multiplying its riverine concentration [mg m^{-3}] by the actual discharge [m^3 d^{-1}] (Schärer et al. 2006; Moosman et al. 2005; Langenberg et al. 2003). The annual load is extrapolated by multiplying the average concentration times the annual discharge [m^3 yr^{-1}] of the river. The specific load for each river was obtained by dividing the annual load by the catchment area of the indicated river. The specific loads for the 21 rivers (13 in Congo and 8 in Rwanda) sampled by Muvundja et al. (2009), allowed us to extrapolate the total riverine nutrient load by multiplying with the entire river-active catchment area of the lake (factor 2.76, Muvundja et al. 2009). The volcanic area near Goma and Nyiragongo without rivers (Fig. 1) was not considered

c) River contribution to primary production

Nutrient limitation and primary production as well as Elemental ratio composition of algae have recently been documented by Sarmento et al. (2006, 2008). In this study, I referred to the current primary production (260 g C m^{-2} yr^{-1}, Sarmento et al. 2006 &2008b) and elemental composition of the phytoplankton (C:P= 256.3, Sarmento et al. 2008b) to estimate the relative contribution of the riverine inputs of nutrients to the epilimnion of the lake.

3.5 Soil and Water Assessment Tool Model

(a) Meteorological data

In this study, data collected by Davis Weather Wizard III units (Davis Inotek Instruments, USA) at four meteorological stations, consisting of the three DR Congo stations at Bukavu (ISP), Lwiro (Centre des Recherches en Sciences Naturelles, CRSN), and Goma (Observatoire Volcanologique de Goma, OVG) and that of Kibuye (CB) in Rwanda are used. This weather data were added to the long-term meteorological data of the region (35 stations in Rwanda and DR Congo) from Bultot (1954&1977), the INEAC publication (1960) and the Rwanda Meteorological Service**.** Daily precipitation data and minimum and maximum temperature data are from Rwanda Meteorological Institution. Long-term rainfall averages are a combination of data by Rwanda Meteorological Institution and the publications of Bultot (1954). As the different precipitation and temperature records have variable length and quality, only 3 temperature and 6 precipitation records, spanning the period 1944 to 1993, have been used. Additional daily data are taken from nearby stations or generated by the SWAT weather generator (Neitsch *et al*. 2005) based on climate data from three different weather stations.

(b) Land use modeling

To extrapolate the load measurements of the 13 sampled rivers, which are highly variable due to flow dynamics, vegetation and soil types (Fig. 3), to all the 127 river of the drainage area, alternative approaches were applied. Besides scaling up from the 1017 km^2 observed to the entire river-active area (Table 3), also the measured loads are correlated to the load estimations of the SWAT model (Neitsch et al. 2005).

The SWAT model is a physically based basin-scale model developed for continuous simulations of water and soil quality. The required input data are weather, soil properties, topography and vegetation. The study area was divided into the 127 river basins and the hydrological parameters are determined for each by the dominant land use and soil types (Fig. 3). The model was applied for the time period from 1944 to 1993. As there is no river flow data prior to the study measurements, the model is not calibrated and the model output was interpreted only relative to the study measurements. The SWAT model for the Lake Kivu catchment was constructed by Rinta (2009) based on the following readily and freely available datasets:

(i) Digital Elevation

The model is constructed from the US Geological Survey's HydroSHEDS database (Lehner et al. 2008, www.hydrosheds.cr.usgs.gov/). HydroSHED elevation data have been derived from of the Shuttle Radar Topography Mission at 3 arc-second resolution (~90 m on the latitudes of Kivu). Therefore the 127 river watersheds are mapped and their areas determined, using MapInfo Software.

(ii) Land use

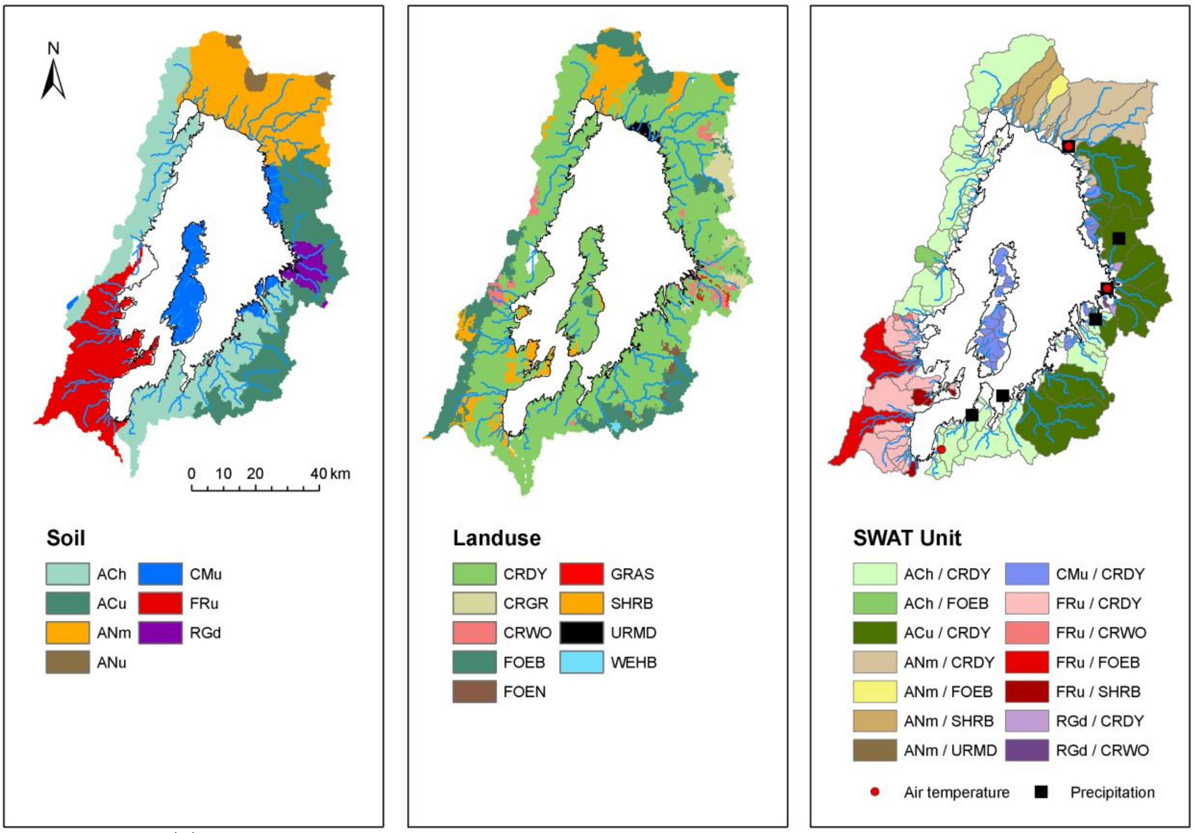
For both countries Congo and Rwanda, information is accessed from the FAO public domain. Spatially Aggregated Multipurpose Land cover database was provided by Africover (2008). The land cover data are based on remote sensing images**.** Land cover classes are assigned to African land use classes of Schuol et al. (2008) and the respective characteristics were used for nutrient assessment.

(iii) Soil map and soil parameters

These data are extracted from Batjes (2007) and the FAO database (SOTWIScaf 2008), derived from Soil and Terrain database for Central Africa and profiles on the ISRIC-WISE database. Soil properties, such as permeability, rooting depth, texture, albedo are obtained from Schuol et al. (2008) or calculated by using pedotransfer functions in the model ROSETTA (Schaap et al. 2001, http://www.ars.usda.gov/Services/docs.htm?docid=8953; USDA NRCS 2009).

3.6. Statistical Analysis and Graph design tools

Data compilation and statistic calculations both descriptive and regressions were performed using Microsoft EXCEL package (Office 2003). Graphs were designed using Dplot (Alwil Software Engine, version 4.8.1368.0).



(a) (b) (c)

Figure 3. GIS maps showing soil types (a), land use (b) and SWAT units (c) for the Lake Kivu catchment. The indicated rivers are created with the SWAT model (not necessarily existing rivers).

(a) Soil map, as obtained from Batjes (2007) SOTWIScaf database representing FAO soil types (left).

(b) Land use, as obtained from FAO Africover (2008) database, reclassified to land use classes used by SWAT model (middle).

(c) Map of the hydrological watersheds, representing land use and soil classes, as used in the SWAT model (right). Locations of air temperature (3) and rain measurements (6) are indicated (right).

The abbreviations have the following meaning:

Soil types: ACh = Haplic Acrosols, ACu = Humic Acrosols, ANm = Mollic Andosols, ANu = Umbric Andosols, CMu = Humic Cambisols, FRu = Humic Ferralsols, RGd = Dystric Regosols.

Land use: CRDY = Dryland Cropland and Pasture, CRGR = Cropland-Grassland Mosaic, CRWO = Cropland-Woodland Mosaic, FOEB = Evergreen Broadleaf Forest, FOEN = Evergreen Needleleaf Forest, GRAS = Grassland, SHRB = Shrubland, URMD = Residential of Medium Density, WEHB = Herbaceous Wetland.

CHAPTER FOUR: RESULTS

4.1 River Discharge and Lake Water Balance

The 144 discharge measurements performed on the 13 rivers of the Congolese side (Table 2), covering a monitored drainage area of 1017 km^2, add to a total discharge of 22.1 $m^3 s^{-1}$ (~0.7 $km^3 yr^{-1}$). The 127 rivers of Lake Kivu discharges are highly variable (min, max in Table 2) on top of the seasonal flow regime shown in Figure 4. Due to the small catchment, the seasonality of the flow is hardly delayed relative to the precipitation regime, dominated by ~9 months of rainy season (September to May) and ~3 months (June, July, August) of dry season (Fig. 4). The average annual precipitation in the catchment is 1404 mm yr^{-1} (section 4.3).

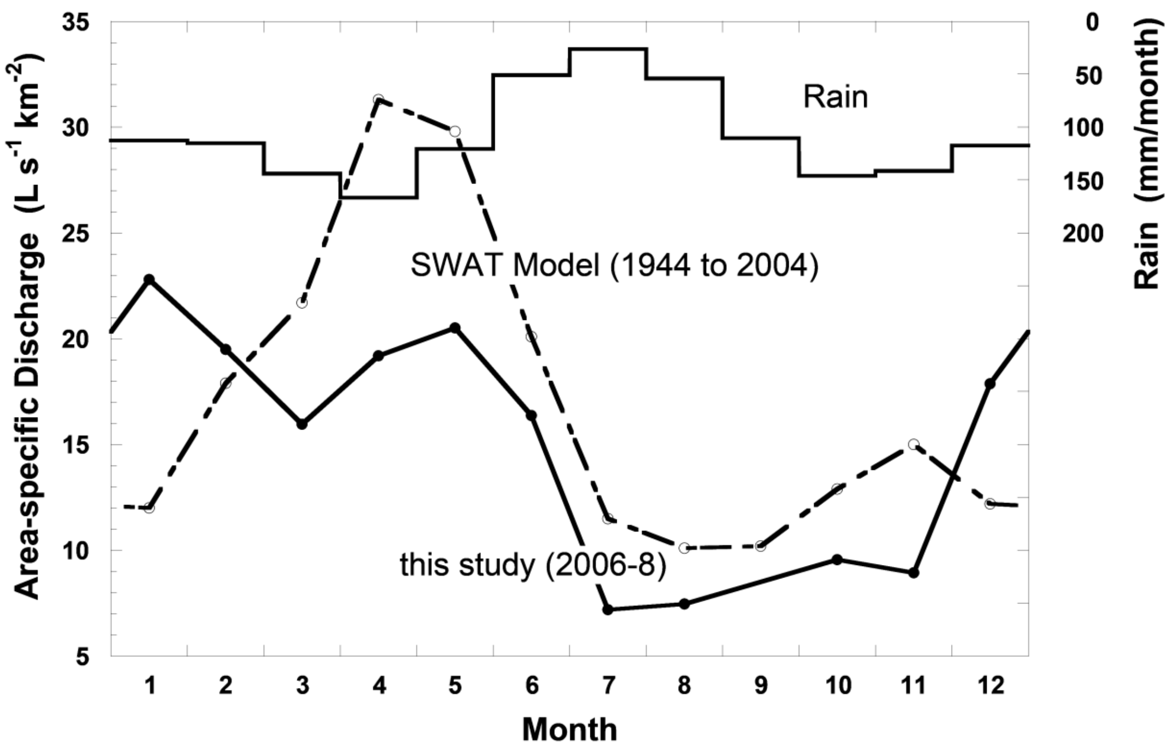


Figure 4. Monthly-averaged specific discharge of river inputs (left scale) as measured from October 2006 to July 2008 compared to the SWAT model data for 1944 to 2004 and compared to the monthly-averaged precipitation for 1932 to 2008 (right scale). The SWAT discharge is multiplied by a factor of 1.35 to yield the same specific discharge of 17.0 L s^{-1} km^{-2}. (For data source: see text).

The 13 rivers whose flows were measured (Fig. 4) show a reasonably good positive relationship ($R^2 = 0.62$) of average discharges on the catchment areas (both listed in Table 2). When combined with data from the Eastern part of the catchment (Muvundja et al., 2009), the calculated average area-specific flow falls to 17 L s^{-1} km^{-2}.This allows us to estimate the annual lake inflow of all the 127 rivers by multiplying it with the river-active area of 4274 km^2 (Table 1). Depending on the regression used, the inflow totals to ~2.4 km^3 yr^{-1}, with at least 10% error. Additional uncertainty stems from the systematic flow measurement errors. The ~830 km^2 large river-free dry volcanic area of Goma–Nyiragongo (Table 1), excluded from this riverine inflow estimate, is assumed leaking the rain water into the underground, where it most probably feeds the sub-aquatic sources of the lake, estimated to 1.3 km^3 yr^{-1} by Schmid et al. (2005). This value is consistent with the assumption of infiltration of the entire estimated precipitation (1404 mm yr^{-1}) over the 830 km^2 (1.2 km^3 yr^{-1}). The third input to Lake Kivu is precipitation onto the 2370 km^2 of the lake surface, yielding 3.3 km^3 yr^{-1} (Table 1).

These inputs are in surprisingly good agreement with the outflow and evaporation estimates. Data on Ruzizi outflow recorded in the last 64 years by the Congolese hydropower Company (SNEL) at Ruzizi I dam and evaporation data from Bergonzini (1998) yield water outputs of 3.6 and 3.4 $km^3 yr^{-1}$ for outflow and evaporation, respectively.

In summary, the total water input of 7.0 km^3 yr^{-1} [2.4 (rivers) + 1.3 (sub-aquatic) + 3.3 (rain on lake)] coincides perfectly with the total output of 7.0 km^3 yr^{-1} [(3.6 (Ruzizi) + 3.4 (evaporation)]. Despite this agreement, the overall uncertainty in the water budget is probably ~15 to 20%, which is comparable to the mean annual lake level fluctuation of 0.44 m (64 years of data by SNEL), equivalent to ~1 km^3.

Table 2: Discharges, particle concentrations, catchment areas, population and the major land use patterns of the 13 sampled rivers of Congo

River Code name	Discharge Average	Discharge (min - max)	Discharge N[2]	Particle Concentration Average	Particle Concentration (min - max)	Particle Concentration N	Catchment Area	Catchment Pop	Land use[1] CRDY	Land use[1] CRWO	Land use[1] SHRB	Land use[1] FOEB
	$m^3 s^{-1}$			$mg L^{-1}$			km^2	1000	%			
B1 Kawa	0.27	(0.16 - 0.62)	14	244	(46 - 535)	10	11	90	38	0	56	0
B2 Mugaba	0.26	(0.07 - 1.22)	15	396	(134 - 895)	11	11	2	76	0	24	0
B3 Murhundu	1.29	(0.18 - 2.26)	15	561	(163 - 970)	11	81	17	64	0	21	15
B4 Kakumbu	0.69	(0.27 - 1.91)	14	180	(35 - 326)	10	24	5	66	0	34	0
B5 Mpungwe	1.87	(0.37 - 5.81)	14	80	(22 - 279)	10	119	25	16	0	13	70
B6 Mushuva	1.04	(0.50 - 1.81)	13	97	(35 - 180)	10	106	22	75	0	0	25
K1 Lwiro	7.30	(3.08 - 12.72)	10	60	(35 - 118)	5	142	29	36	0	25	39
K2 Cirhanyobwa	1.60	(0.59 - 2.93)	9	99	(61 - 148)	5	50	10	53	0	0	47
K3 Nyabarongo	0.50	(0.26 - 0.95)	9	70	(42 - 96)	5	20	4	49	2	9	39
K4 Luzira	2.33	(1.26 - 4.50)	9	23	(12 - 35)	5	50	9	29	28	1	42
G1 Mubambiro	1.11	(0.84 - 1.59)	7	2	(1 - 3)	2	77	9	0	0	86	14
G2 Binyabihira	0.21	(0.17 - 0.26)	7	82	(54 - 107)	3	30	4	43	0	27	29
G3 Kihira	3.60	(2.67 - 4.52)	8	110	(29 - 191)	2	296	37	98	0	1	0
Sum	**22.1**						**1017[3]**	**263**				
Average	**1.7**		**11**	**107[4]**		**7**			**57[5]**	**1.4**	**16**	**25**

[1] The land use abbreviations have the following meaning: CRDY = Dryland Cropland and Pasture; CRWO = Cropland-Woodland Mosaic; SHRB = Shrubland; FOEB = Evergreen Broadleaf Forest.
[2] N = number of water samples collected on the particular river
[3] The 13 sampled rivers cover an area of 1017 km^2 out of the total river-active area of 4274 km^2 (total catchment = 5097 km^2; Table 1)
[4] Discharge-weighted average of the particle concentration
[5] Area-weighted averages of the land use percentages.

4.2 Water physico-chemical parameters of the rivers

Data from the Bukavu and Kalehe basins are provided in Table 3. River Kawa had a slight high mean values of pH (7.4) and high conductivity (638 µS cm^{-1}) compared to rivers in Kalehe basin (Table 3). The mean electrical conductivity of the sampled rivers of Bukavu and Kalehe basins varied from 144 to 638 µS cm^{-1}. This represents salinities ranging between ~120- ~480 mg l^{-1} (calculated by a conversion factor of 0.76, Vandelannoote et al. 1999). However, a limited number of data showed that the rivers in Goma basin experience higher salt contents (G1: 2,530 µS cm^{-1}, G2: 1,620 µS cm^{-1} and G3: 2,630 µS cm^{-1}) most probably due to the leaching of volcanic ashes by rivers.

Table 3: Physico-chemical measurements of water samples in rivers of two sub-basins of Lake Kivu

Code	River name	No. Samples	pH	±	SD	EC (*)	±	SD	T	±	SD	v	±	SD
Unit						(µS cm^{-1})			(°C)			(m s^{-1})		
B1	Kawa	10	7.4	±	0.4	638	±	324	21.5	±	2.1	1.0	±	0.1
B2	Mugaba/Bwindi	11	6.8	±	0.9	184	±	131	22.4	±	2.6	1.3	±	0.4
B3	Murhundu	11	6.4	±	0.7	176	±	109	21.2	±	2.3	0.9	±	0.4
B4	Kakumbu	11	7.1	±	0.4	278	±	106	23.1	±	2.6	1.1	±	0.3
B5	Mpungwe	11	6.7	±	0.6	144	±	66	19.1	±	2.0	2.7	±	0.8
B6	Mushuva	10	6.7	±	0.4	161	±	59	19.4	±	1.9	1.2	±	0.5
K1	Lwiro	7	7.0	±	0.5	179	±	108	19.8	±	2.3	0.9	±	0.3
K2	Cirhanyobwa	7	6.8	±	0.5	186	±	69	19.9	±	1.2	1.1	±	0.3
K3	Nyabarongo	7	7.0	±	0.4	201	±	63	21.1	±	3.8	1.0	±	0.4
K4	Luzira	7	6.8	±	0.4	152	±	60	18.0	±	0.8	1.2	±	0.4

(*) Averages are temperature-weighted. EC is the mean electrical conductivity; T is the mean temperature and v the mean water velocity.

The mean water temperatures of these rivers varied from 18.0 to 23.1°C. The rivers affected by human disturbance (such as B1 to B5 and K3) showed higher temperature fluctuations (Table 3). The mean velocity of the water current ranged between 0.9 to 2.7 m s^{-1}(Table 3). Mean river discharges were comprised of discharges between 0.21 and 7.30 $m^3. s^{-1}$ (Table 2).

4.3 Riverine Nutrients

4.3.1. Concentration variability and land use in river basins

The concentration of total suspended solid and nutrient from 89 measurements (TSS), 68 samples (SRP), 46 (TP), 53 (NH_4^+), 26 (NO_3^-), and 92 (SRSi) samples, are summarized in Table 2 (for TSS) and Table 4 (for other parameters). Depending on the flow dynamics, there is substantial variability of at least an order of magnitude (min to max in Tables 2 and 4) in the few samples from each individual river. Mean concentration values estimated are 49, 730, 168 and 505 µg.L^{-1} for soluble reactive phosphorus, total phosphorus, ammonia and nitrate respectively (Table 4). Mean value of silica concentration was 11.2 mg. L^{-1}.

Water quality of River Kawa was considerably different from other rivers in terms of nutrient concentrations other than silica (Table 4). The variation between rivers is much smaller, except for Kawa (B1). Due to the high population density in the catchment (Table 2), Kawa receives the highest loads of P (SRP and TP) and NH_4^+ (Table 4). However, TP was also quite varaiable for some of the rivers (Table 4). Rivers of the southern basin (Fig.1), in general, experienced higher concentrations of ammonia (14 to 9850 µg N. L^{-1}) compared to rivers in the Norhern basin (5 to 81 µg.L^{-1}, Table 4).

Table 4: Measured nutrient concentrations in the 13 sampled rivers of Congo

Nutrient	SRP			TP			NH_4^+			NO_3^-			SRSi		
River Code name	Ave	(min - max) µg P L^{-1}	N[(1)]	Ave	(min - max) µg P L^{-1}	N	Ave	(min - max) µg N L^{-1}	N	Ave	(min-max) µg N L^{-1}	N	Ave	(min-max) mg Si L^{-1}	N
B1 Kawa	686	(20-1340)	7	7960	(1060-31600)	5	9850	(4160-17400)	9	510	(5-1140)	3	11.6	(0.4-15.1)	9
B2 Mugaba	42	(15-73)	7	1346	(171-3970)	5	166	(24-540)	7	440	(89-770)	3	9.2	(1.4-15.9)	9
B3 Murhundu	23	(5-36)	6	2000	(290-5010)	5	106	(3-320)	8	440	(278-520)	3	7.3	(0.7-14.6)	9
B4 Kakumbu	51	(22-72)	6	925	(165-3080)	5	150	(8-500)	7	540	(425-650)	3	12.5	(5.0-19.7)	9
B5 Mpungwe	34	(19-47)	7	301	(214-470)	4	47	(5-130)	3	110	(35-160)	3	6.4	(2.6-9.6)	9
B6 Mushuva	80	(19-276)	7	417	(93-1000)	4	65	(5-210)	4	120	(47-190)	3	6.9	(3.2-11.2)	8
K1 Lwiro	46	(24-61)	5	143	(87-200)	2	14		1	320	(296-340)	2	10.5	(7.8-12.3)	6
K2 Cirhanyobwa	35	(14-58)	5	453	(68-1190)	3	98	(25-170)	2	260	(135-390)	2	9.3	(4.2-13.0)	6
K3 Nyabarongo	33	(12-45)	5	511	(67-1340)	3	57	(4-110)	2	190	(137-240)	2	13.0	(4.9-19.0)	6
K4 Luzira	23	(11-42)	4	75	(50-100)	2	10		1	370	(297-440)	2	8.3	(6.6-9.6)	6
G1 Mubambiro	45	(24-85)	3	476	(2-950)	2	5	(1-10)	2				21.4	(9.6-27.4)	5
G2 Binyabihira	139	(121-154)	3	2500	(198-5170)	3	57	(45-70)	3				15.9	(12.0-17.1)	5
G3 Kihira	37	(22-65)	3	1710	(62-3700)	3	81	(43-120)	4				16.8	(4.6-28.7)	5
Average[(2)]	**49**			**730**			**168**			**305**			**11.2**		

[(1)] N = number of water samples analyzed on the particular river

[(2)] Averages are discharge-weighted (Table 2).

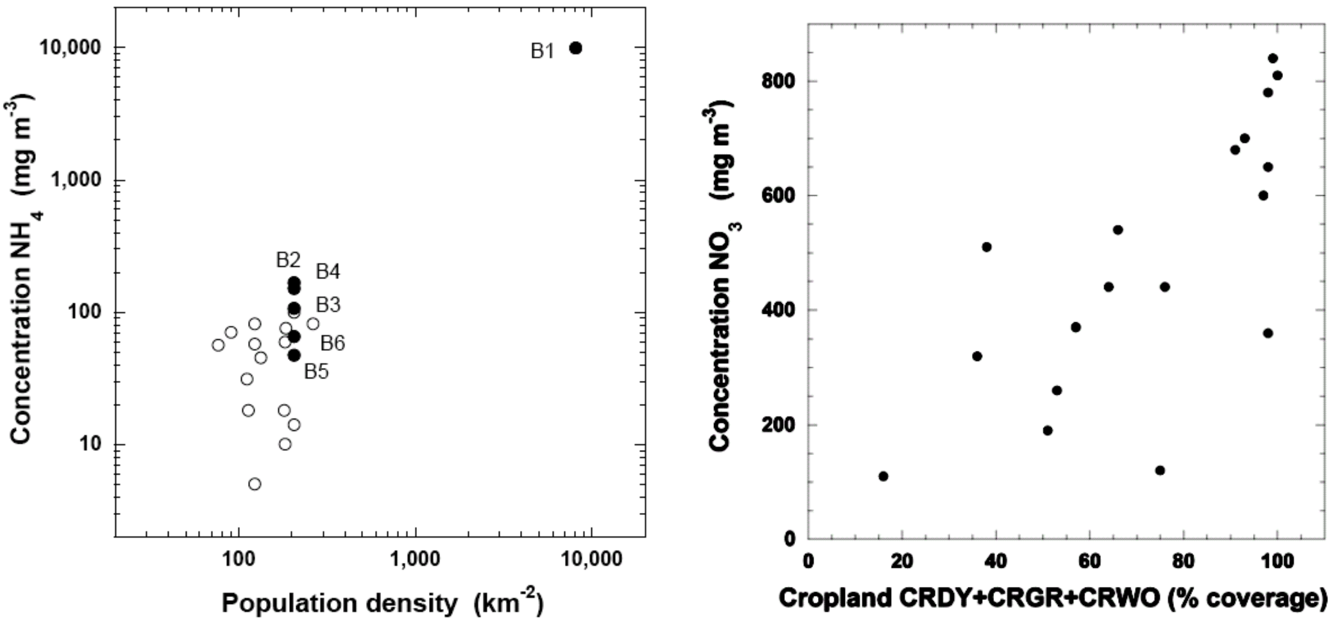
Although the other river catchments are much less affected, there are positive correlations for NH_4^+ (Fig. 5a) and SRP (Fig. 6a) with population density (Muvundja et al. 2009). Especially high NH_4^+ concentrations were observed in the populated areas near Bukavu (Fig.1): the four rivers with higher NH_4^+ concentrations (Table 4) are all located in the Bukavu basin, whereas all other rivers (with smaller populations in their catchments) show NH_4^+ concentrations of < 0.1 mg l^{-1} (Fig. 5a, Muvundja et al. 2009).

The NO_3^- concentrations show variability (factor ~2) among the rivers and average at 0.40 mg N L^{-1}. It should be noticed that the NO_3^- concentrations in the heavily polluted Kawa (B1) are not enhanced, despite the much higher NH_4^+ levels. There is a systematic difference in riverine NO_3^- concentrations, which are about two times higher on the Rwandan compared to the Congo territory (Muvundja et al. 2009; Namugize 2009) probably owing to a difference in the use of fertilizer for agriculture. A small number of analyses conducted for NO_2^- (not shown in this study), all indicated minor and hence negligible values in comparison to NO_3^- and NH_4^+. The average NO_2^- concentration is 8 µg N l^{-1} (ranging between 0 and 21 µg N l^{-1}). Therefore, nitrite loads can be ignored for the estimation of the dissolved inorganic nitrogen (DIN) inputs.

In agreement with the low variability of nutrient concentrations among the tributaries, no systematic differences among the three western basins of Lake Kivu were observed, except: (i) in Bukavu basin, where NH_4^+, TSS and - evidently related - TP are higher, emphasizing anthropogenic influences (Table 5) and (ii) the higher NO_3^- content in the Rwandan rivers (Muvundja et al. 2009; Namugize 2009).

The mean value of total suspended solids in the Congolese part of the catchment was 107 mg. l^{-1}, the minimum was 2 mg.l^{-1} (River Mubambiro in Goma) and the maximum was 561 mg.L^{-1} (River Murhundu in Bukavu , Table 2, Fig.1).

River Luzira showed the lowest concentrations of all N and P species compared to River Kawa which showed the highest values except for nitrate as it serves as a waste disposal. SRSi concentrations show much less fluctuation than the other nutrients analyzed during this study. Silica concentrations varied between 7.3 (River Murhundu) to 21.4 mg Si l^{-1} (River Mubambiro, Table 4), The average for all the rivers studied was 11.2 mg Si l^{-1}. Rivers located in the Northern basin experience higher silica concentrations than those located in the south (Table 4).



(a) **(b)**

Figure 5.

a) Average NH_4^+ concentration of the 21 sampled rivers versus population density in the corresponding catchment. Black circles represent the six rivers from Bukavu basin (B1 to B6, Table 2), and the open circles represent the other 15 rivers (Muvundja et al. 2009; Namugize 2009).
b) Average NO_3^- concentration of 18 rivers versus cropland coverage (CRDY+CRGR+CRWO, Fig. 3, Table 2; Muvundja et al. 2009; Namugize 2009).

The catchment size of the Congolese river basins was comprised between 11 and 296 km^2 (Table 2). The land use in the studied site was 57% cropland-dryland and pasture, 25% Evergreen Broadleaf Forest, 16% Shrubland and 1.4% Cropland-Woodland Mosaic (Table 2). Only River Mubambiro basin is free of agricultural activities as its drainage area is part of the Virunga National Park. The dominant land use in this basin is shrubland (86%) followed by Evergreen Broadleaf Forest (14%) (Table 2). However, the most cultivated basin is Kihira basin with 98% of Cropland-dryland occupation and 1% of Shrubland. The catchment of River Mpungwe is dominated by forest (70%) compared to those of Kawa, Mugaba and Kihira basins (Table 2).

The most densely occupied by human population is the basin of Kawa with 8,182 inhabitants per km^2 as it drains the most populated areas of the town of Bukavu (Table 2). No river basin drains the town of Goma. The least populated basin is, of course, Mubambiro basin with 117 inhabitants per km^2. In the 5097 km^2 large Kivu catchment, the population has steeply increased to 2.1 millions over the last decades (Table 2, Census 2007a, 2007b, 2007c) yielding an average density of at least 260 inhabitants km^{-2} (Table 2).

4.3.2. Nutrient loads from the Congolese lake basin

(i) Phosphorus load

The average SRP level is only 0.05 mg P L^{-1} - unexpectedly low for such intensely used catchments – resulting in an overall load from the sampled catchment (Congo side, 1017 km^2) of 34 t P yr^{-1} or 33.7 kg P km^{-2} yr^{-1} (Table 5). The corresponding values for TP load is 541 t P yr^{-1}, or 533 kg P km^{-2} yr^{-1}, respectively, with a TP:SRP ratio of 16. The largest contribution of ~34% of the total SRP load to the lake from the Congolese side~~,~~ comes from River Lwiro (K1), partly due to the large catchment (~14% of the total area of the considered side) and partly due to the higher specific loads (factor ~2.4 above the average, Table 5). The higher specific loads of these rivers are due to the population density as for River Kawa (B1), the main sewer of Bukavu town, carrying enormous amounts of organic waste and nutrients and contributing more than 5 t P yr^{-1}, or ~15% of the entire Congolese riverine load to Lake Kivu (Table 5). Specific loads of SRP and TP of the River Kawa are 14 and 9 times higher than the mean values of the sampled rivers (Table 5). Other neighboring rivers (Fig. 1) showed the same features (Table 5). Muvundja et al. (2009) found a good positive relationship between the SRP load and the log transformed data of absolute population in the corresponding catchments (Fig. 6a). In Bukavu region, Total Phosphorus load corresponds to 0.95 kg P per inhabitant and per year. The largest TP contribution of 45.8 % stems from River Kihira (G3), mainly due to its large drainage area (~29 % of the total Congolese side area). The load estimation has, however, a high uncertainty due to the wide concentration ranges experienced in the few samples (Tables 2 and 5).

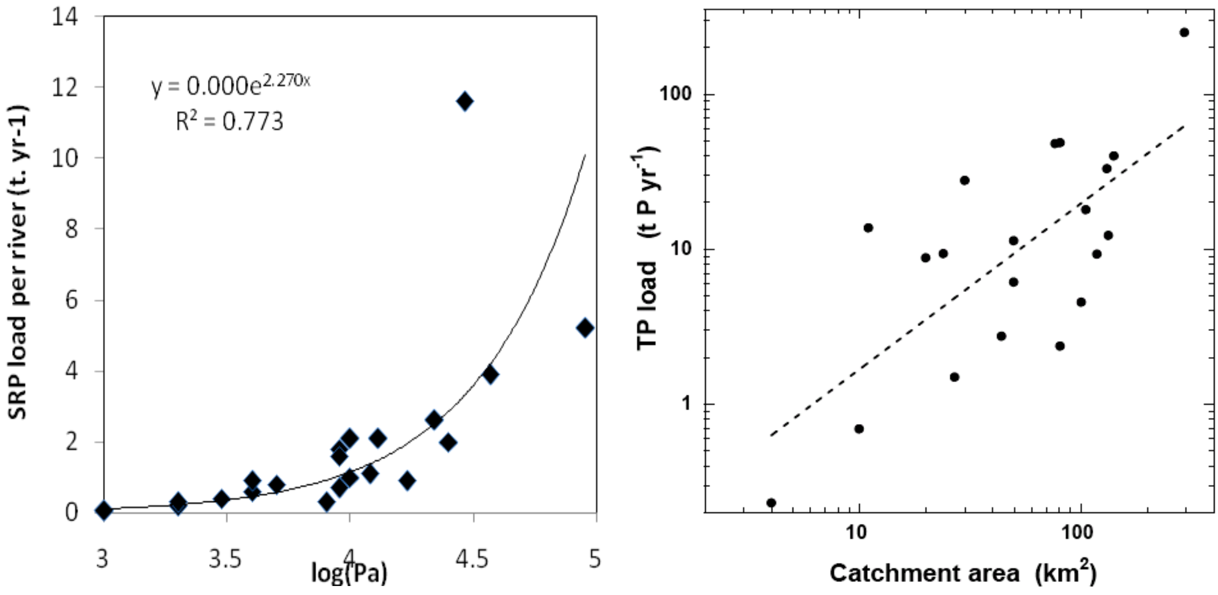
A slight positive relationship was observed between TP loads and river catchment area (Fig.6b).

Table 5: Annual riverine nutrient loads of the 13 sampled rivers of Congo

Nutrient	SRP		TP		NH_4^+		NO_3^-		SRSi		TSS	
River Code name	Load t P yr^{-1}	Sp. load[1] kg P km^{-2} yr^{-1}	Load t P yr^{-1}	Sp. load kg P km^{-2} yr^{-1}	Load t N yr^{-1}	Sp. load kg N km^{-2} yr^{-1}	Load t N yr^{-1}	Sp. load kg N km^{-2} yr^{-1}	Load t Si yr^{-1}	Sp. load t Si km^{-2} yr^{-1}	Load t yr^{-1}	Sp. load t km^{-2} yr^{-1}
B1 Kawa	5.2	472	54.0	4,910	74.3	6,750	3	298	81	7.4	2,500	229
B2 Mugaba	0.3	28	13.7	1,240	1.2	113	2	192	51	4.6	3,800	345
B3 Murhundu	0.9	11	48.3	596	5.9	73	12	150	234	2.9	27,400	338
B4 Kakumbu	0.8	35	9.3	388	4.8	200	9	364	211	8.8	4,200	175
B5 Mpungwe	2.0	16	9.2	78	8.2	69	4	32	273	2.3	5,100	43
B6 Mushuva	2.6	25	17.9	169	2.4	23	4	33	237	2.2	3,000	29
K1 Lwiro	11.6	82	39.7	280	5.5	39	79	554	2,210	15.6	15,500	109
K2 Cirhanyobwa	2.1	43	11.3	226	5.7	114	14	281	443	8.9	5,700	113
K3 Nyabarongo	0.6	28	8.8	438	0.8	41	2	108	205	10.3	1,300	64
K4 Luzira	1.8	36	6.1	122	1.4	28	29	581	623	12.5	1,600	32
G1 Mubambiro	1.6	20	47.7	619	0.1	2			745	9.7	80	1
G2 Binyabihira	0.9	28	27.6	920	0.4	13			107	3.6	520	17
G3 Kihira	3.9	13	248	840	9.1	31			1,900	6.4	14,700	49
Sum	**34**		**541**		**120**		**157**		**7,300**		**85,000**	
Average	**3.8**	**33.7**[2]	**91**	**533**	**6.4**	**120**	**15.2**	**256**	**1060**	**7.2**	**10,110**	**84**

[1] Specific load = load divided by the catchment area of the particular river

[2] Averages are weighted by catchment area (Table 3) (= total sampled load per total sampled area).



(a) **(b)**

Figure 6. (a) Riverine SRP load of the 21 sampled rivers versus log transformed absolute population (Pa) in the corresponding catchment. The regression line indicates the power law of 0.00010 x $e^{[2.270 \log (Pa)]}$ with R^2= 0.77 as fit to data.
(**b)** Riverine TP load of the 21 sampled rivers versus area (A) of the corresponding catchment. The dashed line indicates the power law 0.14 x $A^{1.07}$ fit to data.

(ii) Nitrogen load

Estimates of the riverine DIN loads for the 13 rivers yield 120 and 157 t N yr^{-1} (Table 5) for $NH_4^+$and NO_3^-, respectively, with specific loads of 118 (NH_4^+) and 256 (NO_3^-) kg N km^{-2} yr^{-1}. As mentioned above, Kawa is heavily contaminated with NH_4^+ and contributes about 15% of the total riverine NH_4^+ load to Lake Kivu with an area-specific load 56 times more than the mean value of the 13 rivers (Congolese side). There is a positive relationship between NH_4^+ concentrations and the population density in the respective basin (Fig.5a) whereas for NO_3^- the positive relationship was found between nitrate concentrations and the rate of land vegetation coverage in the catchment (Fig.5b). In the densely populated region of Bukavu, dissolve inorganic nitrogen load corresponds to 0.81 kg N per inhabitant and per year.

(iii) Silica load

An estimated 7,300 t Si were transported annually from the western Kivu catchment, resulting in a specific load of 7.2 t Si km^{-2} yr^{-1} (Table 5). On the Congolese side the sampled rivers showed higher specific loads (7.2 ± 4.2 t Si km^{-2} yr^{-1}) than in the Rwandan side (2.6 ± 1 t yr^{-1} km^{-2}, Namugize 2009), probably due to difference in soil types (Fig.3).

(iv) Total suspended solids

TSS loads are high, 85,000 t yr^{-1} for the 1017 km^2 sampled area in Congo, with a mean specific load of 84 t km^{-2} yr^{-1}. Rivers from catchments, which are dominated by high shrubland and humic ferrosols soil (B1 to B4, Fig. 3) showed high sediment load per area (Tables 2 and 5). Rivers B1 to B4 from the Bukavu basin represented alone 44% of the measured TSS indicating strong erosion in this highly populated area (Table 5). On average, the erosion of TSS seems stronger on the Congolese than on the Rwandan side of the catchment (Muvundja et al. 2009; Namugize 2009) probably due to more erosion as a result of the steepness of the slope together with the advanced degree of deforestation of this western part of the lake.

4.3.3. SWAT model output

The SWAT model applied by Rinta (2009) (section 3.3) provides load estimates for SRP, TP, $NH_4^+$and NO_3^-, which were compared to the monitored riverine load estimates provided by Muvundja et al. (2009). It was found that all four loads were overestimated by the SWAT model by factors of 15 (SRP), 2.2 (TP), 12 (NH_4^+), and 4 (NO_3^-), relative to the estimated loads listed in Table 5. Due to these large discrepancies, the non-calibrated SWAT output was used only to integrate for the non-observed rivers, by correlating the SWAT model with the measured loads. As Figure 7 shows, the SWAT model does not allow improving correlations with external parameters beyond the simple correlations shown in

Figures 5 and 6. It was therefore not possible to reduce the uncertainty of the estimates presented in Table 5.

4.3.4. Load extrapolations based on correlations

Beside the measurement errors, the extrapolation from the 21 sampled rivers by Muvundja et al. (2009) to the entire catchment causes additional uncertainties, as the procedure is arbitrary to a large extent. Therefore correlations between nutrient loads and catchment properties (such as population, catchment area, land use) were searched for to scale-up to the 4274 km^2 large river-active area.

Table 6 shows that in the Bukavu basin, the positive relationship between the loads and the population is more pronounced than with the catchment sizes except for silica (r^2 = 0.73), which is probably related to enhanced erosion. Also in the Goma basin (G), only SRP showed good correlation to the catchment size (Table 6). The Kalehe basin, however appears to be less disturbed by anthropogenic activities as its river loads show high correlations (r^2 = 0.91 to 0.98) with catchment size (Table 6).

Table 6: Results of regression analysis of discharge and nutrient loads for the three sub-basins

Basin name	Sampled area(km^2)	Basin area (km^2)	Para-meter[(1)]	Linear regression r^2 (-)	slope[(2)]	Intercept[(2)]
Goma (G)	403	417	SRP	1.00	0.01	0.59
Kalehe (K)	262	1340	Discharge	0.99	0.056	-0.77
			SRP	0.98	0.10	-2.22
			TP	0.91	0.28	-2.01
			NO_3^-	0.97	0.62	-9.94
			SRSi	0.98	17.0	-246
			TSS	0.93	120	-1874

(1) Results of linear regression for basins reported only when $r^2 > 0.9$ (p-value<0.05)

(2) Units for slope and intercept are according to the units of catchment area (km^2), discharge ($m^3\ s^{-1}$) and nutrients/TSS loads ($t\ yr^{-1}$).

Muvundja et al. (2009) used linear and power law least-square fits to express the loads as a function of the catchment area in order to extrapolate to all 127 rivers. The integration for SRP turned out stable: Linear extrapolation provides 121 t P yr^{-1}, power law extrapolation sums up to 85 t P yr^{-1}, the SWAT model adds to 86 t P yr^{-1}, whereas scale-up with the specific load yields 111 t P yr^{-1}. This example indicates, that by the extrapolation to the entire catchment, only another ~20% error is added. Also for Si the scatter was in a reasonable range. However for NO_3^-, the respective values are 3600 t P yr^{-1} (linear), 1000 t P yr^{-1} (power law), 1540 t P yr^{-1} (SWAT) and 1560 t P yr^{-1} (specific load approach).

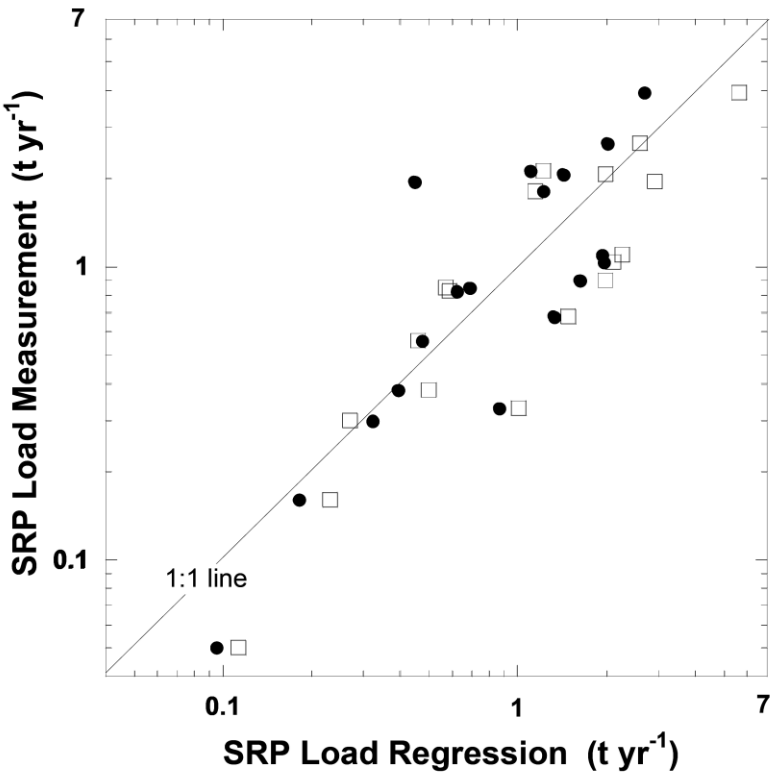


Figure 7. Comparison of riverine SRP loads from regression analysis (horizontal) versus measured SRP loads according to Table 4 (vertical). Black circles: SRP load = power law 0.077*[SWAT-SRP-load]$^{0.77}$.
Open squares: SRP load = double-linear as a function of population (Fig. 5a) and catchment size (Fig. 5b). The comparison illustrates that: (i) the SWAT model output as well as a linear regression are almost equally predictive, and (ii) the parameterizations are within a factor of ~2 relative to the measured loads (As SWAT is not sensitive to local point pollution, the 3 most affected rivers are not included in the analysis).

Table 7. Overall nutrient balance for Lake Kivu

Nutrient budget component	SRP $t\ P\ yr^{-1}$	TP $t\ P\ yr^{-1}$	Bio-P [(1)]	NH_4^+ $t\ N\ yr^{-1}$	NO_3^- $t\ N\ yr^{-1}$	DIN/TN	SRSi $t\ Si\ yr^{-1}$
Epilimnion input							
River inflow	111	1,650		370	1,550	1,920	23,300
Total atmospheric deposition [(2)]	118	2,940		2,220	1,230	3,450	1,340
Total external epilimnion input	**230**	**4,600**	**230** [(1)]	**2,600**	**2,800**	**5,400**	**24,600**
Internal load (upwelling)[(3)]	1,800	~0	1,800	18,500	0	18,500	29,600
Total input to epilimnion	**2,100**	**4,600**	**2,100**	**21,100**	**2,800**	**23,900**	**54,200**
Epilimnion output							
Outflow[(4)]	50	20	50	~0	60	**300**[(3)]	11,200
Gross sedimentation (epi export) [(5)]	0	1,060	1,060	0	0	6,500	12,700
Epilimnion net sedimentation[(6)]	0	40	40	0	0	200	1,500
Total output from epilimnion	**50**	**1,100**	**1,180**	~0	**60**	**7,000**	**25,400**

(1) Bio-P refers to P available for primary productivity, taking part in the lake internal P cycle. See section 4.4 for details.

(2) Nutrient loads from wet and dry atmospheric depositions estimated by Muvundja et al. (2009).

(3) The internal upflux determined in Pasche *et al*. (2009) is multiplied with lake area at 50 m depth (2078 km^{2,} the lake with islands excluded).

(4) Sarmento *et al*. (2006): surface concentration times outflow (Table 1)

(5) Gross sedimentation = export into deep waters; determined by the sediment trap measurements (average 90;130 and 172 m) times the lake area in 50 m depth (Pasche *et al*. 2010)

(6) Net sedimentation rate to sediment area in top 50 m (292 km^2), referring to Pasche *et al*. (2010)..Delta sedimentation at river mouths is not included.

4.3.5. Nutrient ratios, limitation and contribution to primary production

Referring to the nutrient loads given in Table 7, the SRSi:DIN:SRP molar ratio of riverine nutrient inputs to Lake Kivu is 232:38:1. Silica is obviously more available than any other nutrient species and Phosphorus is the most deficient nutrient in the nutrient fluxes from the river basin. Based on the molar ratios of the overall external inputs (Si:DIN:SRP = 192:52:1) and the lake internal concentrations (Si:DIN:SRP =238:12:1; Sarmento et al. 2006) and internal upwelling fluxes (Si:DIN:SRP = 18:22:1, Pasche 2009), the bio-available P input (SRP, Bio-P) is the limiting factor for the primary productivity in Lake Kivu.

Sarmento et al.(2008) measured the elemental C:N:P ratios of the phytoplankton of Lake Kivu, and these were 256.3, 9.6 and 26.8 for C:P, C:N and N:P respectively. Phosphorus was then found to be the limiting nutrient in Lake Kivu as the lake experiences severe and frequent Phosphorus deficiencies (Sarmento et al. 2008). Therefore in the condition of P-deficiency and based on the seston C:N:P ratios provided by Sarmento et al.(2008), an uptake of 2.62 g P m^{-2} yr^{-1} is needed to yield the 260 g C m^{-2} yr^{-1} (Sarmento et al. 2006), either a total uptake of 6209.4 t P yr^{-1} for the whole lake surface (2370 km^2, Table 1). Therefore, the annual load of 111 tones of phosphorus to the epilimnion from the river basin (Table 5) will induce a new primary productivity of ~5 g C m^{-2} yr^{-1}, either 2% of the overall annual production.

4.4. Nutrient balance in the epilimnion

The nutrient export via outflow (Table 7) is calculated by multiplying the lake discharge (Table 1) with the nutrient surface concentrations provided by Sarmento et al. (2006). With averages of 13; 70 and 2800 mg. m^{-3} of SRP, DIN and SRSi, respectively, outflow results to loads of 46; 251 and 11200 t yr^{-1}, respectively.

The estimated riverine nutrient inputs to the surface are compared with the upwelling from deep-water (Pasche et al., 2009) and the export off the epilimnion (top ~50 m) in Table

7. Export consists of (i) outflow (quantified as surface concentrations times discharge), (ii) epilimnion net sedimentation (net sedimentation rate times the epilimnion sediment area), and (iii) export production (gross sedimentation rate times the lake area at 50 m depth). As SRP and DIN are assimilated by the biomass and cause net export production in particulate form, the budget for P and N in adequate forms had to be performed. In accordance to Wüest (2007), for P, Bio-P is defined as the relevant quantity (Table 7), containing phosphorus which becomes bio-available (e.g. SRP plus the part of TP, which is easy bio-degradable biomass, such as algae). For N, DIN (NH_4^+ plus NO_3^-) is compared with the organic particulate nitrogen (TN), which is contained in algae biomass (Table 7, Yacobi and Ostrovsky 2008). The nutrient inputs to the epilimnion [external (riverine +atmospheric) deposition and upwelling] result to ~2,100 t P yr^{-1} (Bio-P); 24,000 t N yr^{-1} (DIN) and 54,000 t Si yr^{-1}. Overall external nutrient contributions to the total epilimnion inputs can be deduced from Table 7. These are 10% for SRP, 25% for DIN, and ~45% to Si, respectively. The nutrient losses from the epilimnion (also derived from Table 7) are respectively~ 2%, 29% and 47% of the total SRP, DIN and Si inputs to the epilimnion.

CHAPTER FIVE: DISCUSSION

5.1. Riverine Nutrient Fluxes

(a) Water balance

The water balance established for Lake Kivu, combining the water flows measured in this study with previous estimates of various sources for precipitation, evaporation and Ruzizi outflow (Bergonzini 1998, SNEL) results in an excellent agreement of input and output, both of 7.0 km^3 yr^{-1}. Despite this agreement, the overall uncertainty of the water budget is probably ~15 to 20%. This accuracy on precipitation and river flow estimated for 2006 to 2008 can be assumed to be representative for the last few decades of meteorological observations.

(b) Water quality of tributaries

Water quality depends much on the human population density and land use in the watershed. In the 5097 km^2 area of the lake Kivu catchment, the population has steeply increased to 2.1 million over last four decades (Census 2007a, 2007b, 2007c) yielding an average density in all the basin of ~400 inhabitants per km^2.

Among the physico-chemical parameters which have been measured in this study, temperatures are the most affected and were higher compared to values measured by Marlier (1954). The observations made in the present study (Table 3) are higher than what he found (temperature range of 12-15.5 in Mushuva with annual mean value of 14°C against an average of 19.4°C in this study). The major driving factor is the local microclimate, which is likely modified by deforestation in the catchment (recession of the bamboo forest of Mulume-Munene and Kalonge in Kahuzi-Biega National Park) but also some uncertainties due to the differences in sampling times. The other rivers follow the same trend.

Rivers in highly populated areas of Bukavu showed higher electrical conductivities than those of Kalehe (Fig.1). The pH values were slightly acidic and did not vary so much in these rivers (6.4 to 6.8) and are comparable to the observations made by Marlier (1954): 6.6 and 6.8 for the Lwiro (K1) and the Mushuva (B6), respectively. However in populated areas, the sampled rivers showed neutral to alkaline pH values (Table 3) probably due to organic pollution except some (e.g Nyabarongo).

Ammonia is a useful indicator of organic matter and concentrations should not exceed more than 100 µg L^{-1} in natural surface waters (Bartram and Balance 1996). Therefore, the sampled rivers of this study can be divided into two parts organically polluted rivers (the first 4 rivers of Bukavu, $N\text{-}NH_4^+$ concentrations > 100 µg L^{-1}, Bartram and Ballance 1996), Kawa (B1) being the most polluted; and unpolluted ones (B5 to G3, Table 4, $N\text{-}NH_4^+$ < 100 µg L^{-1}). The same trend is observable with Nitrate concentrations. But phosphorus concentrations did not follow the same trend.

Silica concentrations are not unusual as they vary from 1 to 30 mg L^{-1} in natural waters (Bartram and Ballance 1996). In this study the Rivers of Goma (Fig.1), for which drainage areas are localized in an active volcanic region, showed higher silica concentrations (Table 4) and this is a normal situation according to Bartram and Ballance (1996). Among all of these rivers, Mushuva and Mpungwe seem to drain water of the best quality according to the measured variables (lower temperatures, salinities, nutrients, good ranges of pH and higher velocities).

(c) Riverine nutrient inputs

The nutrient loads contain two dependencies: (1) on the population (Figs. 5, 6a), and (2) on the catchment size (Table 6), as exemplified for TP in Figure 6b. Muvundja et al. (2009) found that using functions with population and catchment area as independent variables, although reducing the scatter, do not significantly change the outcome of nutrient inputs (Fig. 7) meaning that the uncertainty of the riverine input estimates is in the range of 50%, and larger than statistical errors of the integrated fluxes alone (~25%). From the results of nutrient and population correlations, it was observed that the nutrient loads were foremost correlated to the population density and the relation to the catchment size is rapidly lost in densely populated areas.

Despite the intense land use (Fig. 3) and the high population density (Table 2), the nutrient input from the catchment (Table 7) is less than expected in comparison to other intensely used catchments. For instance, the annual loads for SRP (111 t P yr^{-1}) and DIN (1900 t N yr^{-1}), corresponding to areal-specific loads (~26 kg P km^{-2} yr^{-1} for SRP but only ~450 kg N km^{-2} yr^{-1} for DIN; Table 5), which are lower than in typical agriculturally used catchments in Europe (Reinhardt et al. 2005). In agrarian regions of Switzerland with mainly arable land and grassland, ~1500 kg N km^{-2} yr^{-1} of DIN originates from agricultural soils (Gächter et al. 2004) due to more intensive agricultural activities (fertilized farming).

These specific nutrient loads observed (Table 4), can also be compared to estimates of other East-African regions. Hecky et al. (2003) determined 55 kg P km^{-2} yr^{-1} (SRP) and 143 t km^{-2} yr^{-1} (TSS) by monitoring advanced agriculture-converted river catchments of two Lake Malawi tributaries (Rukuru and Lufira). These specific values are about twice as much as our estimates. Bootsma et al. (2003) estimated the mean riverine input of silica to Lake Malawi surface waters to 23 t Si km^{-2} yr^{-1} compared to 74 t Si km^{-2} yr^{-1}in our study, which is in a volcanic environment.

The whole catchment related TP:SRP ratio of ~16 is not unusual compared to other catchments where erosion dominates riverine TP (Müller et al. 2007). However the TP load is large compared to the TSS input of 85 kt yr^{-1} (specific load of 84 t km^{-2} yr^{-1}; Table 5), indicating that the P content in the TSS is unusually high (~0.006 fold). It implies that TP contains much more P than only the inorganic P in minerals (typically ~0.0007 fold; Müller et al. 2006) and therefore, some of the TP is obviously of organic matter origin. Meanwhile a substantial part of total phosphorus (in organic form) is expected to become bio-available phosphorus (Bio-P) after microbial decomposition into the lake (Table 7). In the volcanic active region of Goma and surroundings (Fig.1), rivers experienced higher silica specific inputs (Table 5). The reason why is the geological features of this part of the basin which is made of recent soluble volcanic ashes which end in the rivers and which also support the high dissolved solids in these waters.

(d) Molar ratios of nutrient inputs and primary production

The molar ratio of SRSi:DIN:SRP of riverine inputs was 232: 38: 1. The low P-content of the nutrient inputs confirms the strong P-deficiency in Lake Kivu as it has been discussed by many authors (Sarmento et al. 2008, Hecky and Kilham 1988, Kilham and Kilham 1990). The nutrient inputs from rivers are almost the same as from atmospheric deposition (Table 7, Muvundja et al. 2009). However the internal loading (upwelling) is the main source of nutrients to the epilimnion of Lake Kivu (Pasche et al.2009 &2010), which is in accordance with Kilham and Kilham (1990). Annual external loading potentially induced approximately 5% of the annual primary production. This estimate is relatively lower than that made for Lake Tanganyika (7-14%, Langenberg et al. 2003) most likely due to differences in land use and river basin development.

(e) Nutrient inputs, land use and anthropogenic activities

This study showed that there is a positive dependence between riverine nutrient inputs, specifically phosphorus, and: (1) absolute human population (Fig.6a), (2) river catchment size (Fig.6b). A study conducted in West African towns showed that approximately 0.5 to 1 kg P and 1 to 2 kg N per person per year are yielded from anthropogenic waste and about half of it is refound in rivers and streams (Drechsel et al. 2007). Results of this study showed the same trend (0.95 kg P and 0.81 kg N per person and per year respectively, in the densely populated region of Bukavu). Soil Erosion demonstrated by higher particle concentrations and water turbidity of rivers suggest that human activities such as agriculture are deteriorating the soil texture and consequently the water quality of the rivers in the catchment. In addition, Fig. 5a&b and Fig. 6a demonstrated that the water quality based on nutrient concentrations (ammonium and nitrate) and SRP loads depend much on the human population and the prevailing land use in the catchment. Lacks of hygienic facilities among the populations and waste water treatment justify the higher amounts of ammonia in waters. Hence, they are contributing factors to the degradation of water quality of these natural aquatic systems.

5.2. Plausibility Check of Nutrients Inputs

The accuracy and representation of the input estimates are not well-defined due to the enormous variability and heterogeneity of the processes and the catchment properties (Marlier, 1954). However riverine SRP loads estimated by SWAT compared to SRP loads measured indicated that SWAT model is almost equally predictive as a linear regression and that the parameterizations are within a factor 2 relative to measured loads (Fig. 7). Errors in such loading studies are often ~50%, except when a careful allocation of the sampling regime is undertaken (Moosmann et al. 2005). In this particular study on Lake Kivu, there was a fortunate possibility to compare the nutrient loading with (i) the internal flux analysis of Pasche (2009, 2010) and (ii) primary production estimates by Sarmento et al.(2006), both providing independent limits, by balancing the nutrient for (a) the epilimnion and (b) for the entire lake.

The largest terms of the epilimnion balance, are internal loading (upwelling) and gross sedimentation of the biomass (sinking plankton). Although it has been observed in other large East-African lakes (Kilham and Kilham, 1990), that internal loading is the substantial source of primary production, the internal loading is especially important for the epilimnion of Lake Kivu, as deep sub-aquatic springs cause a continuous advective uplift of the nutrient-rich deep-water. Therefore, the internal loading estimated by Pasche et al. (2009), exceeds all the external input for all three nutrients (Table 7). The contribution of external loads is 10% for SRP, 25% for DIN, and ~45% to Si, respectively compared to the epilimnion input. These results indicate that the major source of nutrients for Lake Kivu is internal nutrient cycling. These nutrients are loaded to the epilimnion (mixolimnion) by the upwelling. The output/input ratios of nutrients (see results section) show that almost the SRP inputs are used by primary producers whereas 29% and 47% of DIN and Si inputs are lost from the epilimnion via outflow, gross sedimentation and epilimnion net sedimentation pathways. These results support the hypothesis according to which the recent methane increase in Lake Kivu might have been caused by the introduction of *Limnothrissa miodon* which affected the whole lake food web and the nutrient cycling of higher sinking organic materials (Pasche et al. 2009). An increase in internal nutrient loading (upwelling) caused by more water inputs from sub-aquatic springs may have also exacerbated the change in nutrient cycling, the driving force being the increase in rainfall recorded since the 1960s in the region (Pasche et al. 2010).

Gross sedimentation, the other large component in the epilimnetic nutrient balance, is much more important than the outflow (Table 7). It is suggested that the estimated SRP inputs of 230 t P yr^{-1} is somehow an underestimate for Bio-P input to the lake, as some of the 4,600 t P yr^{-1} of TP will become bio-available (Wüest, 2007). The larger Bio-P input than output in Table 7 is not a contradiction, as some of the settling material is obviously digested -occurring even in much colder lakes- before being collected from the sediment traps.

Also the N budget indicates additional N removal from the settling organic matter, not showing up in gross sedimentation. Most likely additional sinks are denitrification and the decomposition of the sinking organic material. Despite the larger DIN-input to the epilimnion than the export from the epilimnion, the DIN input of 5,400 t N yr^{-1} is not expected to be an overestimate.

The most accurate input estimate is that of SRSi, as relative contribution of the external load (40,000 t Si yr^{-1}) does not differ much to the total input to the epilimnion (47,000 t Si yr^{-1}).

5.3. Riverine nutrient inputs and primary production

Based on the molar ratio provided above by Sarmento et al. (2006) and considering P as the limiting nutrient (Sarmento et al. 2008), the annual load of 111 t P yr^{-1} of SRP (Table 7) from the riverine basin may induce a new primary productivity of ~5 g C m^{-2} yr^{-1}, either 2% of the overall annual production considering P as the limiting nutrient**.** However, the contribution of the overall external inputs counts for 5% of the total annual production. These contributions of external nutrient inputs fall in almost the same ranges to that of Lake Tanganyika where the dissolved soluble phosphorus load from external sources could support roughly 3 to 5% of the annual production with P-limiting conditions and 7 to 14 % for N-limiting conditions (Langenberg et al. 2003).

CHAPTER SIX: CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

In this study the focus was devoted to the assessment of the riverine nutrient fluxes to Lake Kivu. The motivation is driven by (a) the extraordinary aquatic ecosystem surrounding the methane use and the introduction of the sardine L*imnothrissa miodon* and (b) the concerning increase of anthropogenic activities in the catchment.

The measurements of water quality parameters indicated that water quality of rivers in the Northern basin is not yet so disturbed compared to that of the densely populated southern basin

Although, the uncertainties of individual load components are in the usual range of up to 50%, the following conclusions can be made:

(i) Lake Kivu river basin is made of small rivers and streams (0.9 to 2.7 $m.s^{-1}$ of discharge). The water quality of some rivers of the Lake Kivu basin is already deteriorated. These include those draining urban area (in Bukavu) but other rivers drain rural areas (Kalehe and Goma) are still experiencing good natural quality water. Rivers in the Northern part of the basin are highly salty while those of the southern part are typically freshwater bodies. Rivers situated in between the North of Bukavu and the North of Kalehe are still draining good water quality for resource uses.

(ii) The molar ratio of the riverine nutrient inputs Si:DIN:SRP is 232:38:1. This study agrees that Phosphorus is the limiting nutrient for algal production. However, the SRP input of 230 t yr^{-1} is considered as lower limit of the Bio-P load, as some of the TP was found in easy-degradable organic waste from polluted rivers. Riverine loads and atmospheric deposition contributes equally to the SRP input.

(iii) Ammonia and nitrate contribute equally to the DIN input of 5,400 t N yr^{-1}, although from the different sources: ammonia enters via rainwater; nitrate is supplied by river flows.

(iv) Silica input has specific characteristics with ~45% to the surface, ~55% to the deep-water by the sub-aquatic sources. The deep-water inputs contribute indirectly to the productive surface layer, where Silica eventually appears via lake internal upwelling (internal loading). River fluxes represent ~95% of the external inputs (24,600 t Si yr^{-1}) and the atmospheric deposition of silica to the lake surface is negligible.

(v) This study shows that the higher the anthropogenic disturbance in the catchment, the weaker the correlation between the riverine nutrient loads and the catchment size. However a positive dependence of SRP load and NH_4^+ concentrations to the population in the catchment was found. Nitrate concentrations increased with the agricultural land use intensity.

(vi) For all three nutrients, the riverine inputs contribute the least to the total epilimnetic nutrient input (uptake) in the epilimnion of 111 t P yr^{-1}; 1,920 t N yr^{-1} and 23,300 t Si yr^{-1}, respectively: ~5% (Bio-P), ~8% DIN and ~43% (SRSi). The major contribution to the epilimnion input is the upwelling (internal loading) as predicted by Kilham and Kilham (1990).

(vii) The recently measured primary production of ~260 g C m^{-2} yr^{-1} (Sarmento et al. 2006) is consistent with the observed inputs. The Riverine inputs of nutrients induce a new primary productivity of ~5 g C m^{-2} yr^{-1}. Human influence on lake pelagic water is still small and a potential anthropogenic increase of external nutrient fluxes cannot be responsible for the reported increase in the methane production of the last decades.

6.2 Recommendations

Based on this study, the following recommendations are of great importance for east-african rivers and lakes:

(i) An effort should be made by policy-makers to promote sustainable practices in the lake basin landscape and watershed management to combat erosion and pollution;

(ii) There is high need of getting the urban solid waste and waste waters treated before they are discharged into rivers and lakes in order to prevent water pollution and eutrophication;

(iii) It is also necessary to design and implement permanent lake and river basin monitoring programs to gather more data allowing model development for future trends and management of water resources.

(iv) The removal of nutrients in water bodies is expensive and unaffordable by low income countries. Therefore, given that water quality issues are becoming more and more challenging in East-Africa including Uganda, there is a need of conducting such baseline studies on water bodies of the sub-region to generate enough data on the current nutrient loads and budgets in order to assess the prevailing water quality, and prevent its further deterioration by acting on watershed management practices.

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