Ministry of Water and Irrigation
International Lake Environment Committee

11TH WORLD LAKES CONFERENCE
NAIROBI, KENYA, 31 OCTOBER TO 4TH NOVEMBER 2005

PROCEEDINGS VOLUME II

EDITED BY:

Eric O. Odada, Daniel O. Olago, Washington Ochola,
Micheni Ntiba, Shem Wandiga, Nathan Gichuki and Helida Oyieke

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FOREWORD

The International Conference on the Conservation and Management of Lakes (World Lake Conference) is a biennial conference co-organised by ILEC and a local host. Previous conferences have been held in Japan, USA, Hungary, China, Italy, Argentina and Denmark. In 2005 the Conference moved to Kenya and was held in Africa for the first time. This conference was held from 31 October to 4 November 2005 in Nairobi, Kenya. The organisers of the conference were the Ministry of Water and Irrigation, Kenya; the International Lake Environment Committee Foundation, Japan; and Pan African START Secretariat, Nairobi, Kenya.

The main theme of the Conference was "Management of Lake Basins for Their Sustainable Use: Global Experiences and African Issues". The principle objective of the 11th World Lake Conference was to bring together diverse groups of people and organizations dealing with lakes to provide a rich forum for exchange of knowledge and experiences on the management of lakes in general and African lakes in particular, noting that African lakes contribute significantly to socio-economic development of the African region but are subject to high levels of rapid population growth, urbanization, industrialization, mining development, growth of irrigated agriculture, and impacts of climate change. These pressures have altered ecosystem processes and resulted in several threats on the lakes including: loss of biodiversity, over-fishing, eutrophication, proliferation of invasive weeds, siltation, toxic contamination and over-abstraction of water. To ensure their sustainable use, these important but fragile ecosystems need to be managed properly. The Conference reviewed progress on ongoing lake basin initiatives as well as set future goals for lake basin management.

The Conference Proceedings have been produced in two volumes: the first largely captures the socio-economic aspects of lakes and lake management, while the second volume deals more with the biophysical aspects of lakes and their basins. They offer a very rich and diverse range of new information and detail in all these aspects, and cover all parts of the globe. From them, very many useful lessons can be drawn for the better and sustainable management of the world's lake basins, from the smallest to the largest. Below, a synopsis of the various themes under which the papers are grouped are presented, and these themes can be tracked by the reader with reference to the Table of Contents. The Statement of the 11th World Lakes Conference is also presented in full below.

The Editors would like to acknowledge all the effort put in by the numerous authors and reviewers without whom this volume would not have come to fruition. We would like also to acknowledge the critical funding and other support provided by the Ministry of Water and Irrigation, ILEC and the all the co-sponsors of the 11th World Lakes Conference that has enabled the production of the Conference Proceedings.

SYNOPSIS OF THE VARIOUS SESSIONS THEMES

Session 1: Governance and Water Resources Management

1A Governance and lake management; Lake management and corruption in the water sector
1B Lakes, IWRM and the Millennium Development Goals
1C Management of transboundary lakes; treaties and agreements on transboundary basins

This session focused on governance and management of lakes and their basins, their use and the related policy, economic and political implications from local, regional as well as international scales.

Session 2: Scientific Research and Monitoring

2A Scientific research and monitoring; technology and lake management
2B Harmonisation between research and sustainable management; education, research and training

Papers on scientific research, monitoring and improved technologies and the manner in which they have contributed to better and sustainable lake management were presented. However scientific, gaps that exist need to be bridged and mainstreamed in national and international policies and activities.

Session 3: Poverty Reduction and Cross-Cutting Issues

3A Poverty reduction and cross cutting issues
3B Lakes, sanitation and health issues
3C Lakes and agriculture/food security
3D Economic returns from lakes and their basins

Poverty is both a cause and result of degradation of lakes. The session dealt with poverty reduction as a means to enhancing environmentally sustainable livelihoods, particularly in areas where the population is largely dependent on land/lake-based resources. It also addressed natural, socio-economic, health, development and political factors that might enhance or mitigate poverty levels.

Session 4: Effects of Emerging Issues on Lake Management

4A Lakes and persistent organic pollutants; Emerging issues and lake management
4B Climate change and lake management

This session examined critically the emerging threats to lakes as a consequence of disasters, persistent organic pollutants and climate change. In a wider sense the session explored issues related to emerging disaster risks and vulnerability, and the new paradigms in Lake Management and research that need to be considered and effected in view of these emerging changes.

Session 5: Public Participation in Lake Management

5A Public participation, local communities and lake management; Confidence building and stakeholder participation in lake management
5B Gender issues in lake management
5C The role of education; awareness raising in the management of lakes

This session addressed issues of public participation in lake management from both a holistic point of view where all stakeholders are included; to a group or community focused approach, including aspects such as gender, youth, education and awareness building in participatory lake management.

Session 6: Lake Basin Initiatives

6A International Lake Basin Initiatives; Development aid and lakes
6B Lakes and water for African cities; Lakes and transportation issues
6C The role of international organisations and NGO’s in lake management

This session dealt with experiences and lessons learned in the many lake basin initiatives that have been carried out throughout the world, including the role of international organizations and NGO’s in Lake Management. It also emphasised lakes as water sources for African cities and as important transportation media.

Session 7: The Lakes Ecosystem Health
7A Lake ecosystem health
7B Lakes and biodiversity; Lakes and fisheries
7C Lake pollution, including eutrophication; Impacts of industrial and agricultural development on lakes
7D Comparative limnology: tropical verses temperate

This session focused on ecosystem health, particularly the lakes vitality, resilience, as well as their functional and structural components. It also highlighted the role of human activities that either promote or degrade the health of lake ecosystems.

Session 8: Threats to Lakes: With Special Emphasis on the African Realities
8A Lakes and land use change issues
8B Threats to African lakes
8C Invasive species

This session encompassed two major threats to lakes in general, i.e. land use change and invasive species, as well as other threats specific to African lakes.

Session 9: Cultural Issues
9A Cultural Traditions and Lakes
9B Modern Lifestyles and the Health of Lakes

This session included all aspects of culture that influence perception and uses of lakes and natural resources within the lake basins. Aspects of interest included cultural values and beliefs, traditional modes of environmental stewardship, resource use and gender, and modern lifestyles and their effects on the environment.

Special Plenary Sessions: There were several such special sessions. ***The Youth and Young Water Professionals Conference*** gave voice to the concerns and actions of the youth with respect to the conservation and sustainable management of lake basins. Young water professionals discussed and shared with the youth their experiences in the management of lakes and the provision of water supplies and related services in diverse lake settings. Major issues such as the implementation of integrated land and water management programmes, as well as obstacles to the implementation of such programmes, were discussed. The strengthening of capacity of water professionals to sustainably manage the lake basins was an important consideration. ***The Mayor's Special Session*** allowed municipality managers to share experiences and lessons learned in relation to the urban lakes that are under their management or jurisdiction. ***The World Lakes Vision*** brought together various stakeholders to share experiences and lessons learned in managing lakes for their sustainable use. ***The Lake Victoria Environmental Management Programme*** and the ***Nile Basin Initiative*** special sessions reviewed progress on ongoing activities and set future goals for the lake management programme. The ***Launching of the Lake Basin Management Initiative Report*** prepared by ILEC, World Bank (WB) and Global Environmental Facility (GEF) was officially launched at the 11th World Lakes Conference, formally placing the lake agenda on the global strategy. Finally, the ***Ministerial Round Table Discussion*** which was convened by Kenya Ministry of Water and Irrigation and UNEP, provided a rich forum for the exchange of knowledge and experience on the management of lakes in general and African lakes in particular, and was attended by Ministers and high –ranking government officials from around the world.

STATEMENT OF 11TH WORLD LAKES CONFERENCE 4 NOVEMBER 2005: NAIROBI, KENYA

Lakes, both natural and artificial, provide a wide range of important values, such as water resources and fisheries, both to sustain human livelihoods and to support economic activities. As particular wetland features, they also provide habitat for biodiversity, buffering capacities against hydrologic and climate fluctuations, and receptor functions for inflowing materials collected across their basins. Small or large, fresh or saline, ancient or transient, they are among the most dramatic and picturesque features of our global landscape, offering important aesthetic and spiritual values to many. On the other hand, they are among the most vulnerable ecosystems on earth, and are easily subjected to a variety of stresses originating from within and outside their drainage basins.

In view of their importance to human well-being and ecosystem health, as expressed in the Nairobi Statement of the 11th World Lake Conference, Kenya, October-November 2005, the management challenge of lakes and their basins must be addressed, recognizing that the future of lakes depends on our understanding and appreciation of their wider connections:

- With the surrounding landscape and human activities taking place on it;
- With the linking water system of rivers, groundwater, and wetlands;
- With the winds that carry nutrients and contaminants in from far distances; and
- With the rapid human changes to the Earth's atmosphere that are driving climatic instability.

The World Lake Vision, launched at the 3rd World Water Forum in Japan, March 2003, and the lessons learned from the Lake Basin Management Initiative launched at the 11th World Lake Conference held in Nairobi, Kenya on 31 October 2005, highlight these issues and suggest ways to achieve sustainable use of lakes and their resources, including a lake basin governance framework that involves;

- Adequate *institutions* implementing change;
- Efficient, effective and equitable *policies;*
- Meaningful *participation* of all involved stakeholders;
- *Technical measures to* ameliorate certain lake problems;
- Appropriate *information* about current and future conditions; and
- Sufficient *financing to* allow all the above to take place.

For the above governance framework to be successfully pursued, we must, first of all, recognize the primary importance of the people who directly use lake resources and therefore immediately experience damaging consequences of their misuse or degradation. These lake dwellers, both men and women, carry the cultural memory of the community and the lake through time, often having the best knowledge about the underlying causes of lake problems and viable solutions. This long-term perspective is essential because lakes have long memories when abused, and harbor many secrets in their complex dynamics. For these reasons:

- We must consider local knowledge and insight in making management decisions; and
- We must use available resources to build institutional capacity and scientific understanding at the community level, and to enhance the power of local people to find solutions, thereby bridging the gap between scientists, decision-makers, and society.

At the same time, however, local people on the front line must assume responsibility along with power, since local behaviour is often the source of damage to lakes. They must recognize that a healthy lake comes at a cost, and also that an unhealthy lake has its costs. As pointed out in the Kampala Declaration of the 9th Ramsar Convention, 12th November 2005, new innovative efforts to assign economic value to ecosystem services derived from lakes represents a powerful tool for identifying and justifying lake management interventions directed to conserving these important aquatic ecosystems; the accompanying increasing loss of lake - and wetland - dependent biodiversity also has major implications for local well-being and indigenous economies in many parts of the world; and that this degradation is occurring most rapidly in locations where populations are experiencing the greatest increases, thereby requiring increasing quantities of freshwater to meet human and ecosystem needs, especially being the case in developing countries, which often have limited availability to technological solutions to these problems.

National institutions also are vital for fostering awareness, promoting participation, and bringing together diverse interests within lake basins. When capable and effective, they provide the arena for developing broad management efforts that consider the lake basin as a whole, and its broader connections with the linking water systems and atmospheric influences. They also provide a forum for addressing the often conflicting needs of those who inhabit lake basins and who depend on lake resources. Without such an overarching framework and comprehensive perspective, there are few means for resolving conflicts over water or lake resources, or for integrating local efforts to maintain lake health into national programs and development plans. In setting these policies, national authorities must consider lake communities, as well as ensuring that the widest range of interests dependent on lakes enjoys their benefits. Experience around the world demonstrates further that effective lake management requires a cross Sectoral approach directed to maintaining wetland ecosystem services within the context of achieving sustainable development and improving human well-being.

National leaders also act in the international arena, where they can illuminate problems - -such as transboundary management, long distance air pollution, and climate change - -, and press for viable solutions.

International collaboration, including assistance from technical collaboration and funding programs, can provide a vital impetus for lake management efforts. Experience around the world shows that international technical collaboration and funding can plan a catalyst role in the management of lake basin activities, although in the longer term, individual citizens, local communities, and local and national governments must work to ensure that the ultimate goal of sustainable use resources is fully appreciated and pursued by all stakeholders, while also introducing a variety of policy tools, including innovative financing approaches.

These lake management experiences, which scientists and managers have gathered, analyzed and synthesized, provide important lessons for sustaining the health of both natural and manmade lakes that provide water for humans and nature, with one of the major lessons being that, where lakes exist, lake basin management is critical for sustainable development and responsible economic growth. It is imperative to embrace these lessons and build on them if we are to meet our pressing water needs in the decades ahead, as emphasized in the high-level African Water Ministerial Dialogue at the 11th World Lake Conference, which recommended that integrated management of lake basins be a long-term element of government and public priorities, planning and financing processes, habitat and biodiversity programs, and economic and development programs. The Ministerial Dialogue also recommended that the United Nations establish an International Year of Lakes, which would provide a global forum for the dissemination of these study results as guidance for lake stakeholders and decision-makers.

Over recent decades, we have been slowly learning how to manage the interactions between human activity and these living water systems. This experience underscores not only the key role of lakes in integrated water resources management, IWRM, but also the key contributions from the experience of Integrated Lake Basins Management or ILBM that provides many subtle, but crucial, dimensions of basin system management that have generally been neglected in the past. Indeed, the core elements of virtually every lake basin system are the flowing water with impounded water pools; namely, the lakes of natural or artificial origin that serve as the focal reserves of resources, as well as the barometers of basin vulnerability. Water also underpins virtually all the Millennium Development Goals, as recognized in the commitments of 170 heads of states and governments at the 2000 Millennium, and subsequently reinforced by world leaders at the 2005 World Summit, and achieving them depends on mainstreaming lakes and wetland issues into the overall global water agenda.

As fossil footprints in ancient lake bends dramatically testify, the rich resources of lakes were a magnet for early humans tens of thousands of years ago in Africa, and have continued to be so throughout human history to our own day. The challenge now facing us is to preserve the world's lakes, these complex life-supporting ecosystems that contain more that 90 percent of all the liquid freshwater on the earth's surface, so they can continue to provide physical and spiritual support for the generations that follow us.

The physical limnology of Winam Gulf and Rusinga Channel of Lake Victoria during April-May and August of 2005

Jason P. Antenucci[1], Ryan Alexander[1], Greg Attwater[1], Tom Ewing[1], Sheree Feaver[1], Jorg Imberger[1], Patrick Khisa[2], Carol Lam[1], Henry Njuguna[2], José R. Romero[1], and Kenji Shimizu[1*]

[1]Centre for Water Research, University of Western Australia, Crawley, WA, 6009, Australia
[2]KARI/LVEMP Coordination Office, Re-Insurance Plaza, 2nd Floor, Kisumu 40100, Kenya
[1]Email: antenucc@cwr.uwa.edu.au

Abstract

Winam Gulf is a large (surface area ~ 1400 km^2) and shallow (<20 m) bay of northeastern Lake Victoria with only one connection to the open lake through Rusinga Channel. To understand the exchange dynamics between Winam Gulf and the offshore waters of Lake Victoria and the hydrodynamics of the region, field studies were carried out from Apr. 22-May 4 and Aug. 5-16 of 2005. A meteorological station (shortwave, total radiation, air temperature, relative humidity, wind speed and direction), thermistor chain (0.75 m vertical resolution) and ADCP (40 cm vertical resolution) were deployed in Rusinga Channel in a depth of 20 m. Similarly, at an offshore station in northeastern Lake Victoria another thermistor chain was deployed in a water depth of 40 m along with wind speed and direction sensors.

Over both field campaigns the exchange dynamics through Rusinga Channel behaved similar to a tidally-driven system with surface level fluctuations of between 5-15 cm at the ADCP location, and much larger excursions at the eastern end of Winam Gulf. In general, these surface level movements led to barotropically driven flows into the Gulf during rising surface levels and currents towards the open lake during falling lake level. The frequency of these currents was found to vary between 6 and 12 hours and current speeds ranged from 10-50 cm s^{-1}. Field data and ELCOM simulations indicate that despite the high current velocities in the channel the net exchange is low due to the oscillatory nature of the forcing. This implies that the Gulf is relatively decoupled from the main lake.

Key words: Lake Victoria, Exchange flow, Flushing times

Introduction

The Winam Gulf region of Lake Victoria is an important regional resource to Kenya (Figure 1). In recent years, particular problems have appeared in the Gulf due to the presence of water hyacinth, which has had the impact of reducing fish catches and potentially impacting directly on human health (Opande et al 2004, Williams et al 2005). The nutrient status of the Gulf and the main lake was investigated by Gikuma-Njuru and Hecky (2005), who found the Gulf to be well-oxygenated but potentially light limited for phytoplankton. They also found the Gulf likely to act as a source of nitrogen to the main lake, but were unable to determine whether the Gulf was a source or sink of phosphorus for the main lake. The role of Winam Gulf in the management of the greater Lake Victoria, and the impact of Lake Victoria on the Gulf, remains somewhat unknown.

To help in the understanding of these problems, a study was commissioned by the Kenya Agricultural Research Institute (KARI) and the Lake Victoria Environmental Management Project (LVEMP) titled "Pilot Study of the Hydraulic Conditions over Rusinga Channel and Winam Gulf of Lake Victoria". The study was conducted by the Centre for Water Research (University of Western Australia) in conjunction with scientists and staff of KARI, LVEMP and the Kenya Marine and Fisheries Research Institute (KMFRI).

The study objectives were:

- To establish the major patterns of water circulation over the Rusinga Channel and the factors controlling these patterns
- To assess the influence of the Rusinga Channel on the mixing between Winam Gulf and the main lake
- To determine whether other 'bays' constitute comparatively 'dead' zones
- To improve on existing estimates of the mean hydraulic retention period of the lake

We discuss the methods used to achieve these objectives and the preliminary findings of this investigation in this paper.

Materials and methods

In order to address these objectives, two major field campaigns were conducted. The first was carried out between April 22 – May 4 and the second between August 5 – 16 2005. The timing of the first experiment was set to coincide with the wet season, whereas the second experiment was timed to coincide with the dry season.

Each experiment required significant logistics planning and organization. A substantial amount of time and effort was spent on ensuring the participation of the local riparian communities of the study region. This involved meetings conducted between local experts in community participation and the scientists conducting the study. The experimental work was based in Mbita at the ICIPE research station.

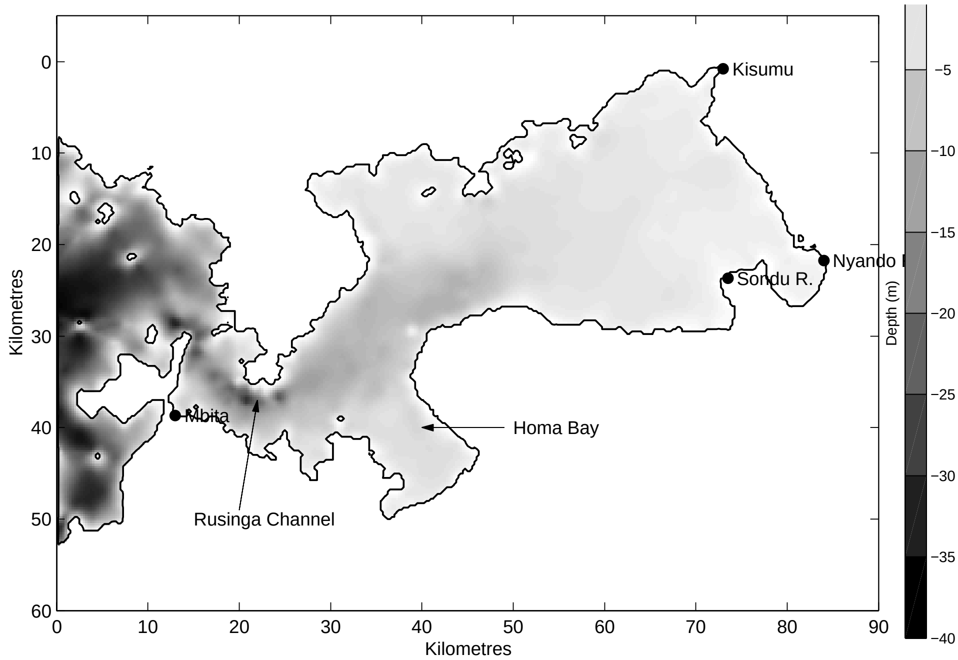

Figure 1: Bathymetric map of Winam Gulf showing the locations of Kisumu city, the Mbita township, the Rusinga channel, the Homa Bay region, and the locations of the Sondu and Nyando Rivers.

For each experiment, three moorings were deployed. At an offshore station (T1), a thermistor chain measuring the vertical temperature profile every 1 minute was deployed along with a wind speed and direction sensor. For the second experiment, this station was also fitted with a pressure sensor to determine the water level. In the channel, a second thermistor chain (T2) was deployed that included a full meteorological station measuring shortwave radiation, net radiation, air temperature, relative humidity, wind speed and direction (Figure 2a). At the same location an acoustic Doppler current profiler (ADCP) was deployed to determine the vertical structure of the currents flowing through the channel (Figure 2c). During the second experiment, a pressure sensor was also installed in Kisumu Bay to measure the water level.

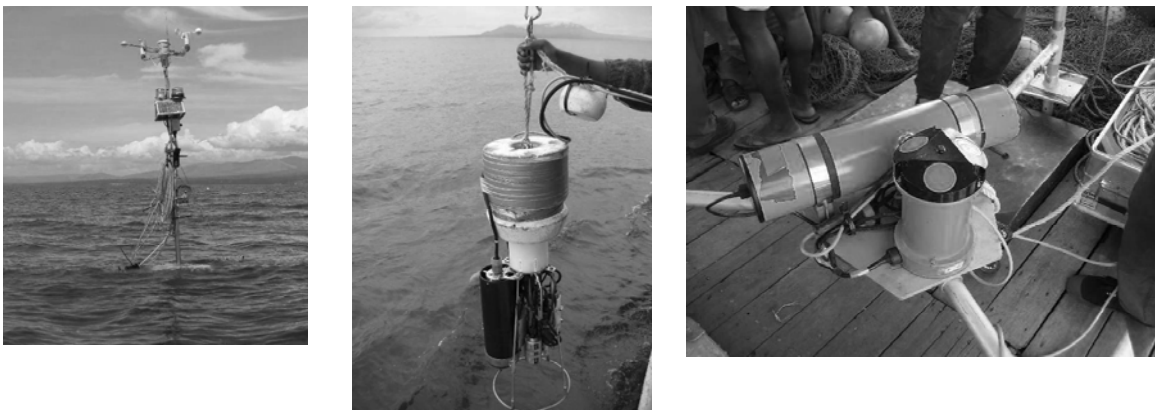

Figure 2 Instrumentation deployed in Winam Gulf during the experiments. From left to right: Lake Diagnostic System consisting of 20 high frequency thermistors, solar radiation, net radiation, air temperature, relative humidity, wind speed and wind direction; F-probe profiler consisting of conductivity, temperature, depth, pH, turbidity, dissolved oxygen, chlorophyll a and coupled with a BBE Fluoroprobe measuring 4 response regions of phytoplankton; ADCP measuring horizontal velocities and variation in water height. These fixed point measurements were complemented by profiling conducted on the RV Utafiti.

The main instrument used was the F-probe profiler (Figure 2b), which measures conductivity, temperature, depth, pH, dissolved oxygen, turbidity and chlorophyll a at approximately 2cm intervals in the vertical. This instrument was deployed in a free-falling mode at numerous stations in the channel and Gulf (Figure 3). Also incorporated into this probe was a BBE Fluoroprobe (http://www.bbe-moldaenke.de/english/fluoroprobe_e.html) , which measures chlorophyll fluorescence in four different algae classes and can be used to differentiate between algal groups in-situ. A microstructure profiler was also used, consisting of high vertical resolution (1mm) measurements of temperature, conductivity and depth. This was used to determine turbulence and mixing characteristics in the Gulf.

Results

The pattern of stratification due to temperature at moorings T1 and T2 is shown in Figure 4 and Figure 5 for the two field experiments. During the April/May experiment (Figure 4), the offshore waters of Lake Victoria demonstrate a strong daily heating pattern with surface temperatures reaching in excess of 28°C before cooling to approximately 27°C during the night. There was persistent stratification, with the water at 40 metres depth typically 1-2°C cooler than the surface waters. From April 27 onwards, however, this pattern changed markedly in the offshore station (T1), with a large volume of cold water (< 25.5°C) upwelling from the deeper parts of the lake into the waters offshore of Winam Gulf. The surface temperatures also cooled to approximately 26.5°C. In the shallower channel region, the water was characterized by heating at the surface during the day and mixing to the bottom at night. During this period the upper 15 metres of the water column in the Gulf was approximately 0.5°C cooler than the main lake, though this temperature differential was eliminated as the system cooled towards the end of the experiment.

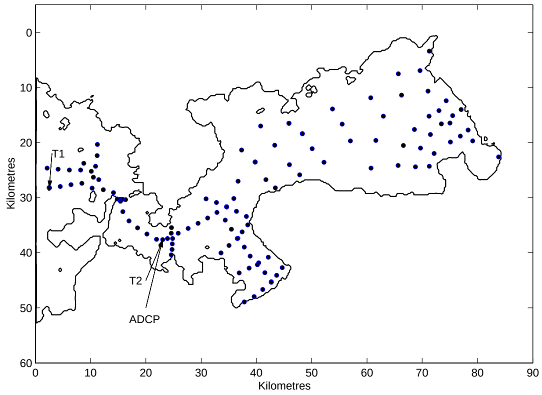

Figure 3. Sampling locations within the study area.

The location of the T1, T2 and ADCP station are shown. Samples were taken using the F-probe profiler at each location arranged in transect formations in order to capture the dominant spatial and temporal patterns in physical, chemical and biological parameters. During the second experiment (Figure 5), the lake was up to 2°C cooler than during April (note the change in scale between Figure 4 and Figure 5). The surface layer in both the offshore (T1) and channel (T2) regions showed similar characteristics to during April 2005, where the persistence of stratification was more evident in the offshore regions than in the channel. As with the first experiment, during this period the upper 15 metres of the water column in the Gulf was approximately 0.5°C cooler than the main lake, though this temperature differential decreased towards then end of the experiment to less than 0.2°C.

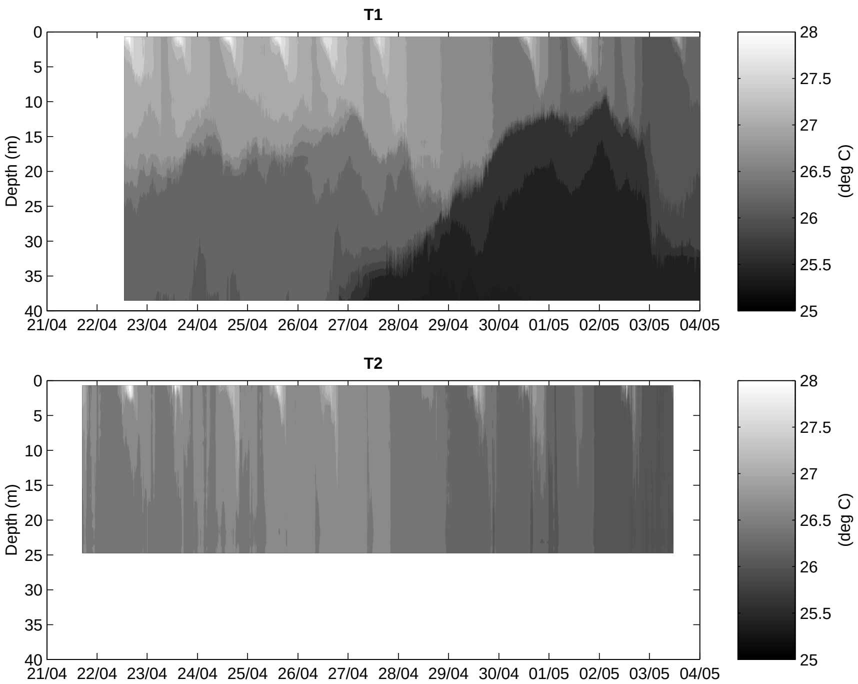

Figure 4. Temperature profiles collected in the offshore (T1) and channel (T2) locations during the April experiment. Depth is the vertical axis, time is the horizontal axis, and the shading represents temperature.

Measurements of currents are presented in Figure 6 along with the water level difference between station T1 and T2. The depth averaged velocity in the channel is highly oscillatory in nature, varying between zero and 15-30 cm/s over approximately a 6 hour cycle. The direction of flow is either into the Gulf (a direction of 90°) or out of the Gulf (a direction of 250°). The flow in and out of the gulf is dominated by the water level difference between the gulf and the main lake. We highlight one period on day 226 (14 August 2005) where the current increases rapidly from near zero to 35 cm/s and is flowing into the gulf from the main lake. The water level difference between T1 and T2 shows a strong gradient with the water level higher at T1 than at T2. This pressure gradient drives the flow into the Gulf from the main basin. The next cycle immediately after this shows a flow reversal with flow out of the Gulf into the main lake associated with a water level pressure gradient from the gulf into the main lake. It appears this is the dominant transport mechanism for flushing water into and out of the gulf from the main lake.

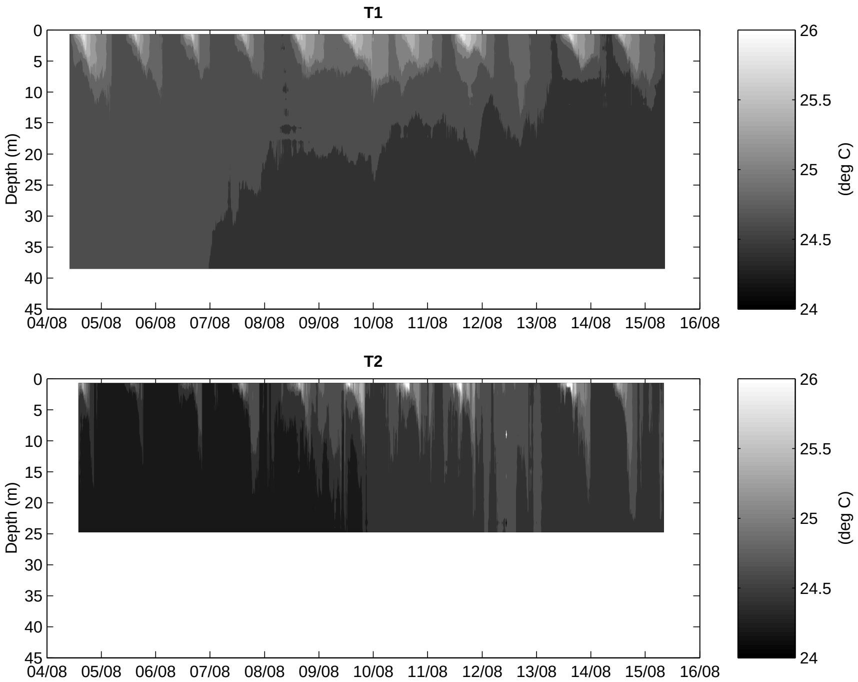

Figure 5. Temperature profiles collected in the offshore (T1) and channel (T2) locations during the August experiment. Depth is the vertical axis, time is the horizontal axis, and the shading represents temperature.

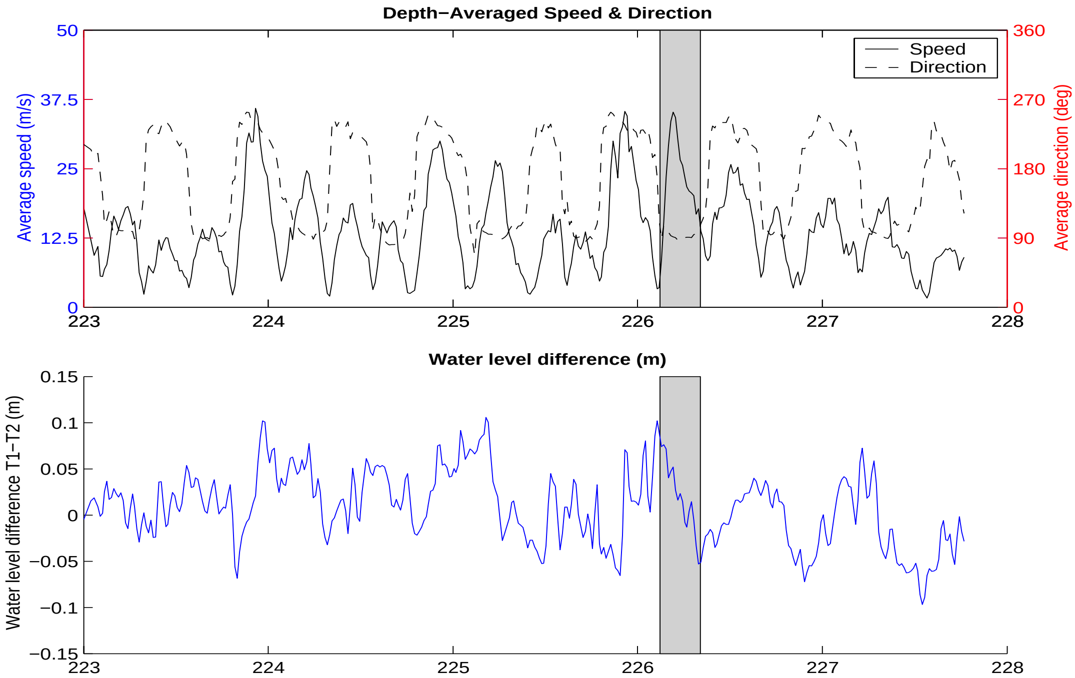

Figure 6. Depth-averaged speed and direction (upper panel) and water level difference between T1 and T2 (lower panel). The shaded area is discussed in the text.

We also present data for the full record available (Figure 7). These data demonstrate the variability in the flow regimes in the Rusinga Channel. Up until day 222 (August 10), the flow shows a 24 hour cycle of one period of water flowing into the Gulf, followed by flow out of the gulf. This changes dramatically on day 222 (August 10), where the flow now changes direction twice in each day, with two peak periods of flow into the gulf and two peak periods of flow out of the gulf. There is one day earlier in the record (day 219) where this pattern also occurs. These data demonstrate that even over a relatively short period (10 days in total), the nature of the flow regime into and out of the gulf can change dramatically. The factors causing the water level changes in the main lake and the reason for the change in the frequency of the currents in Rusinga Channel on August 10 remain subjects of further investigation.

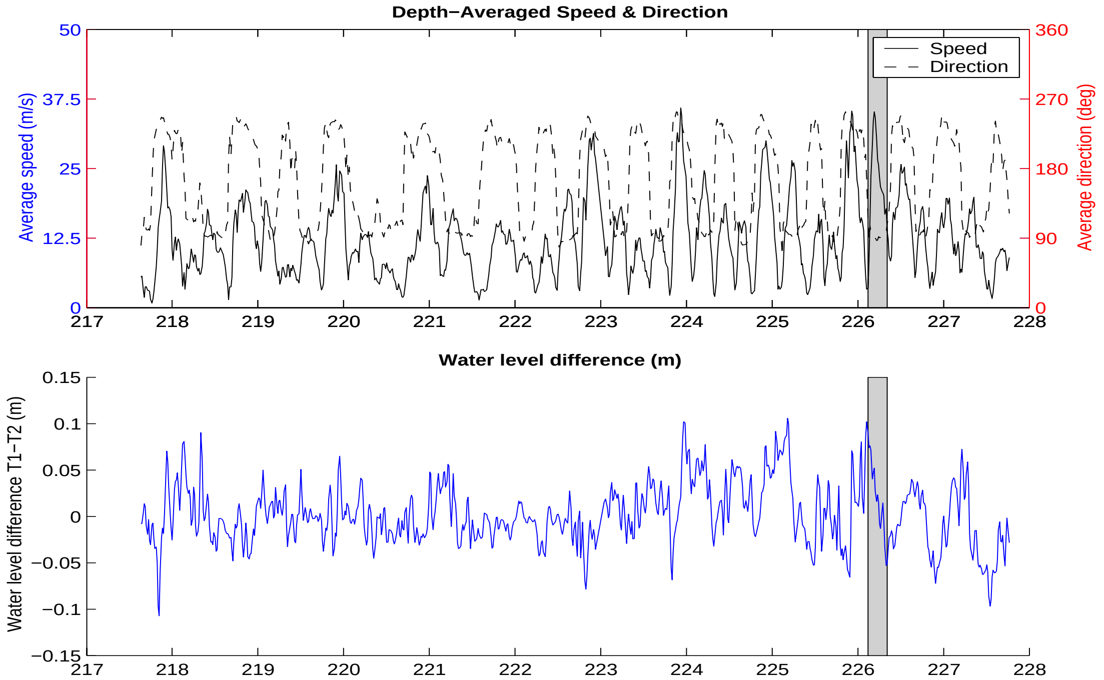

Figure 7. As for Figure 6, but for the entire record during the August 2005 experiment.

In order to estimate the flushing rates of the Gulf, three-dimensional simulations were conducted using the model ELCOM (see companion paper by Romero et al 2005.). As shown in Figure 6, the transport into and out of the gulf was dominated by a tidal-like signal driven by water level gradients. The absolute instantaneous transport into and out of the gulf ranged from -10000 (out of the gulf) to 10000 m^3 s^{-1} (into the gulf), equivalent to an average maximum velocity of approximately -10 to 10 cm s^{-1}, which corresponds to approximately 0.15 km^3 of water passing through the cross-section for each cycle. As the volume of Winam Gulf is 6.3 km^3, 2-5% of the volume of Winam Gulf passed across the cross-section on a daily basis over the course of the simulation during westerly currents and again upon the return flow during easterly currents. A more quantitative understanding of the flushing time of the Gulf was developed using tracer simulations whereby the concentration in the gulf was initialized with a known mass. Simulations were then run, and the flushing time was determined from computing the rate at which the tracer was removed from the Gulf. During the April 2005 experiment, the flushing time of the Gulf was computed to be approximately 300 days.

Summary

Two extensive field experiments were conducted in the Rusinga Channel and Winam Gulf of Lake Victoria in order to understand the exchange dynamics of the Gulf with the main lake. During both experiments, there were periods when the gulf was relatively cooler than the main lake, and periods when it was of similar temperature. The main finding of the experiments was that the flushing was driven by differences in water level between the main lake and the gulf. These water level changes resulted in periodic flow direction changes of between 6 and 12 hours, and resulted in currents in the channel of up to 35 cm/s. Despite these large flows, the amount of exchange between the gulf and the main lake was relatively small as the flows were oscillatory.

Further work is to be completed on understanding more about the processes resulting in water level changes in the gulf, and the currents driven by the water level changes compare to those driven by wind forcing and inflow events. The impact of these processes on water quality is also being investigated using a coupled hydrodynamic and water quality model.

Acknowledgements

This work was completed under the consultancy "Pilot Study of the Hydraulic Conditions over Rusinga Channel and Winam Gulf of Lake Victoria" by the Centre for Water Research for the Kenya Agricultural Research Institute and the Lake Victoria Environmental Management Project. The authors would like to thank numerous people involved in the successful completion of the study, including Peter Njuru-Gikuma, John Okungo, David Njoroge, Felix Sangele, and Rueben Ngessa of the Lake Victoria Environmental Management Program (LVEMP); the study's community participation officers Jean Boroto, Carol Adhisa, John Onyango, and Mary Kuria; the Kenya Marine and Fisheries Research Institute (KEMFRI) for the use of the Utafiti Research Vessel; and the crew of the Utafiti RV.

References

Gikuma-Njuru P. & Hecky R.E., 2005. Nutrient concentrations in Nyanza Gulf, Lake Victoria, Kenya: light limits algal demand and abundance, *Hydrobiologia*, 534 (1-3): 131-140.

Opande G.O., Onyango J.C., & Wagai S.O., 2004. Lake Victoria: The water hyacinth (Eichhornia crassipes [MART.] SOLMS), its socio-economic effects, control measures and resurgence in the Winam gulf, *Limnologica,* 34 (1-2): 105-109.

Romero J. R., Imberger J., Antenucci J. P., Ewing T., Khisa P. & Njuguna H., 2005. *Management Implications of the Physical Limnological Studies of Rusinga Channel and Winam Gulf in Lake Victoria.* Proceedings on the 11th World Lakes Conference, 31 Oct.-4 Nov. 2005, Nairobi, Kenya.

Williams A.E., Duthie H.C., & Hecky R.E., 2005. Water hyacinth in Lake Victoria: Why did it vanish so quickly and will it return? *Aquatic Botany*, 81 (4): 300-314.

Groundwater links between Kenyan Rift Valley lakes

Robert Becht[1] , Fred Mwango[2] , Fred Amstrong Muno[3]

[1] ITC PO Box 6, 7500 AA Enschede, the Netherlands
[2]Ministry of Water and Irrigation, PO Box 49720, Nairobi, Kenya
[3]Ministry of Water, Lands and Environment, PO Box 7096, Kampala, Uganda

Abstract

The series of lakes in the bottom of the Kenyan Rift valley are fed by rivers and springs.

Based on the water balance, the relative positions determining the regional groundwater flow systems and the analysis of natural isotopes it can be shown that groundwater flows from lake Naivasha to lake Magadi, Elementeita, Nakuru and Bogoria.

Introduction

The African Rift system is an extensive graben system extending from the Red Sea to Southern Africa. The drainage can be described as a chain of closed drainage basins with inflow from the flanks of the rift. The Kenyan Rift valley lake basins from North to South are: Turkana, Logipi, Baringo, Bogoria, Nakuru, Elementeita, Naivasha and Magadi.

Lake Logipi, L. Bogoria, L. Elementeita and L. Magadi receive a considerable part of the inflow from springs. Lake Turkana, Baringo, Nakuru are mainly fed by rivers. Only Lake Naivasha and Lake Baringo are freshwater lakes.

The lakes have been subject to various studies. Numerous studies address the ecology of the lakes (Harper, 2002). The size of the lakes correlate well the climatic conditions and therefore the coverage and sediment cores have been used to reconstruct the past climatic conditions over East Africa (Olago, 2000,Verschuren, 2002).

Especially the water balance of the economically important Lake Naivasha has been studied for more than half a century. The outflow of Lake Naivasha has been debated since the lake is studied. Due to its freshness the existence of outflow has hardly been doubted. However, the magnitude and direction of outflow is subject of discussion.

Previous studies of exploration of the Naivasha area began as early as 1880's by European explorers. Thompson of the Royal Geographical Society of England noted the freshness of the lake water and attributed it to being either of recent origin or the lake having an underground channel. Gregory (1892) suggested that the lakes freshness was due to undiscovered underground outlet. Nilsson (1938) proposed that the Lake's freshness was a result of water both entering and leaving the Lake via underground seepage.

In 1936, Sikes made the first statistical attempt to estimate monthly and annual water budget for Lake Naivasha and magnitude of the proposed underground seepage. It is uncertain which methods he used, but he estimated that water was seeping out of the lake at a rate of 43 million m^3 $year^{-1}$.

In his note on Lake Naivasha the hydraulic engineer acknowledged outflow and assumed it would be between 0 and 46 million m^3 $year^{-1}$ (Tetley, 1948). The higher estimate tallies remarkably well with more recent estimates.

The first attempt to address the direction of flow in the Naivasha area is made by Thompson *et al.*, (1958). They constructed a basic piezometric map with inferred flow directions.

Mcann (1992) was the first to attempt to look at the integral water balance of lakes Naivasha, Elementaita, Nakuru, Solai, Baringo and Bogoria.

He estimated a subsurface outflow to the Nakuru and Elementeita catchment to be about 37 million m3 year-1 using Darcys law. Based on the Naivasha lake water at least 34 million m^3 $year^{-1}$ infiltrates to Naivasha groundwater reservoir. He further estimated that about 14 million m3 year-1 and 23 million m^3 $year^{-1}$ of groundwater inflow is required to maintain constant lake water level for Lake Elemeinteita and Lake Nakuru respectively.

Gaudet and Melack (1981), extensively studied the chemical and water balnce of Lake Naivasha. Thy concluded that there is a subsurface water outflow from the Lake Naivasha but that this plays a minor role to explain the freshness of the lake. The water balance for 1973-1975 show equal groundwater in and outflow. The outflow constitutes 20% of the total outflow.

Åse *et al.,* (1986) worked on the surface hydrology of the lake and mass balance equation to derive possible subsurface outflow from the lake. He doubted however the possibility of an outflow due to the thick impeding clay layer covering the lake bottom

Clark *et al.,* (1990) and others used groundwater hydraulics and isotope methods to determine the water flow. They refer to a previous water balance study suggesting an outflow of 50 million m^3 $year^{-1}$. The flow to the south is via relatively shallow aquifers less than 500 m depth, and these may account for 50 to 90% of the total flow. Estimate of the northerly flow by the same authors was 11.3 million m^3 $year^{-1}$.

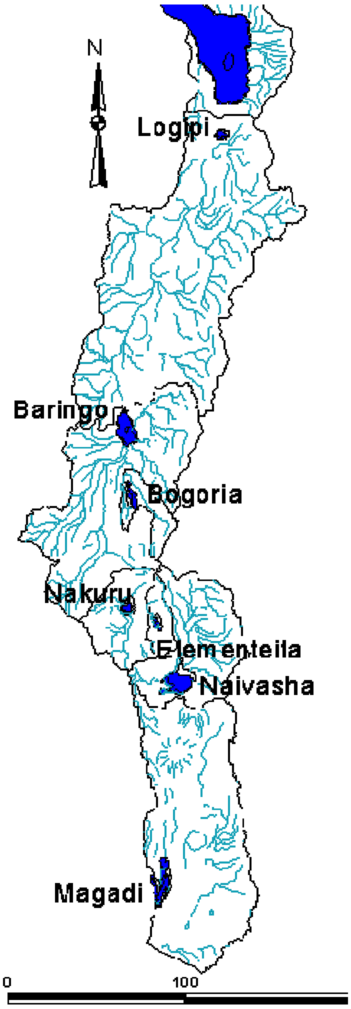

Figure 1. Map of the study area.

Darling *et al.,* (1990) were able to determine the direction of the outflow from lake Naivasha. They used stable isotopic composition of the fumaroles steam from volcanic centers in the areas to infer groundwater composition. Using simple modeling techniques they traced the outflow from the lake up to 30 km south. Lake water has also been detected in Olkaria steam. The work confirmed that of Allen (1989) that most of the water leaving the lake goes out between Olkaria and Longonot, whilst a smaller portion goes north between Eburru and Gilgil.

Suttard *et al.,* (1992) in a study commissioned by the European Union derived an outflow from Lake Naivsha of 45 million m^3 $year^{-1}$.

Ojiambo (1992) deduced from piezometric surfaces that the subsurface outflow from Lake Naivasha originates from the southern shores of the lake, and then flows southerly and southwesterly toward Olkaria. He estimates that the main lake outflow fluxes ranges from 18 and 50 million m^3 $year^{-1}$. Becht *et al.* (2002) calibrate a water balance model for Lake Naivasha simulating the monthly water levels from 1932 to 2000 and derive a groundwater outflow is 55 m^3 $year^{-1}$

Lake Baringo is likely to have groundwater outflow to explain its freshness. The flow from Lake Bariningo is exclusively directed towards the North feeding the perennial hot springs of Kapedo and Lurosio. (Darling, 1996) (WRAP, 1987)

Odhiamdo *et al*., (2005) have published the water balance of Lake Baringo.

In Ethiopia the ground water link between Rift valley lakes and the water balances has been described by several authors (Legesses (2004), Ayenew (1998), Darling (1996).

Physical setting

The Southern Rift Valley Lakes (SRVL) in Kenya are from North to South the Lakes Nakuru, Elementeitata, Naivasha and Magadi. Of these lakes Lake Naivasha is fresh. This is explained by its location at the culmination of the rift valley floor (1885 m asl) provoking the outflow of groundwater. This refreshes the lakes and prevents the accumulation of salts.

To the North Lakes Elementaita and Nakuru have elevations of 1776 and 1755 m asl, respectively. Lake Magadi has an elevation of 550m asl and acts as an absolute sink of water and solutes. This is manifested by the thick soda deposits covering most of the lakes surface.

The climate of the rift valley bottom is semi arid with average rainfall of 400-600 mm $year^{-1}$. The open water evaporation is in the order of 1700-2500 mm $year^{-1}$, leaving a water deficit of more than 1 m yr^{-1}. To replenish this deficit the lakes are fed by surface or groundwater.

Lake Naivasha and Lake Nakuru are mainly recharged by perennial rivers and ephemeral streams during the rainy season. The surface water inflow into Lake Magadi is limited and the main recharge comes from a series of springs. Lake Elementeita is fed by rivers and springs. The perennial rivers originate from the eastern and western flanks of the rift with rainfall up to 2000mm $year^{-1}$

Groundwater links: the regional picture

The regional picture of groundwater flow between the RVL can partly be deduced from lake properties. In a lake with only inflow all dissolve solids accumulate and the lake turns salt. The most extreme example of this is Lake Magadi that acts a final sink for large drainage area, and contains thick deposits of soda. In contrary, fresh water lakes are flushed by either surface water overflow or groundwater outflow.

The position of the lake with respect to topography and other lakes plays an dominant role in the regional hydrogeology. In general the lakes at the topographic high elevation are fresh through an outflow of groundwater (eg. Naivasha) and lakes in the low parts act as the final collector (eg. Magadi).

The relative position of the lakes will thus govern the regional flow patterns, where Naivasha constitutes the main recharge area and Magadi and Baring/Bogoria the main discharge areas.

The long-section of the rift valley topography shows the position of the lakes. In the Naivasha are elevation of the deep aquifer is known from geothermal wells. In this area the top of the geothermal aquifer is at approximately 1500 m asl. In the next valley South of lake Naivasha (Kedong valley) no water has been found in boreholes at an elevation of 1400 m asl. This observation constraints the outflow mechanism of lake Naivasha to the South to the deep regional (geothermal) aquifer. Such a deep flow system could also feed lake Bogoria and Lake Baringo. Flow further to the North is unlikely. The deepest point of the Northern part of the rift valley is at the Logipi swamp, just South of Lake Turkana.

If this area would be the terminal of groundwater flow towards the North, similar soda deposits as in the Magadi area would be deposited here, which is not the case. A rather shallow flow system from Lake Baringo feeding the Kapedo Springs to the North is likely.

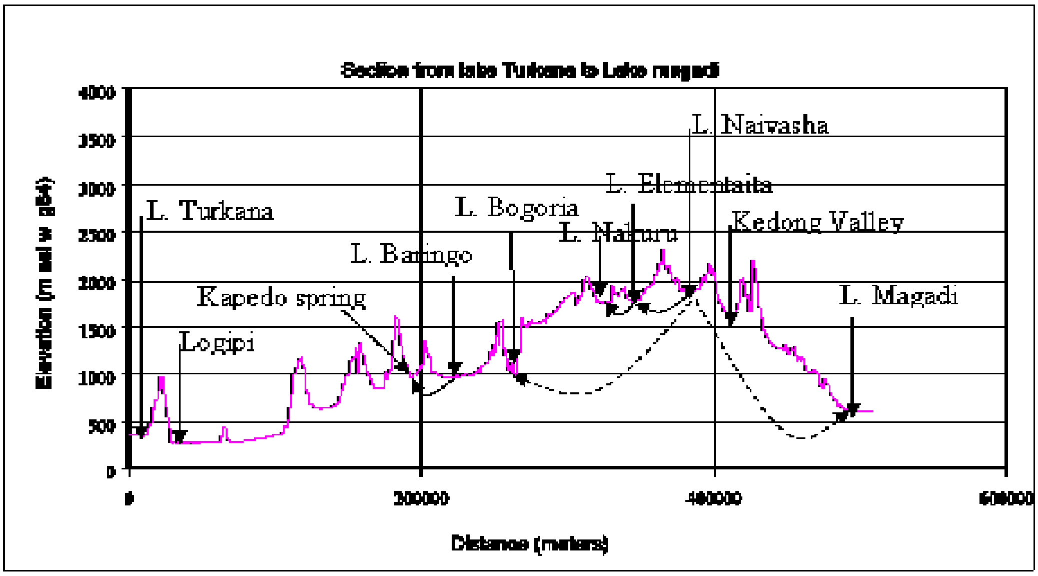

Figure 2. The water balances of the lakes show groundwater outflow from the fresh lake Naivasha and lake Baringo and groundwater inflow for all others.

Isotopes play an important role in deciphering the flow path between the lakes. The natural ^{18}O and ^{2}H stable isotopes occur in all natural waters. The isotopic composition is given as a deviation in ‰ from the international standard, the Standard Mean Ocean Water (SMOW).

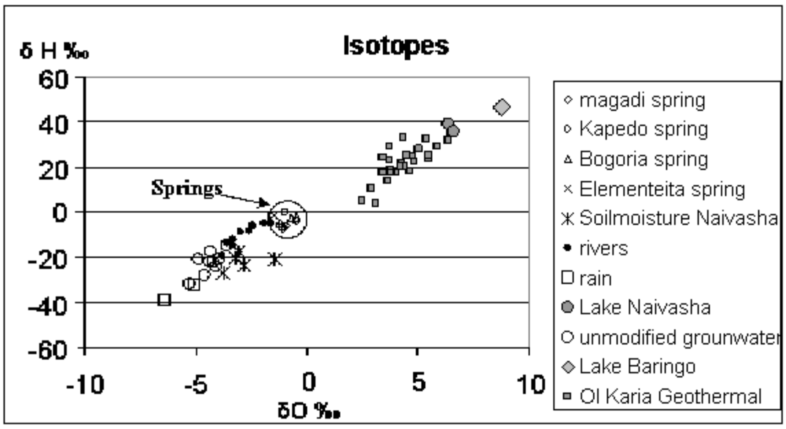

Figure 3.

The isotopic composition of natural waters from the rift valley is shown in figure 2. The natural rain is most depleted in the natural isotopes with a mean value of -5 ‰ ^{18}O and -30 ‰ ^{2}H form the lower limit whereas the lake water form the upper limit.. The isotopic composition of the enriched lake water (Naivasha 6.6 ‰ ^{18}O and 35 ‰ ^{2}H) provides an effective tracer for axial groundwater movement.

The isotopic composition of the geothermal water from the Naivasha area clearly shows a large distribution of lake water. The unmodified groundwater forms a cluster with a mean isotopic composition of (-4.6 ‰ ^{18}O and -28 ‰ ^{2}H. The springs presented in the graph (Magadi, Elementeita, Bogoria, Baringo, and Kapedo) all have a rather similar isotopic signature (-1.5/0 ‰ ^{18}O and -5/0 ‰ ^{2}H). The cluster is indicated by an enclosing circle in figure 2. This composition can be explained as a mixture of 70 % unmodified groundwater and 30 % lake water. It has to be stressed that other factors such as variation of the isotopic composition of the local rainfall and the evaporation from the hot underground reservoirs (fumaroles) may equally affect the isotopic composition of the deep refit valley groundwater.

The water balances of Southern Rift Valley Lakes (SRVL)

Assuming that the lakes are hydraulically linked by groundwater flow, this flow is reflected in the individual water balances of the SRVL.

The outflow from Naivasha (elevation: 1886 m asl) cannot disappear. Possible exit mechanisms are: Lake Magadi towards the South, the Lakes Elementaita, Nakuru , Bogoria and springs towards the North and in the form of steam in the geothermal areas. Mcann (1972) estimates the steam discharge in the Naivasha area in the order of 5 million m^3 $year^{-1}$. This is only 10 % of the total outflow from the lake.

The water levels of the SRVL can be simulated using a lake water balance model.

The model is based upon the monthly change in a simplified water balance. Components used are inflow from rivers, rainfall on the lake surface, evaporation from the lake surface, a constant groundwater flow and a dynamic groundwater component to take into account the interactions with the aquifer surrounding the lake. The lake Level–Area–Volume relationship is built into the model and allows the calculation of the rain and evaporation as a volume and the conversion from volume to level. The model uses a monthly time step, and is expressed as:

Lake volume change = inflow + (rainfall − evaporation)*lake_area + regional groundwater flow + Q_{aq} (m^3 $month^{-1}$)(1)

where Q_{aq} is the inflow to or outflow from a hypothetical dynamic groundwater aquifer linked to the lake. It is derived as:

$Q_{aq} = C\,(H_{lake} - H_{aquifer})\ (m^3\ month^{-1})$ (2)

where C is the hydraulic conductance between the lake and aquifer (m^2 $month^{-1}$) and H is the water level (m). The water level in the aquifer is updated using the in/outflow calculated for the previous month:

$H_{aquifer} = Q_{aq} / A \times Sy\ (m^3\ month^{-1})$

and

$H_{aquifer\text{-}new} = H_{aq.old} + H_{aquifer}$ (m)

where A is the surface area and Sy is the specific yield (porosity) of the hypothetical aquifer.

The lake volume is updated for every time step with the lake volume change. The new volume is than converted into a lake level.

On the long term the balance of Q_{aq} is zero; if the lake rises lake water will recharge the aquifer and water is released from the aquifer to the lake during recession.

The regional groundwater flow is the discharges or recharges the lake and is set to a constant for each model run. It is the main calibrating parameter determining the overall groundwater loss or gain. It also lumps and to a certain extent and balances out all missing parts and errors in the water balance.

The above model has been used to evaluate the water balances for the Lake Naivasha, lake Elementeita and lake Nakuru..

Lake Naivasha

The inflow data for Lake Naivasha are available since 1932 and therefore the model is calibrated for the period 1932 to 2001. The data set is of good quality as demonstrated by the good match between modeled and observed lake levels. The deviation between the modeled and observed series starting in 1980 is attributed to the abstractions from the basin mainly for irrigation.

Lake Elementeita

Lake Elementeita (elevation: 1776 m asl) is the first lake to the north of Naivasha and more than 200 m below lake Naivasha is the most logical place to intercept the northerly outflow of Lake Naivasha. The lake is situated in shallow pan floored by rather coarser salt impregnated sedimentary material. The lake receives inflow from the Mereroni, Kariandusi streams and groundwater sources. The Maji Moto spring on the south-eastern end of the lake is a major contributor to the lake. This spring is associated with a large N-S oriented fault system, most likely acting as a conduit between Naivasha and Elementeita. A model of the water balance is calibrated for the period 1958-1998.

This inflow corresponds to a mean flow of 0.5 m^3 sec^{-1}, much more than the measurable flow of the springs along the southern shores of the lake. In September 2005 the flow of these springs was 0.2 m^3 sec^{-1}, and locals confirmed a very constant flow. A large portion of the groundwater inflow to the lake seems to occur under the water surface as diffuse inflow. The isotopic composition of the springs shows a Lake Naivasha water contribution of 30 percent (Darling, 1996).

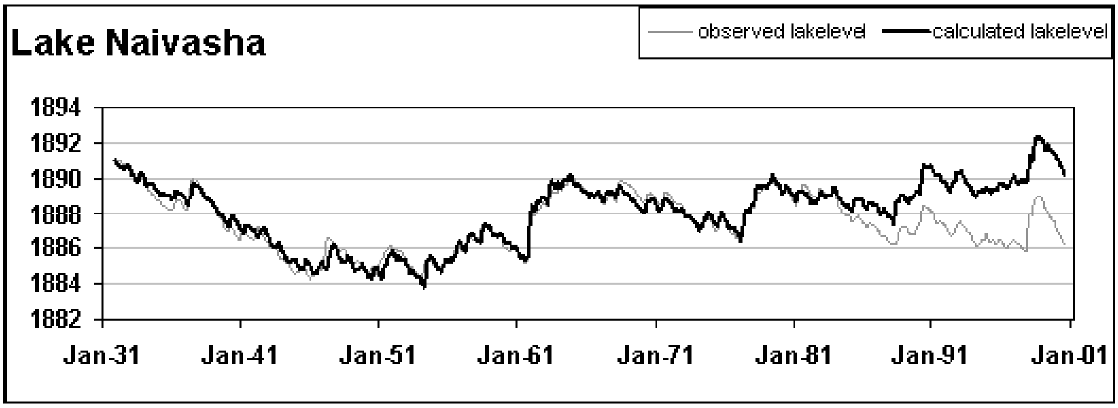

Figure 4 : The groundwater outflow, the main calibrating parameter of the model, is 54 million m^3 $year^{-1}$.

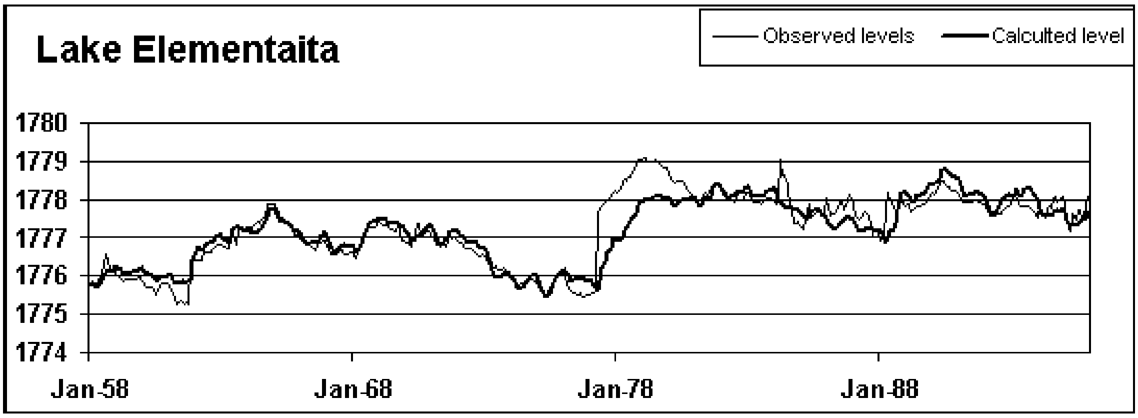

Figure 5: The groundwater inflow to achieve optimal calibration is 16 million m^3 $year^{-1}$.

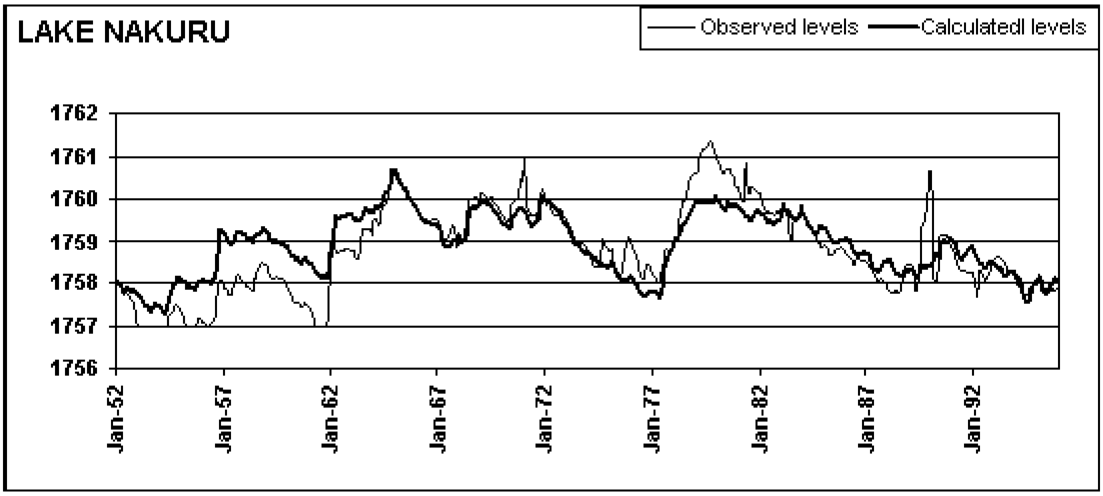

Figure 6. The calculated groundwater inflow based on the WBM is 24 million m^3 $year^{-1}$. This translates in an upward flow in the order of 0.5 m $year^{-1}$.

Combining the water balance and isotopic indicators one may conclude that 4.8 million m^3 $year^{-1}$ flow from Naivasha to Lake Elementeita.

The levels of Lake Nakuru (elevation: 1758 m asl) are modelled for the period 1952 -1996. During the period 1932-1952 only a few lake level observations exists but oral records indicate the lake only contained water after rains. In the early 1950's plans existed to construct a across the lake retaining the water of Njoro river and the springs. In the framework of this project hydrological surveys where carried out. Under the lake a shallow artesian aquifer exists indicating some groundwater recharge. The fact that the lake goes regularly dry indicates that the amount of upward flow to small to prevent the lake from falling dry.

Lake Nakuru

The modelled levels using the water balance model is shown in figure 3.

The origin of the groundwater flow is yet unknown. Along the North Western shore of lake Elementeita the groundwater gradient is sloping away from the lake indicating some flow from Elementeita towards Nakuru. Also Lake Naivasha may be directly linked to Nakuru. Most likely however the bulk of the water originates from the flanks of the Rift Valley. Isotope samples taken in 2004 from the area South of lake Nakuru did not indicate mixture with lake water, excluding a shallow connection between the two lakes.

Lake Magadi

Very few data on the water balance is available for Lake Magadi. A large part of the lake is covered with a thick crust of soda, reducing evaporation. Based on the open lake and the soda covered lake a approximate water balance has been established.

The evaporation rate from an open body of water is partly a function of the degree of salinity. It is well known that the evaporation rate from a saline water surface is lower than the freshwater surface (Salhotra *et al.* 1985) due to vapour pressure reduction.

Several researchers (Allison *et al.,* 1985; Ullman 1985; Jacobson and Jankowski 1989; Malek et al. 1990 Chen and Zawislanski *et al.,* 1992) carried out measurements of evaporation rates on salt lakes in various locations in United States and Australia. They reported that the actual evaporation rates from salt-crusted surfaces are as low as a few percent to 25% of the potential or pan evaporation rate in the same area. The three mechanisms which are considered responsible are: the high shortwave reflectivity of the dry salt crust, vapor density depression due to salinity and high salt crust resistance to moisture transfer.

During the dry season the water level of Lake Magadi does not change much, so one can assume that the evaporative loss is balanced by groundwater inflow. Figure 7 shows the monthly rain and evaporation data. Average yearly rainfall is 472 mm and yearly evaporation is 2433 mm, giving a water deficit of almost 2000 mm year^{-1}, assuming that the fluctuations of the lake are driven by local run-off.

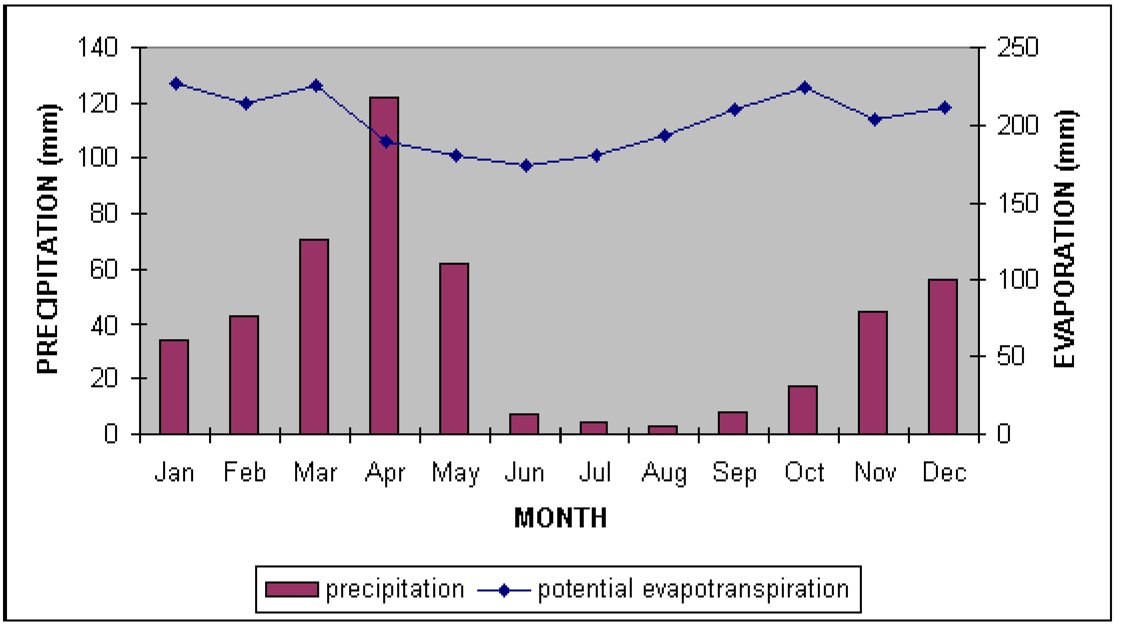

Figure 7.

The crusted surface area covers approximately 144 km^2 of the total surface area. The remaining surface was approximated to be of brine nature and covered the remaining 20 km^2 of the lake surface. The water deficit over the period June to September is 0.7 m, or 0.175 m month^{-1}. The open water deficit during the dry season is 3.5 million m^3 month^{-1}, or 42 million m^3 year^{-1}. The salt crust is wet until a few centimetres below the surface. Therefore we assume that the evaporation through the crust is 10% of the open water evaporation. This translates in an evaporative loss of 30 million m^3 year^{-1} through the trona crust. The total inflow is estimated at 72 million m^3 year^{-1}.

The Ewaso Ngiro river looses water through infiltration that is assumed to discharge in Lake Magadi. The springs along the Western side of the lake have the isotopic signature of rainwater. Furthermore several streams originating from the flanks of the rift valley infiltrate. Therefore it is unlikely that the discharge from Lake Naivasha constitutes a large portion of the inflow.

Lake Bogoria

Onyando et al (2005) estimates the groundwater inflow in Lake Bogoria at 28 million m^3 $year^{-1}$

Conclusions

Due to the reliable long-term data the outflow from Lake Naivasha can be reliably established at 55 million m^3 $year^{-1}$. This flow can only leave the groundwater system through springs and seepage zones associated with lakes and in the form of steam from the geothermal area. Based on water the water balance Lake Elementeita needs a groundwater inflow of 16 m^3 $year^{-1}$, Nakuru requires an inflow of 24 m^3 $year^{-1}$, Bogoria has an estimated groundwater contribution of 28 m^3 $year^{-1}$, and the Magadi groundwater component is in the order of 71 m^3 $year^{-1}$.

The combined groundwater flow towards these lakes is 139 m^3 $year^{-1}$. The outflow of Naivasha represents 40 % of this flow.

The isotopic composition of the spring water is similar and indicates a contribution of 30 % lake water. This is rather close to the factor calculated based on the water balance where the Naivasha outflow represents 40 % of the total inflow to the receiving lakes.

Water balance, hydrogeologic and isotope considerations all favour a considerable distribution of Naivasha water to the recharge of Lake Elementeita. The origin of the Lake Nakuru groundwater is most likely derived from local recharge with possibly a small contribution from Lake Naivasha water. The bulk of the infiltrated lake water is emerging in Lake Bogoria and Lake Magadi. There are no reasons to maintain the in previous literature postulated distribution of 80% Southerly flow and 20 % Northerly flow. More likely the flow to the North is equal or larger than the flow toward the South. Based on geometric considerations he Southerly flow is exclusively through the deep regional aquifer system. The Northerly flow to Bogoria is also through the deep aquifer whereas the flow towards Elementeita and Nakuru is in a shallow system.

Acknowledgements

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References

Allison,G.B., and Barnes, C,J., 1985. Estimation of evaportion from the normally dry Lake Frome in South Australia. *Journ. Hydrol Amsterdam*, 78, 229-242.

Ayenew T., 1998. *The hydrogeological system of the Lake District basin, Ethiopia,* University of Amsterdam, Enschede, The Netherlands.

Ayenew T. 2001a. Numerical groundwater flow modeling of the Central Main Ethiopian Rift lake basin. SINET: *Ethiopian Journal of Science* 24(2): 167–184.

Ayenew T., 2001b. Surface kinetic temperature mapping using satellite spectral data in Central Main Ethiopian Rift and adjacent highlands. SINET: *Ethiopian Journal of Science* 24(1): 51–68.

Åse, L-E., Sernbo, K. & Syren, P., 1986. *Studies of Lake Naivasha, Kenya and its Drainage Area.* Forkningsrapporter 63, Stockholms Universitet Naturgeografiska Institutionen, Stockholm: 75 pp.

Ashton, P. J., 2002. *Avoiding conflicts over Africa's water resources.* Ambio 31: 231–235.

Brind, W. & J. Robertson, 1958. The Hydrology of Lake Naivasha. Hydrology Section, Ministry of Works, Nairobi: 9 pp.

Butzer, K. W., G. L. Isaac, J. L. Richardson, & C. Washbourne- Kamau, 1972. Radiocarbon dating of East African lake levels. *Science* 175, 1069–1076.

Clarke, M. C. G., D. G. Woodhall, D. Allen & G. Darling, 1990. Geological, Volcanic and Hydrological Controls on the Occurrence of Geothermal Activity in the area surrounding Lake Naivasha, Kenya. Minstry of Energy, Nairobi: 138 pp.

Eugster, H.P., Hardie, L.A., 1978. *Saline Lakes*

Legesse,D.,Vallet-Coulomb, C., Gasse, F., 2004. *Analysis of the hydrological response of a tropical terminal lake, Lake Abiyata (Main Ethiopian Rift Valley) to changes in climate and human activities*. Hydrol. Process. 18, 487–504

Fujiyasu Y., Fahey M., 2000. Experimental Study of Evaporation from Saline Tailings; Journal of Geotechnical and Geoenviron. Engineering.

Gaudet, J. & J. M. Melack, 1981. *Major ion chemistry in a tropical African lake basin*. Freshwat. Biol. 11, 309–333.

Goldson, J. 1993. A Three Phase Environmental Impact Study of Recent Developments around Lake Naivasha. *Lake Naivasha Riparian Owners' Association*, Naivasha, December 1993: 109 pp.

Leakey, L. S. B., 1932. East African lakes. *Geog. J.* 77, 497–512.

Malek.E, Bingham,G.E and McCurdy,G.D 1990, Evapotranspiration from the margin and moist playa of a closed desert valley,J.Hydrol,120: 15-34

Nilsson, E., 1932. Quarternary Glaciations and Pluvial lakes in British East Africa. PhD Thesis, Centraltryckeriet Stockholm: 101 pp.

Oestergaard, P., 1974. Abstraction of water from Lake Naivasha. Files WRH 206 vol IV/131, 5/4/74: 2 pp and WRH 206 vol III/132, 28/5/74: 5pp, Ministry of Water Development, Nairobi.

Onyando, J. O., Musila, F. and Awer, M., 2005. The use of gis and remote sensing techniques to analyse water balance of lake bogoria under limited data conditions. Journal of Civil Engineering Research and Practice, Vol. 2 No. 1, April 2005, pp. 53 – 65

Olago, D. O. and E. O. Odada ,2000. An inter-basinal comparison of the sedimentology of Late Holocene to recent sediments in the Rift Valley, Lake Turkana, Kenya. Journal of African Earth Sciences 31(2): 237-252.

Mesny, M.,1977. *Irrigation potential in the Baringo District*. Water Department, Nairobi.

Richardson, J. L., 1966. Changes in the level of Lake Naivasha, Kenya, during postglacial times. Nature, 5620, 290–291.

Richardson, J. L. 1972. Palaeolimnological records from Rift lakes in central Kenya. *In* E. M. van Zinderen Bakker, ed. *Palaeolimnology of Africa, the Surrounding Islands and Antarctica*. Balkema, Cape Town: 131–136.

Richardson, J. L. & A. E. Richardson, 1972. *History of an African Rift lake and its climatic implications*. Ecol. Mongogr. 42, 499–534.

Sikes, H. L., 1936. Notes on the hydrology of Lake Naivasha. *Journal of the East African and Uganda Natural History Society*, 13, 73–89.

Stuttard, M., J. B. Hayball, G. Narciso, M. Suppo, L. Isavwa & A. Oroda 1999. Modelling lake level changes: examples from the Eastern Rift Valley, Kenya. *In* Harper, D. M. & A. Brown eds. *The Sustainable Management of Tropical Catchments*. J. Wiley & Sons, Chichester: 337–358.

Tetley, A. E., 1948. *Notes on Lake Naivasha by the Hydraulic Engineer*. Directorate of Water Development, Nairobi.

Vincent, C. E., T. D. Davies & U. C. Beresford, 1979. Recent changes in the level of Lake Naivasha, Kenya, as an indicator of equatorial westerlies over East Africa. *Climatic Change* 2, 175–191.

Determination of phosphates and total phosphorus in water and wastewater by chromatographic, colorimetric and spectrometric techniques

Boikhutso T.P.[1], R McCrindle[2], N Adam[3]

[1]Department of Analytical Services, Tshwane University of Technology, Pretoria 0001, South Africa.
[2]Department of Chemistry, Tshwane University of Technology, Pretoria 0001, South Africa.
[3]Department of Process Technology, Rand Water, Vereeniging 1930, South Africa.

Abstract

The determination of the concentration of inorganic phosphorus and total phosphorus in water is used as a measure for algal growth, which in turn is an indication of water pollution. A need for accurate, simple and reliable methods for determining phosphorus is particularly relevant to current eutrophication and water quality problems. This paper is concerned with the analysis of phosphorus in water (particularly orthophosphate, organic phosphorus and total phosphorus).

Ion Chromatography, Flow Injection Analyser, Inductively Coupled Plasma Optical Emission Spectrometry and Inductively Coupled Plasma Mass Spectrometry were used for analysis of phosphorus in water. Both composite influent and composite effluent samples from the Sebokeng Wastewater Works were analysed and monitored with a view to meet the phosphorus effluent standard of 1.0 mg/l as prescribed by the Department of Water Affairs and Forestry (DWAF). Orthophosphate, phosphorus, total phosphorus (converted to orthophosphate) were measured in water and wastewater samples.

The biological phosphate removal in Sebokeng wastewater works tends to be sensitive and subject to many fluctuations, making it difficult to achieve full compliance with discharge standards and this has been clearly demonstrated by the composite effluent results.

Results indicated that the methods for determining both orthophosphate and total phosphorus were most reliable when using colorimetric techniques. The composite effluent results clearly show that the concentration of orthophosphates is less, or equal to, that of total phosphate. It was concluded that phosphorus as determined by Inductively Coupled Plasma Optical Emission Spectrometry is equivalent to total phosphorus.

Key words: phosphorus, eutrophication, influent, effluent, water quality, colorimetric, chromatography, spectrometry.

Introduction

One of the effects of sewage effluent discharge in water cause-ways and dams is eutrophication according to Twort, Ratnyaka and Brandt (2000:201-300). The nutrients in sewage, mainly phosphorus, causes a detrimental aesthetic effect, increases chemical requirements in water treatment and excessive growth of algae and other aquatic plants which can cause destruction of habitat and depletion of dissolved oxygen. This process in turn causes death of aquatic life because of the lowering of oxygen level. It is for these reasons that much attention has been given to phosphorus removal in the sewage treatment process.

The increasing awareness of eutrophication effects has led to the introduction of legislation controlling the discharge of phosphorus to receiving waters. In South Africa a special phosphate standard was introduced restricting the concentration of phosphorus in wastewater discharges to 1.0 mg/l as orthophosphate according to Government Gazette, 1984 (quoted by De Haas, Wentzel and Edam, 2000). Wastewater treatment plants use orthophosphate and total phosphorus results to monitor or to meet effluents standard.

Phosphorus is classified as dissolved, (PO4) generally as orthophosphate, condensed phosphorus, particulate (organic) phosphorus and total phosphorus, which includes polyphosphate. There is no chemical method for determining condensed phosphorus. “Condensed” phosphorus and “organic” phosphorus are calculated as small differences between orthophosphate and total phosphate and are consequently of limited validity according to Harwood and Hattingh (1973).

In this paper four sensitive analytical methods are proposed for phosphorus analysis in wastewater, that is Ion Chromatography (IC), Flow Injection Analyser (FIA), Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES) and Inductively Coupled Plasma Mass Spectrometry (ICPMS). Further, the study contributed to quality of test results assurance and Sebokeng Wastewater Works process control, optimization and effluent monitoring.

According to Spivakov and Maryutina (1999)comparison of results using ICPOES has shown the IC method to be reliable and applicable to phosphorus speciation studies in water samples. IC enables the determination of orthophosphate, diphosphate and triphosphate along with chloride, nitrate and sulphate anions when indirect UV detection is being used.

Spivakov and Maryutina (1999) mentioned that FIA with photometric detection seems to be the most developed technique for determining orthophosphate and total phosphorus at mg/ml and mg/l successfully applied to water samples.

The ICPOES is preferred among other techniques for phosphorus analysis due to multi-element analysis capability with high sample through out, the possibility of element determinations in broad concentration range including detection limits and less expensive approach for analysis.

Phosphorus is mentioned by Wilbur (2001) to be a difficult element to be determined by ICPMS as being at mass 31 due to the presence of overlaps by isobaric and adjacent interfering species.

This study is concerned with the methods being used for orthophosphate, phosphorus and total phosphorus differentiation.

Materials and methods

The Sebokeng Wastewater Works influent and effluent samples were taken weekly from June 2002 to April 2004. All samples were collected in plastic containers washed with phosphate free detergent and stored at 4 °C prior to analysis.

Samples for ICP-OES and ICP-MS were filtered through 0.45 µm filter papers and prepared using 1% nitric acid to preserve phosphorus. The FIA and IC samples were only filtered with 0.45 µm filter papers. The total phosphate was determined by digesting samples with persulphate in a Kjeldahl system followed by analysis using the phospho-molybdic photometric method.

The analysis was performed in a SANAS accredited laboratory as per Rand Water procedures, SABS Edition (1999) and Standards Methods (1995). The calibration, quality control and blank standards were used in all the instruments to achieve accuracy and quality of data.

Ion Chromatography

A Metrohm 761 Compact chemically suppressed ion chromatography with a conductivity detector was used in determination of orthophosphate. Separation was performed on an ion exchange analytical column and a peak chromatography workstation was used for data collection. The following operational conditions were used:

- Eluent: 1.7 mM $NaHCO_3$ + 1.8 mM Na_2CO_3
- Injection Volume: 20µL

Flow Injection Analyser

Measurements were based on the formation of phospho-molybdic blue complex, which absorbs light at a wavelength of 880 nm. The ascorbic acid was used as a reducing agent, which is claimed by Fogg and Wilkinson (1958:406) to increase colour stability and to reduce silicate interferences from other ions, including silicate. The measurements were based on the following reactions.

1. $PO_4^{3-} + 12(NH_4)_2 MoO_4 + 24H^+ \rightarrow (NH_4)_3PO_4.12 MoO_3 + 21NH_4^+ + 12 H_2O$
2. $(NH_4)_3PO_4.12 MoO_3 + Sn^{2+} \rightarrow$ Molybdenum blue $+ Sn^{4+}$

Inductively Coupled Plasma (ICP-OES)

Measurements were carried out using Argon radiation collected at 177.495 nm, detected by a Spectro Circular Optical System Charged Coupled detector and turned into electronic signals that were converted into concentration in mg/l. Measurements were taken using the following operational conditions:

- ICP- OES power: 1.5kW
- Coolant Argon flow: 15 l/min
- Nebuliser Argon flow: 1.0 l/min
- Auxiliary Argon flow: 0.8 l/min

Inductively Coupled Plasma (ICP-MS)

A multi-element mode Micromass ICP platform was used to carry out the measurements. A concentric Mynhardt nebuliser was used. An analytical program was established for calibration and analysis. Ions were resolved according to their atomic mass and charge and detected using a dynode-photo multiplier system. The following operation conditions were used:

- ICP-MS power: 1.35 kW
- Coolant Argon flow: 13.5 l/min
- Nebuliser Argon flow: 0.75 l/min
- Sample uptake: 0.3 ml/min

Results and discussion

In order to draw differences between all five techniques used for phosphorus concentration, statistical tests were performed such as Student's t-test and Mann-Whitney test. The t-test compares the actual difference between two means in relation to the variation in the data. If the calculated t value exceeds the tabulated value than the means are significantly different. In order to take decision the calculated t-statistic must fall within the critical region with at least a 005 level of significance for a two-tailed test. The critical values can be found in Appendix F-9 of Jongman (1992) and are calculated using the degrees of freedom (n-1) and the critical alpha (α). If the calculated t-values exceed the tabulated value, then the means are significantly different.

The Mann- Whitney U test, which is referred as the rank sum test in Jongman *et al*. (1995:201-203), was used on results from FIA and ICP, FIA and TP as well as ICP and TP. This test uses ranks instead of actual values in order to determine sample mean differences, which is particularly useful when dealing with non normally distributed data and the effect of outliers is eliminated. Further, Mann-Whitney was important to complete for the determination of reliability and robustness of the results of the t-test. The calculated z value at 95% confidence limit should be between –1,96 and +1.96 for results to be similar.

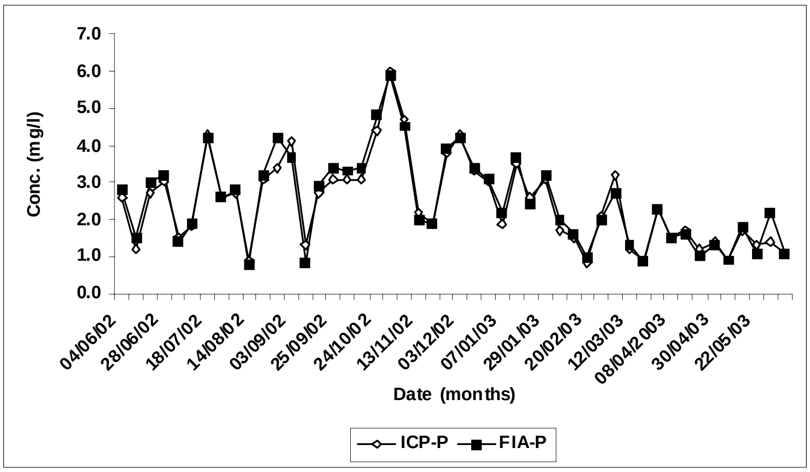

Figure 1 : Phosphorus Composite influent concentrations taken using ICPOES (ICP-P) and FIA (FIA-P).

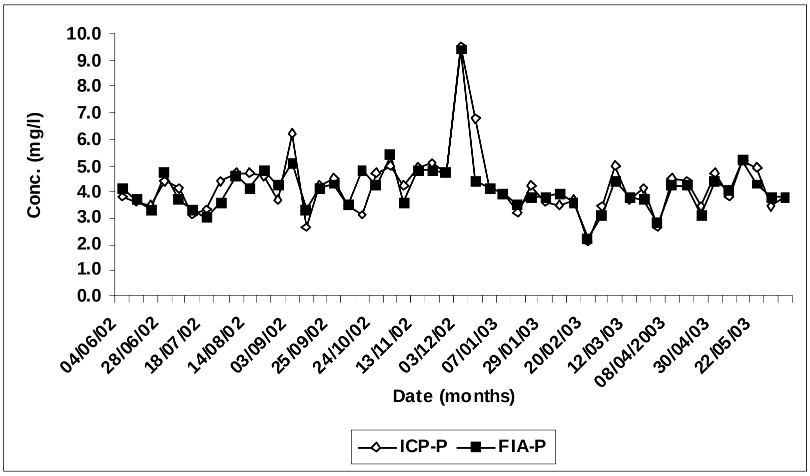

Figure 2: Phosphorus Composite effluent concentrations taken using ICPOES (ICP-P) and FIA.

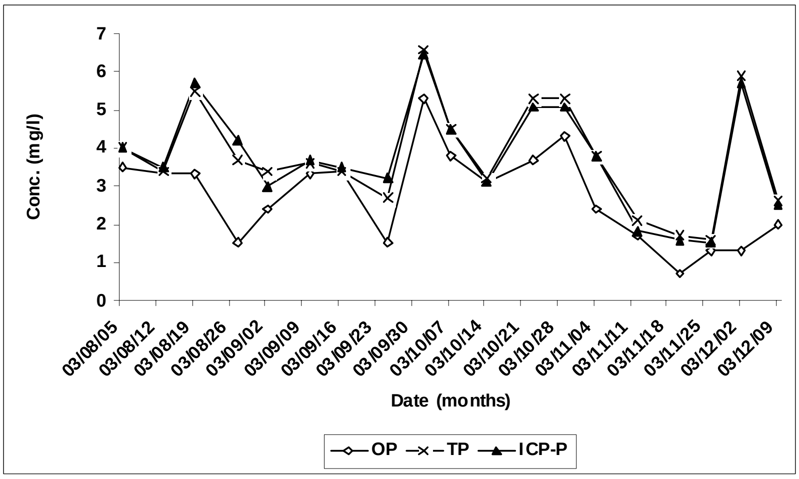

Figure 3: Phosphorus Composite influent concentrations taken using FIA (OP), FIA persulphate (TP) and ICPOES (ICP-P).

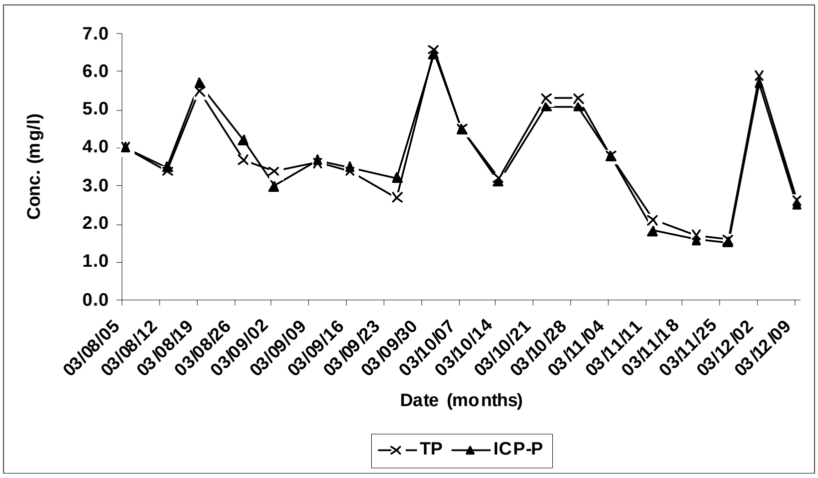

Figure 4: Phosphorus Composite effluent fluent concentrations taken, using FIA (OP), FIA peruslphate (TP) and ICPOES (ICP-P).

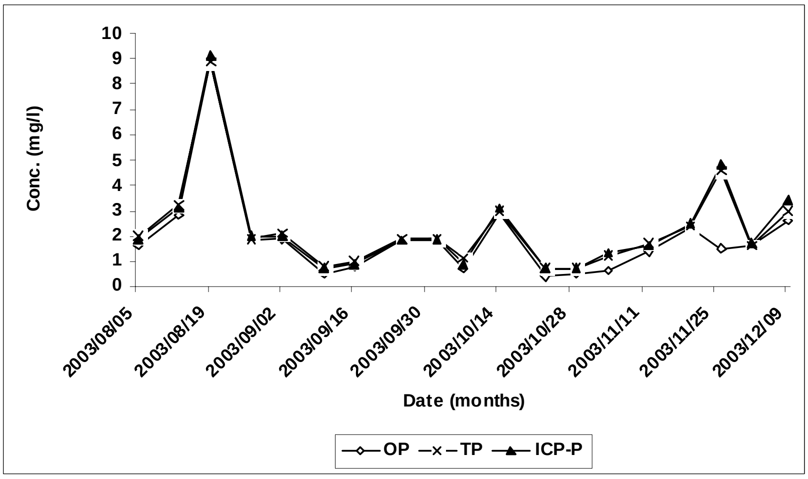

Figure 5: Phosphorus Composite influent concentrations taken over a year using FIA (OP), FIA persulphate (TP) and ICPOES (ICP-P).

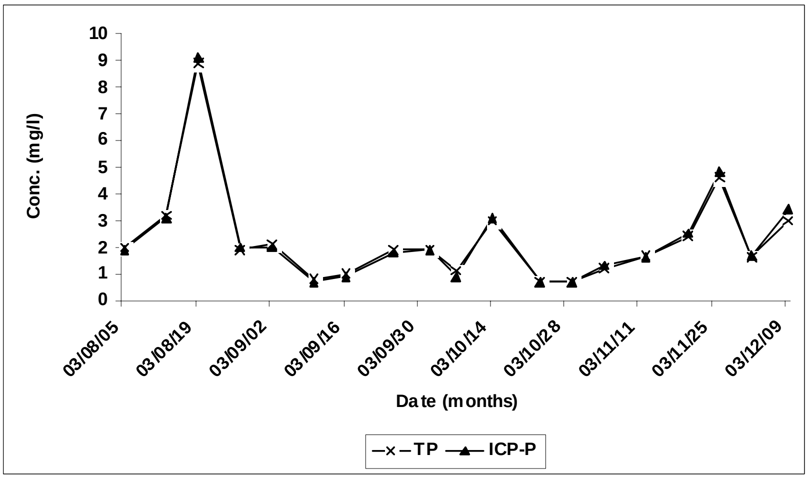

Figure 6: Phosphorus Composite effluent concentrations taken using FIA (OP), FIA persulphate (TP) and ICPOES (ICP-P).

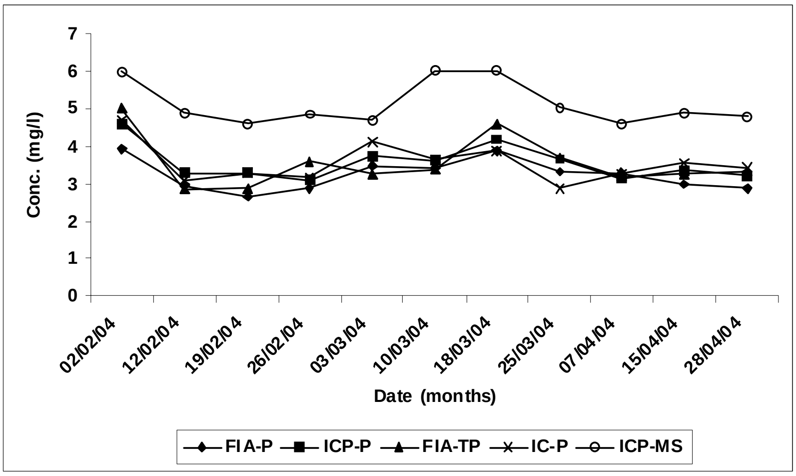

Figure 7: Phosphorus composite effluent concentrations taken using FIA (FIA-P), ICPOES (ICP-P)), FIA peresulphate (FIA-TP), ICPMS (ICPMS-P) measured against 1.0 mg/l Phosphorus Standard.

Conclusions

The techniques used demonstrated that for the composite effluent sample, FIA orthophosphate concentration was equivalent to total phosphate concentration. Therefore, for the composite effluent, the orthophosphate technique is adequate. The association of FIA, IC, ICP, TP has allowed a precise investigation of phosphorus concentration in effluent discharge from Sebokeng Wastewater Works.

The composite influent results obtained from FIA, IC, ICP, TP have evidenced that the orthophosphate concentration will either be equal or less than total phosphorus (refer to Figure 3 and Figure 4). For the composite influent the total phosphorus would be suitable but orthophosphate would be preferable for quality results assurance. Further, both the composite influent and composite effluent results show close approximation between the concentration of ICP phosphorus and total phosphorus concentration (refer to figure 5 and figure 6).

The investigation has proven that sample type is essential when selecting a technique for phosphorous analysis in water and wastewater samples. Further, significance cost savings can be achieved if this option is being practiced.

This report has shown that colorimetric technique is the simple, reliable technique for orthophosphates analysis and spectrometry ICP-OES is the quick and accurate method in determining total phosphate.

For both composite influent and composite effluent samples, there is a significant difference in ICPMS phosphorus concentration (refer to Figure 7 and 8). This requires further investigation, as the expectation is that the ICPMS phosphorus concentration is also equivalent to ICP phosphorus concentration.

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References

De Haas, D.W., Wentzel, M.A.C. and Edam, A.G. 2000. *The use of simultaneous chemical precipitation in modified activated sludge systems exhibiting biological excess phosphate removal*. South Africa: Water SA.

Eaton, A.D., Clesceri, L.S. and Greenbers, A. 1995. *Standards methods for the examination of water and wastewater. 19th ed*. USA: American Public Health Association/ American Water Works Association/ Water Environment Federation, Washington DC.

Fogg, D.N. and Wilkinson, N.T. 1958. *Analyst.* New York: Mcgraw-Hill

Harwood, J. E. & Hattingh, W. H. J. 1973. Colorimetric methods of analysis of phosphorus at low concentrations in Water. *In.* E.J. Griffith, A. Beeton, J.M. Spencer & D. T. Mitchell. *Environmental Phosphorus Handbook.* Canada: Wiley-Interscience.

Johnson, R.R. 1992. *Elementary Statistics. 6th ed*. United States: PWS-KENT and Wadsworth Inc.

J*ONGMAN*, R. H. G., Terbraak C. J. F. & Van Tongeren. 1995. *Data analysis in community and landscape ecology*. Great Britain: Cambridge University Press.

SABS Edition. 1999. *SABS ISO/IEC 17025*, SA.

Spivakov B.Y. & Maryutina, A.1999. *Phosphorus speciation in water and sediments*. Great Britain:IUPAC.

Twort A.C., Ratnayaka D.D. & Brandt M.J. 2000. *Water Supply*: Arnold.

Wilbur S. 2001. *Determination of trace levels of phosphorus in environmental samples with 7500c ICPMS system.* USA: Agilent Technologies.

Restructuring water quality monitoring of Lake Biwa

Yukiko Ikeda[1] and Kazuomi Okuda[2]

[1]Lake Biwa and Environmental Policy Office, Water Policy Administration Division, Department of Lake Biwa and the Environment, Shiga Prefecture. 4-1-1 Kyomachi Otsu-shi Shiga 520-8577 Japan
[2]Environmental Science Research Division, Lake Biwa Environmental Research Institute. 5-34 Yanagasaki Otsu-shiShiga 520-0022 Japan
[1]E-mail:ikeda-yukiko@pref.shiga.lg.jp

Abstract

Lake Biwa is the largest lake in Japan and the third oldest lake in the world. Lake Biwa is well familiar as "Mother Lake" among Japanese, because it is not only the source of a plenty of biodiversity, but also the important resource of fresh water to supply water for 14 million people and industries. On the other hand, the lake is very sensitive to stresses from human activities. Therefore, we need to adopt the management method for the conservation and the use of the lake, which differs from rivers.

Lake Biwa is the important constituent factor of the aquatic environment in the catchment's area, so it is impossible to manage sustainable the lake without considering whole the catchment's area. To manage the lake soundly, we need to consider various factors such as socio-economic aspects, land use and so on. Also, we need to improve policy making process, based on interdisciplinary investigations. To make and implement the policy effectively, it is essential that all the stakeholders such as the government, local governments, NPOs, citizens, irrigation groups, industries, education institutes, research centers, etc. take part.

Based on these points, to promote the sustainable management of the lake, we began to restructure the water quality monitoring system of Lake Biwa. Our new program of water quality monitoring consists of ①intensification of non-point source pollution's measure, ②construction of the original water quality model of Lake Biwa, ③adoption of TOC (Total Organic Carbon) as an indicator to grasp the material income and expenditure, ④adopting new indicators such as biological ones, ⑤introduction of community based monitoring.

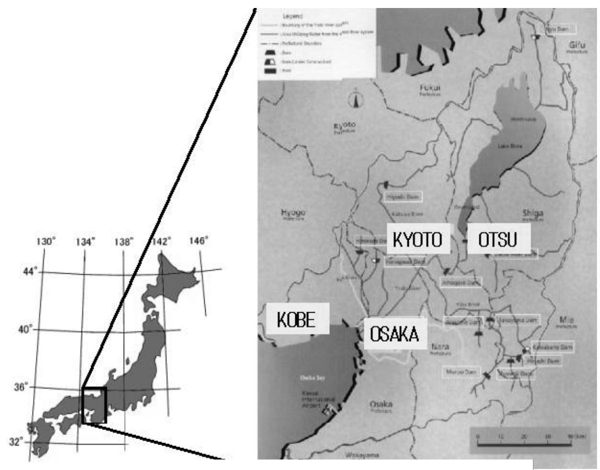

Fig.1 Map of the Lake Biwa-Yodo River Basin.

Key words: Lake Biwa, Sustainable management of the lake, Water quality monitoring system

Introduction

Shown as the nickname "Mother Lake", Lake Biwa is not only the source of a plenty of biodiversity, but also the important resource of fresh water to supply water for 14 million people and industries in Shiga prefecture and the Kyoto-Osaka-Kobe metropolitan area. On the other hand, the lake is very sensitive to stresses from human activities. Therefore, we need to adopt the management method for the conservation and the use of the lake, which differs from that of rivers.

Because Lake Biwa is the important constituent factor of the water environment in the whole catchment's area, it is impossible to manage sustainable the lake without considering the catchment's area. To manage the lake soundly, we need to consider various factors such as socio-economic aspects, the land use and so on. We, also, need to improve policy planning process, based on interdisciplinary investigations. In order to plan and implement the policy effectively, it is essential that all the stakeholders such as governments, local governments, NPO's, citizens, irrigation groups, industries, education institutes, research centers, etc. take part.

On the basis of these points, to promote the sustainable management of the water environment of the lake and the basin, the restructuring of the water quality monitoring system (the monitoring), grasping not only present situations, but basing on new concepts, is demanded. We established a committee consisted of the five specialists and researched new scheme for the purpose above mentioned. Based on the discussion in this committee, it became important to grasp the water environmental situation by the effective monitoring, to share the information among stakeholders and to reflect on environmental conservation policies. Consequently, "Vision for Water Quality Monitoring of Lake Biwa and the Basin" was reported in the Shiga Prefectural Environmental Council.

Current status and challenges for the water environmental conservation in Lake Biwa and the basin

Transition of Lake Biwa

The changing of the water environment in Lake Biwa began to appear as outbreaks of plankton, excess of invading weed species and death of fishes by agricultural chemicals just before Japan's rapid economic growth during the 1960's.

A large scale of land reclamations by drainage began in attached lakes since around 1945. It is one of big environmental changes with Lake Biwa and the area of attached lakes has decreased to 15% compared with that of prewar era. Japan's rapid economic growth was accompanied by industrization, population increase, modernization of lifestyles and various developments. As a result, the environmental impact to the rivers and Lake Biwa had rapidly increased.

The Lake Biwa Comprehensive Development Plan from 1972 to 1997, the constructions of lakeshore's banks, the agricultural infrastructure improvements, sewage systems and the decline of forestry had big impacts to Lake Biwa.

Under the influence of the expansion and diversity of socio-economic activities and of global environmental changes, the water environment has undergone new changes like rising up of chemical oxygen demand (COD) since 1984, outbreaks of picoplankton in 1989, that of weed species; *Ocyratoria* in 1998, an appearance of a lack of oxygen at the bottom of the lake and so on.

Problems around Lake Biwa

Concerning the legal system of Lake Biwa, there are the laws such as the Rivers Law, the Water Pollution Control Law, the Natural Park Law, the Law Concerning Special Measures for the Prevention of Lake Water Quality, etc. and the ordinances of Shiga prefecture such as the Ordinance for Wastewater Quality Standards, the Ordinance on Conservation of Reed Colonies along Lake Biwa, the Ordinance Relating to the Appropriate Leisure Usage of Lake Biwa, etc. Each law has its own objective and superintendent, but no comprehensive management system has been established to comprehend them.

Other problems are like the followings: socio-economic factors to prevent the sustainable possibility, changing of the relationship between Lake Biwa and the industries and the life, influence on the water quality and the ecosystems by the air pollution, influence on the sound water cycle, influence on the water quality, the ecosystems and the fishery by the changing of the water level and the velocity of a flowing fluid, lack of the recognition and the understanding of the influence on Lake Biwa by the human activities, decrease of the biodiversity and habitats, influence on the ecosystem and the fishery by alien species, loss of natural beauties, sceneries and primeval beauties, influence by the climate changing and so on.

Challenges toward the sound management of Lake Biwa

Challenges toward the sound management of Lake Biwa are the followings: ①implementing integrated water resources management (IWRM), promotion of industrial activities with lower environmental impact, measures against chemicals, reinforcement of measures against non-point pollution, evaluating importance of the shore and the littoral zones, promotion of comprehensive researches, implementation of the effective monitoring and assessments, sharing and application of research results, promoting environmental educations and training specialists for the water environmental conservation, etc.

Water must be in a sound condition in every aspect, the quality, the quantity, the place and the time. Having the many-sided values, Lake Biwa is the water resources for the agriculture, the fisheries, the industry, the sightseeing, the maintenance of the ecosystem, etc. as well as the resource of fresh water. Therefore, the challenges for the management is not only about the water quality, but also about the wide spread things like the water level, the water quantity, the living things, the ecosystems, the land use, the industrial structure, the amenity, the consumption styles and lifestyles of residents. The sound management of the lake must be based on the idea of IWRM. With the respect for historical details and customs, the management needs to be integrated promoting based on scientific knowledge and the stakeholders' participation. Therefore, the persons concerned need to have the common idea and vision.

We must select indicators for appraisal standards of its soundness without the restraint of current ones. It is completely necessary to implement the corresponding monitoring intentionally and continuously. Moreover, for the effective monitoring, we need to analyze the past results precisely and to construct the water quality model of Lake Biwa. The scientifically planed monitoring is important by these procedures.

Effective water quality monitoring for the Water Environmental Conservation in Lake Biwa and the basin

Present situation of monitoring

The following classifications are made to summarize the present monitoring.

- Targeted places: Lake Biwa and the flowing rivers
- Targeted researches in Lake Biwa: the surface water quality monitoring and the bathymetric water quality monitoring
- Targeted measurement's methods: the regular monitoring and the automatic monitoring

The present monitoring system of Lake Biwa consists of the researches for plankton, freshwater red tides, water bloom and so on as well as the above researches.

As the surface water quality monitoring of Lake Biwa, the regular monitoring is carried out at 47 points once every month (Shiga Prefectural Government; 11 points, the Ministry of Land, Infrastructure and Transport and the Japan Water Agency; 36 points). Environmental Water Quality Standards for protecting the living environment, ones of protecting human health and monitoring substances determined in the Water Pollution Control Law are monitored. The points by Shiga Prefectural Government are the environmental standard points for COD_{Mn} (8 points) and ones for nitrogen (T-N) and phosphorus (T-P) (4 points). At these points, the achievement situations of the Environmental Quality Standards (EQSs) are observed. The bathymetric water quality monitoring is made at 4 points twice a month. Environmental Water Quality Standards for protecting the living environment and plankton researches, etc. are monitored.

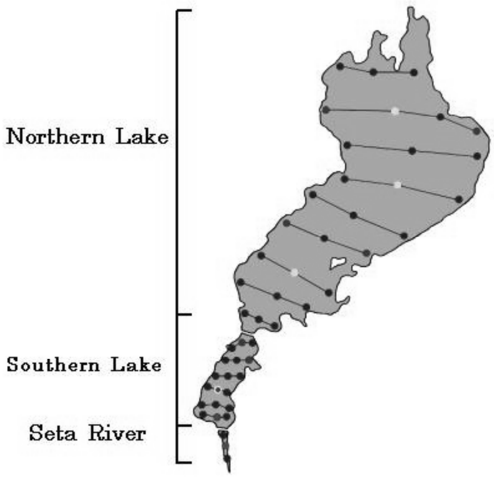

Figure 2. Monitoring Points of Water Quality in Lake Biwa.

The monitoring of the inflowing rivers is carried out at 34 points in 29 rivers once a month. Environmental Water Quality Standards for protecting the living environment, one of protecting human health and monitoring substances are monitored as well as the surface water quality monitoring.

All the monitoring methods above mentioned are the regular monitoring. The other method is the automatic monitoring. It is operated 3 stations in the lake, 7 stations along shoreline of the lake and 8 stations in rivers. Those stations mainly monitor water temperature, pH, DO, COD_{Mn}, T-N, T-P, EC. Two stations in the northern lake among 3 stations in the lake operate the bathymetric water quality monitoring.

Purposes of monitoring

The future monitoring system needs to understand exactly the challenges and to obtain the information leading to the measures, in addition to the existing purposes.

Therefore, the following four points must be reinforced as objectives of tangible monitoring for faced challenges to the water environmental conservation.

- Grasp of the water environment from viewpoints of the quality, the quantity, the place and the time
- Grasp of the water environment through the change of the ecosystems
- Ownership of information and the reflection in the environmental conservation
- Correspondence for challenges by the flexible monitoring

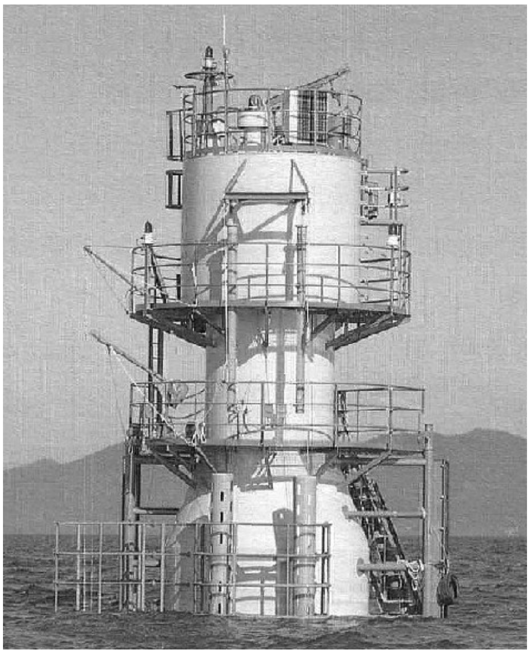

Figure 3. Water Quality Monitoring S Station in the Northern Lake.

Challenges of monitoring

The present monitoring cannot always effectively contribute on the explication of the water quality formation mechanism, because the purposes are basically to measure the concentrations of COD_{Mn}, T-N, T-P, etc. at the surface based on the frame of the water quality assessments like EQSs. Moreover, not grasping the material income and expenditure (the mass balance) of the water environment, we cannot understand the relationship between the measures for environmental impact's reduction in the basin and the water quality. It results in that the monitoring cannot be made good use of to the appropriate planning and implementing of the measures and the assessment to the effects of the measures.

The present monitoring is biased to the water quality, but we need to grasp comprehensively the water environment and to introduce new indicators like biological ones, easy to understand for citizens tackling with the environmental conservation.

Key concepts of the water quality monitoring

Grasp of the water environment from viewpoints of quality, quantity, place and time

We need to investigate the water quality formation mechanism in Lake Biwa and the basin to plan and implement the appropriate measures for addressing the problems and to assess the effects.

For this purpose, it is necessary to grasp exactly the water environmental situation through the accurate grasp of the mass balance in the water environment.

In the deeper lakes like the northern lake, vertical material transitions have a big influence in the water quality formation. Therefore, we need to change the dynamic and dimensional monitoring from the static and horizontal one. In other words, we have to grasp the material behaviors in the water environment not only by horizontal measurements, but also by bathymetric measurements, and the amount of the outflowing pollution loads through the southern lake. It is also important to grasp the exact inflowing pollution loads by the quantity as well as the quality in rivers. We need to select the suitable measurement methods and operate monitoring with their combinations, taking account of each characteristics, so that the regular monitoring to set up measurement points and items flexibility corresponding to challenges and to manage always the precision at high levels, and the automatic monitoring to grasp the short-time water quality changes.

Grasp of the water environment through change of ecosystems

The water of Lake Biwa is not only the water resource for living water, agriculture water and industrial water, but also important component factors for the ecosystems. Therefore, it is important to make living things like fishes, plants, planktons, etc. indicators and to grasp changes of the soundness of the water environment by the long-term monitoring. On the other hand, making common animals and plants indicators is effective to promote the monitoring by residents.

Moreover, it is necessary to grasp the soundness of the water environment by the long-term monitoring, including changes of shores' situations and the land uses of the basin influenced importantly on the inhabiting of living things.

Acquiring information and reflection in environmental conservation

Now, the results of the regular monitoring by the administrations are put together once every year, announced in the environmental white paper of Shiga prefecture, and also provided the homepage for public reading. The indicators are limited to physical and chemical ones, the announcement is only numerical values not to be analyzed, so we cannot make good use of it.

Further announcements need positively and immediately to be referred to the plan and the implement of projects influencing on the water environment, to be usually positively considered the water environmental changes by residents and to apply for various measures to the water environmental conservation by residents, as well as to conserve the water environment. Moreover, the data should be made good use of planning and implementing policies to develop the sustainable society in the area.

By means of the ownership of information among administrations, research institutes and citizens, it is also necessary to establish systems to promote researches.

Corresponding for challenges by flexible monitoring

The basis of the monitoring to grasp the changes of the water environment is to be implemented intentionally and continuously from the long-term vision. On the other hand, it is important to select the appropriate research methods to grasp the effectively information by the projective researches, in order to examine the water quality model, to understand the situations of the basin and the lake and to make clear objectives from the socio-economic viewpoints.

Actions for the future water quality monitoring

Grasp of the material income and expenditure (the mass balance)

The mass balance of the pollution loads amount and their behavior in the lake appear as the water quality. Consequently, to grasp this mass balance is important to predict the future water quality of Lake Biwa and to verify the measures' effects.

The present monitoring is operated at a clear weather and cannot grasp outflowing peaks of rivers on a rainy day. Accordingly, we continue investigating to grasp the pollution loads amount by consecutively measurements of inflowing pollution loads from rivers in the quality and the quantity, the material transitions of the vertical direction in the northern lake and the outflowing amount of pollution loads through the southern lake. Moreover, we verify the unit using for the Plan for Conservation of Lake Biwa Water Quality and investigate the data to support the unit system. We also discuss that as the new indicator to support the present COD_{Mn}, TOC having the addition is positively placed to investigate the water quality formation mechanism and applied. Furthermore, we investigate the grasp of the mass balance of the water quantity and the bottom of the lake in addition of the regular measurement items.

Implementation of continuous water quality investigations

We continue operating the regular monitoring in Lake Biwa and the rivers for the observation of the achievement situations of EQSs, with reconsidering the measurement points, items and frequencies. We also regularly and continuously go on with the plankton researches.

We continue the bathymetric water quality monitoring, with adding the necessary modification to investigate the water quality formation mechanism.

Regarding the important short-term water quality changes to investigate the water quality formation mechanism, not obtained by the regular monitoring, the automatic monitoring at the present stations in the lake is continued, with adding the necessary modification such as measurement items, methods and so on.

Implementation of monitoring connected with modeling

We are working to structure the model for Lake Biwa to predict the water quality with the following purposes.

- To explain the past changes appearing in the past researches' results
- To predict the future water quality changes with the consideration of the effects of conservation's measures and socio-economic changes
- To make good use of data for decision making policies and supports of verifications, including the use by citizens and industries

Analyzing the water environmental factors of Lake Biwa and pigeonholing the available and the future preparing data, we try to make the best model at this point and prepare to enrich the future model. Moreover, based on this model, effective monitoring will be operated at the suitable points and frequencies for the research.

Addition of new indicators

In addition of the present Environmental Water Quality Standards, we investigate the introduction of new appropriate indicators as organic pollution ones, corresponding with water environmental challenges, such as TOC to grasp the mass balance. With the expansion of viewpoints from water quality conservations to water environmental conservations, we try to introduce biological indicators, besides the present chemical ones, from a point of view of the citizens' participation. We investigate the change to regular researches at suitable intervals from 5 to 10 years from the present projective researches at the bottle of the lake, and the introduction of indicators corresponding with the land use and socio-economic factors in the basin to assess the soundness of Lake Biwa and the basin.

Monitoring corresponding with emergencies

We arrange the monitoring system with the consideration of measurements' assumptions to ready for water accidents caused by traffic accidents and accidents in factories, and of emergency measurements, communications and emergency treatments after accidents' occurrences. We investigate the management methods of chemical substances using data in factories based on the Pollutant Release and Transfer Register and aquatic data in the basin by GIS.

Improvement of the precision of monitoring

We always try to improve the accuracy of the monitoring, with coming in sight of the progress of the global warming, impacts from new materials with changes of industrial structures and our lifestyles, improvements of measurement methods and equipments using IT and bio-technology.

In order to objectify the water quality of Lake Biwa, we investigate to select the aquatic area not to have human influences in the basin, to take the water quality data as a benchmark.

Announcements and applications of results

The monitoring results by administrations should be used not only for the planning and implementing policies by administrations, but also for various activities by residents and industries, and activities by researches and specialists. It is also important to encourage residents themselves to have daily concerns about the water environment, to grasp environmental situations by themselves and to apply their measures by means of monitoring results. Accordingly, we try to improve the data disclosure protocol. We try to provide citizens the data to apply as means of the determination of policies and assessment of the results.

Partnership with citizen's activities in the basin

The monitoring by citizens, functioning as a part of activities for water environmental conservation, is important not only for actions' basis, but also for administration institutes and researchers. We investigate systems to support, improve and enrich the monitoring activities by citizens.

Conclusion

Lake Biwa is in the midst of the hearts of citizens of Shiga as well as all people of downstream area who receive benefit from the water. At the same time, Lake Biwa is not only a regional resource but also a national property, and an endowment for future generations.

Appreciating the national importance of Lake Biwa, the Shiga Prefectural Government makes further efforts for conserving a preferable lake environment. For that purpose, science and research activities play a vital role. The complicated mechanisms of lake water quality need to be studied by deploying state-of-the-art monitoring facilities. New information from research can help to refine policy action. The experiences here could be helpful not only for Shiga but also other lakes and water bodies around world.

Acknowledgements

We thanks to the members of the investigation committees; Professor Shigeo FUJII, Professor Hiroshi TSUNO and Lecture Hideaki NAGARE in Department of Urban and Environmental Engineering, Graduate School of Engineering, Kyoto University, Professor Masahide YUMA in Department of Environmental Solution Technology, Faculty of Science and Technology, Ryukoku University and Assistant Professor Takanori MASUDA in Department of Social Systems Engineering, Faculty of Engineering, Tottori University.

References

Shiga Prefectural Government, 2004. *The Environmental White Paper of Shiga Prefecture.* Available from URL: http://www.pref.shiga.jp/biwako/koai/hakusyo/index0.htm

Shiga Prefectural Government, 2005. *Vision for Water Quality Monitoring of Lake Biwa and the Basin*

Water level regulation as a tool for improving the ecological state and management of shallow Lake Võrtsjärv

Arvo Järvet

Institute of Geography, University of Tartu; Vanemuise 46, 51014 Tartu, Estonia

Abstract

The shallow Lake Võrtsjärv (mean depth 2.8 m) ecosystem and the surrounding areas are strongly affected by large fluctuations of the water level in the lake (mean annual amplitude 1.4 m, maximum amplitude was observed 3.1 m). Due to the shallowness of the lake, low-level periods are accompanied by several unfavourable phenomena such as overgrowth by macrophytes, restricted spawning area for pike, and winter fish kills. The low-water periods are more dangerous to the ecosystem. They cause an increase of resuspension, accelerate nutrient cycling and improve water column illumination. This leads to massive growth of planktonic algae and submerged macrophytes. The reed belt enlarges due to vegetative colonisation of new shallow regions as well as generative reproduction in emerging dry areas. Low water level also causes winter oxygen depletion due to a significantly decreased oxygen storage capacity and higher amount of easily degradable organic material produced during the vegetation period. Several winter fish kills in Lake Võrtsjärv (in 1939, 1948, 1967, 1969, 1978, 1987) have been documented during 20. century. Most of them (1939, 1948, 1967, 1969) coincided with low-level periods and hence with a higher primary production in the preceding summer, and oxygen depletion during winter.

- In order to define the regulated water levels on a month-by-month basis, the following important factors are considered:
- ecological state and fishery-related recommendations for achieving as high spring and summer water levels as possible;
- reducing the seasonal and long-term fluctuations in the water level;
- seasonal variations in the natural water level of the lake;
- raising the minimum water level without raising maximum water level;
- use of lakeshore areas.

The level of spring high water is the most important regulated water level, which also affects water levels in the following months. In order to guarantee favourable conditions for fish spawning (i.e. the existence of extensive low water areas), the average water level in April should be 34.50 meters above sea level. During the second half of May, the water level can already be much lower. The recommended regulated mean water level for May is therefore 34.00 meters.

Key words: L. Võrtsjärv, water level, water management

Mean features of water regime

The characteristic features of the annual hydrological cycle of Lake Võrtsjärv are a low water level in winter and a high water level in spring, which decreases gradually during summer and early autumn and is followed by a smaller peak in late autumn. The daily variation in water level is measured in centimetres, the monthly variation in decimetres, but the annual amplitude exceeds 2 m. The average annual fluctuation amplitude of the water level is 1.34 m. The maximum value was recorded in 1951 (2.02 m), the minimum value in 1925 (0.75 m). The absolute amplitude for the observation period since 1922 was 3.08 m, with the maximum water level of 35.28 m on 26 November 1923 and the minimum of 32.20 m on 6 September 1996 (Järvet 2000).

During the period studied, the discharge of the rivers normally peaked in the 1st decade of April, while the level in the lake lagged by one month, reaching its maximum normally in the 1st decade of May. During the spring maximum runoff period the changes in the water level of L. Võrtsjärv were mainly caused by hydraulic conditions in the outflow, as the lake level is hydraulically predominantly connected with the water level in the Emajõgi R. and in L. Peipsi. Because of a great inflow in April (25% of the annual inflow) and very low outflow (0.7% of the annual outflow), the water level remained above the annual mean level until the beginning of August. Spring high water level affected the level during the next 4–5 months, since the decline in the level after the peak period in May lasted until mid-September, and it also had a causal effect on the autumn maxima in November and December.

Fluctuations in the water level in L. Võrtsjärv over the years are considerable and seemingly quite random. Long-term water level measurements in L. Võrtsjärv show a sinusoidal alternation of low and high water states (Figure 1). Long-term changes in water regime can be seen most clearly in the dynamics of the minimum annual water level.

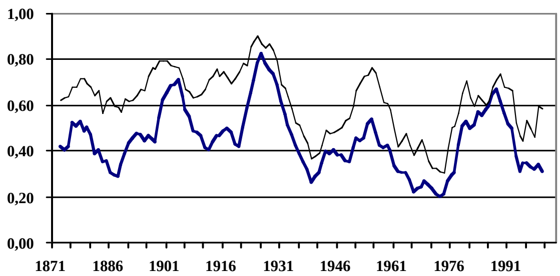

Figure 1. Long-term variation of annual maximum (normal line) and minimum (bold line) water level of L. Võrtsjärv in 1871–2003, smoothed by a 6-year filter.

The period since the beginning of the 1960s has been relatively dry in comparison with earlier times, which is reflected in the long-term water level curve, with more frequent and longer low level (below 33.00 m) periods. Provided that the same periodicity continues, it is possible to forecast the water regime in the near future.

Influence of water level on the ecological state and fishery

The shallow Lake Võrtsjärv ecosystem and the surrounding areas are strongly affected by large fluctuations of the water level in the lake (mean annual amplitude 1.4 m). At a high water level, large surrounding areas are flooded. This causes problems mainly for agriculture and forestry. The flooding of meadows surrounding L. Võrtsjärv starts at the water level of 34.5 cm BS. Low level cause significant changes in the littoral, which is the most visible part of the lake ecosystem for normal lake and shoreline zone users. Lakeside areas are usually inundated from mid-April to the second half of June. In 1965 macrophytes covered only 15% of the lake area (Mäemets 2002), since then, their area has expanded remarkably (Haberman et al., 1998). Due to the prevalence of western winds, the reed belt (mainly *Phragmites australis* and *Schoenoplectus lacustris*, is continuous and lush on the sheltered western shore, and broken on the open eastern shore. The narrow southern end of L. Võrtsjärv is fully populated with macrophytes (*Nuphar lutea* and *Potamogeton lucens* prevail).

Ecological state. Due to the shallowness of the lake, low-level periods are accompanied by several unfavourable phenomena such as overgrowth by macrophytes, restricted spawning area for pike, and winter fish kills (Nõges & Nõges 1998). The low-water periods are more dangerous to the ecosystem. They cause an increase of resuspension, accelerate nutrient cycling and improve water column illumination. This leads to massive growth of planktonic algae and submerged macrophytes. The reed belt enlarges due to vegetative colonisation of new shallow regions as well as generative reproduction in emerging dry areas. Low water level also causes winter oxygen depletion due to a significantly decreased oxygen storage capacity and higher amount of easily degradable organic material produced during the vegetation period. The years with less precipitation are associated with colder winters, when ice cover is thicker and stays longer (Table 1).

Table 1. The distribution of years by minimum wintertime water level and ice cover parameters

Characteristic	Extr. high	Very high	High	Middle	Low	Very low	Extr.low	Total, *Mean*
Number of winters	1	10	16	18	16	8	10	79
Mean ice thickness cm	29	40	49	51	50	53	60	53
Mean ice cover duration, days	115	126	129	130	127	132	143	130

External loading to Lake Võrtsjärv has long been high, leading to a nearly hypereutrophic state with very high internal loading. Therefore the reduction of external loading alone is not efficient to improve the situation. The enhancing of the reduction of internal loading is needed for the restoration of the lake. The most effective method of restoration that can be implemented in practice is the prevention of excessively low water levels. To a certain extent, water-level management can provide a useful tool for lake restoration. It has been found that water levels should not fall below the value of 33.00 m. In very dry years this is impossible, but by means of regulation it is possible to reach a favourable water level from the point of view of trophic status. However, efforts should also be made to reduce external loading.

Water level and fishery. The great role of physical processes in regulating the L. Võrtsjärv ecosystem is obvious. This fact has been mentioned by most earlier investigators. Intensive sediment resuspension as the main factor causing low water transparency was shown by Mühlen & Schneider (1920). Pihu (1959) related the spawning success of pike with differences in the height of the spring flood. He also cited the observations of local fishermen concerning the rapid expansion of reed areas after the low water period in 1939–1940.

Several winter fish kills in Lake Võrtsjärv (in 1939, 1948, 1967, 1969, 1978, 1987) have been documented during 20. century (Table 2). Most of them (1939, 1948, 1967, 1969) coincided with low-level periods and hence with a higher primary production in the preceding summer, and oxygen depletion during winter (Table 3).

Table 2 : Water level and ice cover conditions during fish-kill winters. * No data.

Year	Water level	Ice thickness	Duration of ice cover	End of ice cover	O_2 in bottom, mg l^{-1}
1939	Very low	Very thick	Medium	Medium	*
1948	Extremely low	Thick	Long	Medium	*
1967	Very low	Medium	Long	Medium	*
1969	Extremely low	Thick	Very long	Late	0.3
1978	Low	Very thick	Long	Vary late	1.1
1987	Medium	Very thick	Medium	Late	0.8
1996	Extremely low	Thick	Extr. long	Vary late	0.0

Table 3 : Hydrological parameters of Lake Võrtsjärv during fish-kill winters. All data are presented as deviations from long-term mean values.

Year	h of max. ice thickness, cm	W of total volume, 10^6 m^3	W of active volume, 10^6 m^3	h of active depth, cm
1939	+10	-71	-95	-0.28
1948	+4	-234	-238	-0.74
1967	+1	+3	+3	+0.02
1969	+6	-183	-208	-0.64
1978	+15	-21	-57	-0.17
1987	+14	+40	+5	+0.02
1996	+7	-240	-268	-0.84
Long-term mean	53	727	588	2,25

Water level is one of the main factors determining the success of spawning and therefore the abundance and catches of many fish species. In L. Võrtsjärv, the water level has a particularly important influence on the prosperity of pike, which lays its eggs in shallow flooded places (generally up to 0.5 m), mostly on dead vegetation. As the water level is kept relatively low during the early spring, the spring flood is much lower. This mainly affects the reproduction of spring spawning fish, because most spawning areas are not available during the spawning period.

In the event of high water levels, the spawning areas of pike are quite extensive, which lays a firm foundation for the formation of strong pike generation. There is an obvious positive correlation ($r=0.45$; $n= 30$; $p <0.01$) between the mean water level in spring and the pike catch in the lake (Järvalt & Pihu, 2004). As a rule, abundant pike catches follow, with a 4–5-year delay, periods of high water level and small catches occur, with the some delay, after periods of low water level. This is in accordance with the age composition of pike catches in L. Võrtsjärv, where 4–5-year-old specimens are usually predominant. On the basis of data obtained from an experimental trawl catch, a still stronger positive correlation ($r=0.61$; $n=23$; $p<0.01$) was found between the abundance of a certain pike generation and the water level in the lake during the spawning period (Järvalt & Pihu, 2002).

Principles of water level regulation

The unfavorable discharge conditions – the small slope of the Emajõgi River, the silting up of the mouth of the river and bifurcation between the beginning of the Emajõgi River and the Pede River cause long-term annual high water periods. The high water floods extensive coastal areas, hindering the agricultural use of those lands. However, this situation is rather favourable for the natural regulation of the discharge of the Emajõgi River. Figure 2 illustrates the variation in the number of days of high water and low water. The figure points out the need to regulate Lake Võrtsjärv, especially to raise its low water level, since years of low water tend to go in succession.

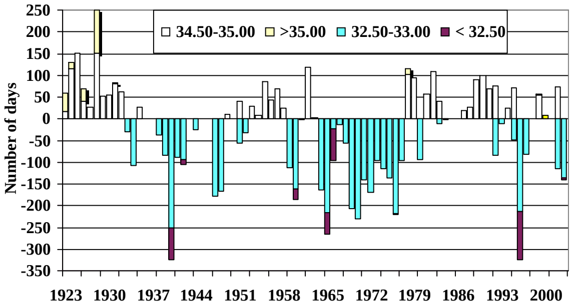

Figure 2 : Number of days per year of high and low water level in 1923–2003..

In order to define the regulated water levels on a month-by-month basis, the following important factors are considered:

- ecological and fishery-related recommendations for achieving as high spring and summer water levels as possible;
- reducing the seasonal and long-term fluctuations in the water level;
- seasonal variations in the natural water level of the lake;
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The level of spring high water is the most important regulated water level, which also affects water levels in the following months. In order to guarantee favourable conditions for fish spawning (i.e. the existence of extensive low water areas), the average water level in April should be 34.50 meters above sea level (Figure 3). During the second half of May, the water level can already be much lower. The recommended regulated mean water level for May is therefore 34.00 meters.

During the low water seasons, both in summer and autumn as well as in winter, it is important to maintain the water level at a minimum of 33.50 meters. Months considered to have such water level are February, July and August. The regulated water levels of the remaining months are seen as transition levels from high and low water situations, considering the natural peculiarities of the Lake

Võrtsjärv water regime, such as the slow change in water level, especially in the fall high water period. In order to have the recommended water level of 33.50–33.70 meters for winter (January, February and March), the water level has to be at least 34.00 meters in November and December (Järvet 2004). The mean annual regulated water level of 33.80 is an average of recommended monthly water levels.

The author of this article has conducted a comparison between the natural long-term water level of Lake Võrtsjärv (from 1922 to 2002) and the recommended regulation levels (Figure 3). The need to regulate Lake Võrtsjärv can be drawn from the comparison between recommended water levels and the actual water levels of the observation period. That will give us an overview of the period during which the water level is below the recommended level. Considering the relatively slow fluctuations in the water level of Lake Võrtsjärv and the unfavourable discharge circumstances, the comparison is based on mean monthly water levels.

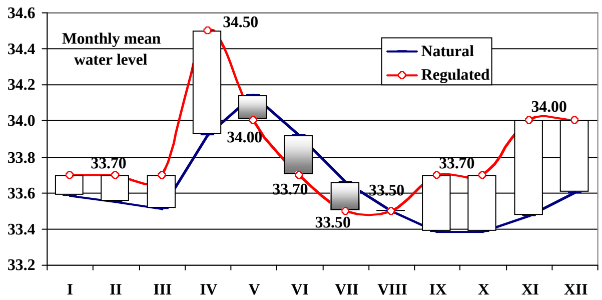


Figure 3. Seasonal variability of natural and recommended monthly mean water level on Lake Võrtsjärv.

The regulated water level would follow natural conditions but at higher levels and in a more regular way. The projected water level during the summer low water period would be 50–60 cm higher than the natural level. However, the use of a very precise and flexible regulation scheme is not possible because of the limited penetrability of the Emajõgi River.

The yearly regulating scheme for the Lake Võrtsjärv can be divided into four cycles:

1. Water level decreasing in winter period (in January and February). It is necessary to guarantee that the lake would hold the spring high water in order to avoid big floods.
2. Spring water accumulation and passing through high water in March and April. If high water amount is small, work-down is negligible.
3. Summer water level decreasing from May to October. Discharge in the River Emajõgi depends on navigation conditions.
4. Autumn-winter water accumulation from October to December. Limiting discharge of Emajõgi River (14 m^3/s) is let into the river, and it is also guaranteed during all the other cycles.

There has not been much discussion of the increasing risk of flooding during water-abundant periods if the water level of Lake Võrtsjärv is raised above the natural level. The regulation facilities will not be able to avoid the damage caused by floods, and the risk of flooding could even increase to a certain extent. Despite the fact that in water-abundant years, when Lake Võrtsjärv would not be regulated because of the high water levels of the upper course of the Emajõgi River and its tributaries, Põltsamaa and Pedja, the natural regime would be maintained, extensive floods could not be avoided.

In a very long-term perspective, the natural conditions of the discharge of Lake Võrtsjärv could deteriorate because of neotectonic movement, as the land uplift on the northern coast of the lake exceeds the uplift in the southern part by 0.2–0.3 mm a year. Neotectonic uplift is also the reason for the paleogeographical lowering of the northern part of the lake and the slow inclination of the water towards the south.

References

Haberman, J., Nõges, P., Nõges, T., Pihu, E., Kangur, K., Kangur, A. & Kisand, V., 1998. Characterization of Lake Võrtsjärv. *Limnologica,* 28, 1, 3-10.

Järvalt, A., Pihu, E., 2004. Influence of water level on the stocks and catches of fish in Lake Võrtsjärv. Fisheries Management and Ecology, Special issue: An International Symposium on Ecohydrology and Physical Fish Habitat Modifications in Lakes. *Mondsee,* Austria, 26–28 November. 2003.

Järvalt, A. & Pihu, E., 2002. Influences of water level on fish stocks and catches in Lake Võrtsjärv. *Proc. Estonian Acad*. Sci. Biol. Ecol. 51, 1, 74-84.

Järvet, A., 2002. Water regime of Lake Võrtsjärv. *In* T. Kaare and J.-M. Punning eds. *Estonia, Geographical Studies, 8*. Estonian Academy Publishers, Tallinn: 72–88. 455.

Järvet, A., 2004. Influence of hydrological factors and human impact on the ecological state of shallow Lake Võrtsjärv in Estonia. *Dissertationes Geographici Universitatis Tartuensis* No 19, Tartu University Press, Tartu, 230 p.

Mäemets, H. 2002 Commented list of macrophyte taxa of Lake Võrtsjärv. Proc. Estonian Acad. Sci. Biol. Ecol., 51, 1, 5-25.

Mühlen, M. von zur & Schneider, G., 1920. Der See Wirzjerw in Livland. Archiv für die Naturkunde des Ostbaltikums, 14: 1–156.

Nõges, P., Nõges, T. (1998) The effect of fluctuating water level on the ecosystem of Lake Võrtsjärv, Central Estonia. *Proc. Acad*. Sci. Estonia Ecol. 47, 2: 98–113.

Pihu, E., 1959. Kalade kudemisest Võrtsjärves 1956. ja 1957. aasta kevadel. *In Eesti NSV Teaduste Akadeemia juures asuva Loodusuurijate Seltsi aastaraamat*, 51, 225-248.

The ecological state of Lake Naivasha, Kenya, 2005: Turning 25 years research into an effective Ramsar monitoring programme

K.M.Mavuti[1] & D. M. Harper[2]

[1] Department of Zoology, University of Nairobi, Kenya

[2] Department of Biology University of Leicester, England

Introduction

Lake Naivasha (0. 45ºS, 36. 26ºE), altitude 1890, lies in the Eastern Rift Valley and currently covers approximately 100 km^2. It is the second-largest freshwater lake in Kenya (after the Kenya portion of Victoria). It is one of a series of 23 major lakes in the Eastern Rift Valley – eight in central Ethiopia, eight in Kenya and seven in Tanzania – spanning latitudes from approximately 7º N to 5º S. The overall climate of the Eastern Rift Valley is semi-arid. Most Eastern Rift Valley lakes are thus alkaline or saline. Lake Naivasha is unique within the central latitudes of the valley in being fresh, and indeed within the Kenyan series of lakes (from north to south are Turkana, Baringo, Bogoria, Nakuru, Elmenteita, Naivasha, Magadi), with a conductivity fluctuating between 250-450 µS cm^{-1} (Figure 1, Figure 2).

The lake has international value as a Ramsar wetland, which in the last two decades has grown to become the main site of Kenya's horticultural industry, one of the largest earners of foreign currency (Harper & Mavuti, 2004).

Lake Naivasha has no surface outlet and the natural fluctuation in water levels over the last 100 years has been in excess of 12 metres. The lake fills a shallow depression with gentle slopes, so an increase in lake level greatly increases the lake surface area and consequently increases evaporation and it shows a very dynamic behaviour (Figure 3). The water levels of the lake respond to long-term wet and dry climatic cycles, with annual variations of the water levels are superimposed on these long-term variations. The water level can change several metres within a few months, causing a horizontal change of several kilometres. These hydrological dynamics add an extra dimension to the riparian ecosystem as well as to the water resource management issues. British colonial law defined the lake edge by the level at 6,210 ft a.s.l. (1892.8m a.s.l.) and enabled riparian owners to cultivate the lake bed below this contour, but with no permanent structures. The lake water was used at this time to irrigate only small areas of fodder crops and vegetables, provide water for cattle, and the domestic supply to a small population. In 1929, the Lake Naivasha Riparian Owners Association (LNROA), was formed and had the legal right to resolve disputes between adjacent landowners over their use of this riparian lake bed (Enniskillen, 2002).

In the early 1980s, successful experiments in production of cut flowers led to the growth of a horticultural industry dependent upon irrigation water. Since the first flower farms started there has been a fairly constant increase of the area cultivated, more rapidly in the last five years, to occupy now about 4,000 ha (Becht, unpublished). There now is uncertainty as to whether the lake can sustain the corresponding increase in demand for irrigation water, because the lake water is treated as a 'common good' available to all, when it is a very finite resource.

Around 1990 the LNROA became more proactive, commissioning two consultants' reports on the scientific status of the lake (Goldson, 1993, Khrodha, 1996), lobbying for the declaration of the lake as a Ramsar site (then Kenya's second), which was achieved in 1995. It changed its name to Lake Naivasha Riparian Association (LNRA) and added Associate member level to bring in a wider membership. In 1999, the LNRA's 70th anniversary, the organisation received the prestigious Ramsar Wetlands Conservation Award in the NGO category, for their conservation work on the lake (LNRA, 1999).

At the end of the 20th Century, a major international conference – 'Science and the Sustainable Management of Tropical Waters' – was held in Naivasha KWS Training Institute and the two publications that followed (Harper & Zalewski, 2001; Harper *et al.,* 2002b) brought all the scientific studies of the past decade into the public domain. Since that time a summary of the management issues that are needed to be implemented in order to make the lake sustainable has been published (Harper & Mavuti, 2004). This present 2005, 11th World Lakes Conference, offers another opportunity for synthesis of knowledge in order to move into a sustainable phase. The present paper summarises the state of ecological knowledge in the lake based upon the research of the teams of Harper & Mavuti, funded by the Earthwatch Institute since 1987. It suggests how the research knowledge should drive the Ramsar monitoring programme that is now evolving.

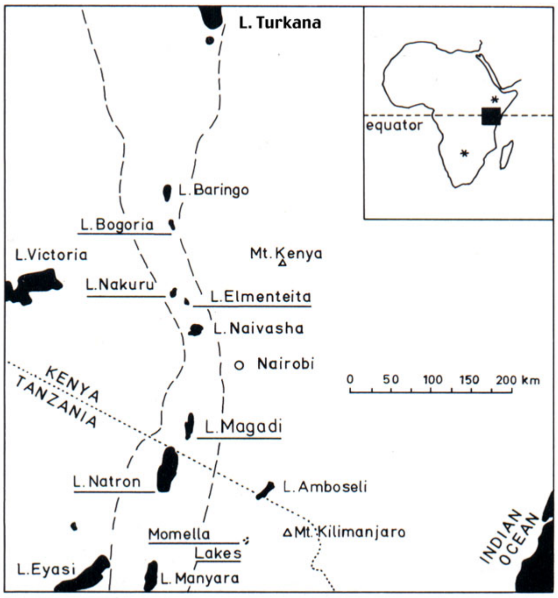


Figure 1. Lakes of the Kenyan Eastern Rift Valley. Saline lakes are underlined. From Harper et al, 2003.

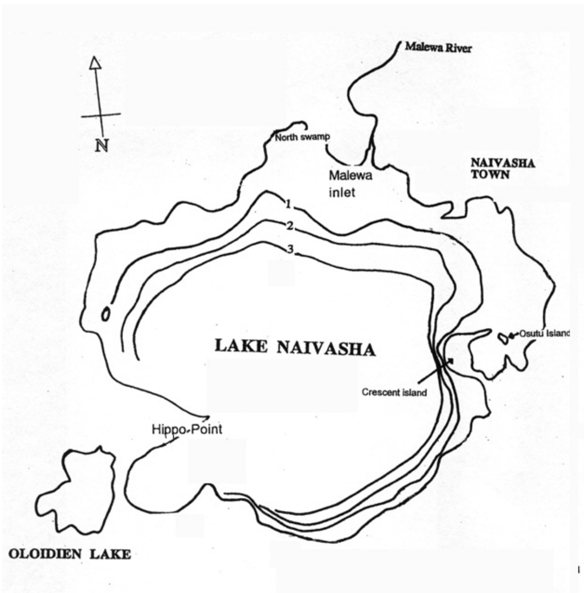
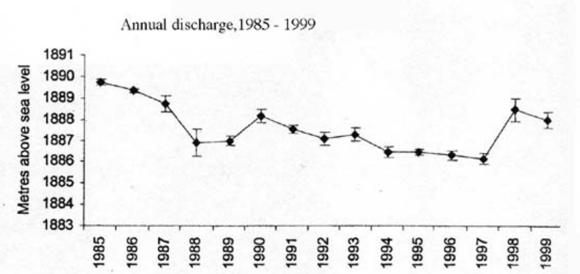
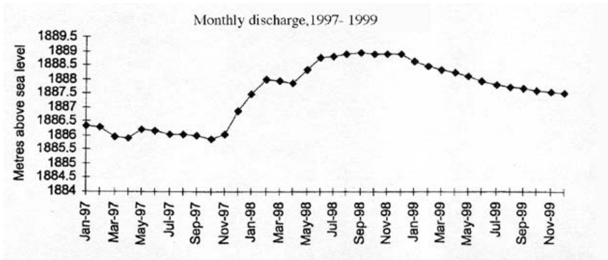


Figure 2. The Lake Naivasha catchment, showing the spatial arrangement of the three main inflowing rivers. (From Harper & Mavuti, 2004).



Lake level, 1985-99.



Monthly lake level, 1997-99

Figure 3. Bathymetry of lake Naivasha in 1991 (upper, depth in metres), annual lake level changes (middle) and monthly changes during the 'El Niño' rains (lower). From Kitaka, Harper & Mavuti, (2002).

Present day ecology

The lake supports an ecosystem with high but uneven biodiversity – rich in birds and plants but no native fish, for example (Harper, Mavuti & Muchiri, 1990). The artisanal commercial fishery is dependant upon exotic species and is currently in severe decline due to overfishing (see Britton et al, this volume). An exotic crayfish, introduced in 1970 to diversify the fishery, has completely re-structured the lake's ecology (Harper *et al,* 2002a).

There are four, chemically distinct, basins at Naivasha (Gaudet & Melack, 1981). Crescent Island Basin, a small extinct volcanic cone, is the deepest part of the lake (up to 18m depth), usually connected to the main lake over a shallow lip. The main lake has a maximum depth of 6m at its southern end (see Figure 3). Oloiden is a smaller crater lake with a depth of 5m to the south end of the main lake, which has been distinct from it since 1982, increasing in conductivity from 250 then to 3000µS cm^{-1} now. Crater Lake or Sonachi is located on the southwestern part of the lake and, in its own distinct volcanic crater, is a soda lake, fully independent from the main lake but its levels are believed to oscillate in harmony with the main lake as a result of groundwater connection.

Little natural vegetation is left in the catchment. The headwaters of the Malewa are situated in the Aberdare National Park and the adjoining gazetted forest. The vegetation consists of humid Afro-montane forest and bamboo. The Kinangop plateau was once large, grassland plains, but now an estimated 30% is now covered with maize or vegetables and scattered exotic, fast-growing tree species. Cultivation often extends right down to stream level and livestock freely water from streams in all but the gorges cutting through successive rift escarpments (Everard *et al.,* 2002) Remnants of the original forest exist in only small patches on the escarpment. The loss of native tussock-grassland threatens the endemic *Macronyx sharpei* (Sharpe's Longclaw) (Muchane, 2001). The Rift Valley floor was an open savannah landscape in the past, which is now only retained in Hells Gate National Park immediately to the south of the lake, and converted to ranch land or settlement elsewhere.

Little native aquatic vegetation exists in the lake any more. Gaudet (1977) described the floral biodiversity of a lake that was, then, largely unaffected by human impact. He showed how the diversity was dependent upon the hydrological fluctuations, which created a series of dynamic vegetation zones through the land-water ecotone. The most landward zone was occupied by *Acacia xanthophloea* (Fever Tree) woodland, its roots reaching the water table and its extent marking the highest water levels of the early 20th Century. This was succeeded by a rooted *Cyperus papyrus* (Papyrus) zone, which together with shallow water Papyrus 'reefs', had been created by germination following rapid water level rises, 3 times in the first half of the 20th Century. Extensive swamps, reefs and floating islands of Papyrus existed until the 1980s, with lagoons of the submerged aquatic plants *Potamogeton* spp*., Najas horrida, Ceratophyllum demersum* and the macro-algae *Chara* spp under 'carpets' of floating-leaved *Nymphaea nouchalii*. When water level declined, the exposed lakebed soils germinated many semi-aquatic species of grasses, sedges and herbs dominated by Compositae.

The situation in the water over the last twenty years has deteriorated markedly, almost entirely as a consequence of the introduction in 1970 of *Procambarus clarkii* (Louisiana Crayfish), which had destroyed the Lily beds and all submerged plants within 10 years (Harper, Mavuti & Muchiri, 1990). The water surface has been filled in their place by exotic aliens; first *Salvinia molesta* (Floating Water Fern) in the 1980s, latterly *Eichhornia crassipes* (Water Hyacinth) in the 1990s (Harper, Adams & Mavuti, 1995; Adams *et al.,* 2002). The submerged plants and small remnant populations of the water lily return to the lake through a sediment seed-bank germination when the crayfish population crashes through predation. This has happened twice recently, in 1987-94 and 2000-02.

The situation on the landward side of the ecotone, has also deteriorated markedly over the same time period, entirely due to the direct and indirect consequences of human impact. Papyrus has been reduced to around 10% of its former area (Figure 4; Harper & Mavuti, 2004) by a sequence of natural causes which have followed the lake level decline – the lake is estimated to be 3 vertical metres below its natural level (Becht & Harper, 2002). This lowers the water table under the papyrus, which dries out the swamp. It is then susceptible to extensive trampling of herds of *Syncerus cafer* (Buffalo) or cattle. Buffalo is a species which has increased 3-fold (to about 1500) in the riparian zone the past ten years, believed to be a result of forest clearance in the Eburru hills driving animals down to the lakeside (Harper & Mavuti, loc. cit.). Cattle have increased to many thousands, Maasai herds, which access the lake shore at one or two sites and then are driven along the lake edge with the tacit agreement of many landowners as long as they stay in the riparian zone. Once cattle and buffalo 'walkways' are made through the swamp, smaller animals follow, grazing any fresh shoots off, so the papyrus clumps die from lack of ability to photosynthesise once all the reserves in the rhizome are exhausted. Aggressive clearance by some horticulturalists using mechanical means or burning has added to the decline and there is severe destruction on those three lakeside sites where the general public has wide access.

These twin 'pincers' of degradation have combined to make the lake eutrophic since the early 1990s. Its phytoplankton has showed a seasonal shift between diatom and cyanobacterial dominance and its assemblage is now dominated by a persistent *Aulacoseira italica* population, both numerically and in terms of contribution to overall primary production. The concentrations of chlorophyll-a have increased from 30 $\mu g\ l^{-1}$ in 1982 to 110 $\mu g\ l^{-1}$ in 1988, and 178 $\mu g\ l^{-1}$ in 1995 and transparency has correspondingly declined to about 60 cm (but briefly rose to 160 cm in 1998-9 due to the diluting effect of the 'El Niño' rains) (Harper et al, 2002b). 170 algal and cyanobacterial species have been identified (Hubble & Harper, 2002). Most of the diatoms are indicators of moderate to high nutrient conditions. Total primary productivity of this phytoplankton population is approximately 160 mg C $m^{-3}\ hr^{-1}$ (Hubble & Harper, 2000). The sediments form a sink for phosphorus (Kitaka *et al.,* 2002), because they are rich in iron (Harper *et al,* 1995) and the main lake is well mixed and does not deoxygenate enough to release this store of nutrients. However, Crescent Island lagoon does stratify temporarily and hypolimnetic deoxygenation occurs. Phosphorus is then released from the sediments, a process not yet seen in the main lake. This indicates that the rate of primary production in the water column could double if conditions change to allow lake-wide nutrient release from sediments (Hubble & Harper, 2000). Kitaka, Harper & Mavuti (2002) showed that the lake did become 'hyper-eutrophic' on the OECD classification after the 'El Niño' rains in 1998, reverting back to eutrophic in 1999; this emphasises that most of the increase in trophic state of the lake comes from the wider catchment in the absence of the 'buffering' formerly provided in the North Swamp at the river inflows. The more alkaline Oloidien and Sonachi lakes are highly productive and *Arthrospira fusiformis* is significant in the latter.

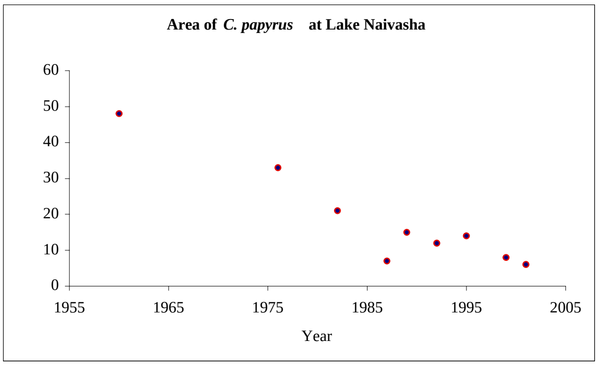


Figure 4. The area of *C. papyrus* at Lake Naivasha since 1960. Small increases occurred in the past decade, against the general decline, which correspond to wet seasons which caused rapid lake level rise (c.f Figure 4), re-flooding bare former lake bed, leading to new germination of new *C. papyrus*. From Harper & Mavuti, (2004).

Future prospects and the need for monitoring

One may summarise the ecological deterioration of Lake Naivasha in the latter 20th Century by a melodramatic "The Invasion of the Aliens" on water and the 21st Century as "The Invasion of the Wreckers". The LNRA is striving to promote sustainable wise use of the lake but is thwarted by the short-termism of individuals with selfish interest who can make money from the deteriorating ecological situation of the lake. The government is slow in implementing conservation legislation that would help arrest the deterioration. Nevertheless, there are certain 'bright spots' in the lake's ecosystem; for example the population of fish eagles, which had reduced to a little over 25 pairs in the mid-1990s due to shortage and availability of food, had increased by 50% on this low by 2000 after the 'El Nino' rains raised the lake level forming shallow lagoons in which fish bred rapidly. It increased again 2002-5 due to the ready availability of *C. carpio* (Britton et al., this volume) and its percentage of juvenile and sub-adult birds was as high as it has ever been in October 2005 (M. Harper pers. comm.; Figure 5).

Despite the 'bright spots', there is now an urgent need to quantify the (mostly negative) changes which are happening at Naivasha. The discussions of this paper, together with those of Becht (this volume) indicate 3 areas of monitoring:-

- Hydrology; quantifying the physical limits of the system and the balance between uses for humans/nature;
- Lake Ecology, quantifying whether the departure from naturalness (post-1970) is becoming worse or better and
- Land-Water Ecotone, quantifying the degree of degradation/recovery.

The hydrology and water balance has been dealt with by Becht (this volume) and earlier Becht & Harper (2002), with little change since that was written. The two ecological parts of the triad above should be undertaken as follows, from the new laboratory (2005) constructed for the LNRA by Sarah & Mike Higgins:-

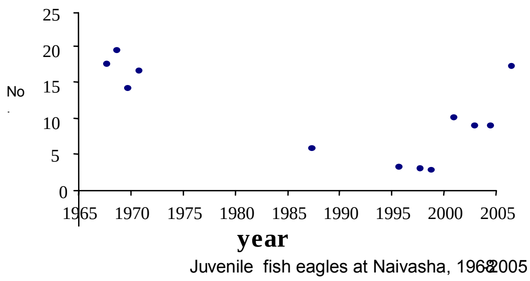
Lake Ecology. The basis for this would be quarterly boat survey undertaken by LNRA. This could also incorporate volunteer survey of birds for key bird species. Components of the survey would be:

A. Oxygen, conductivity, pH, alkalinity and Secchi depth (in boat)

B. Chlorophyll 'a', Phosphate and total P, Nitrate and Total N, dominant phytoplankton species. Dry weight and dominant species of zooplankton (in lab)

C. Crayfish catch/unit effort, Hyacinth vigour, Hyacinth beetle damage, beetles/hyacinth plant, submerged plant density & distribution (in boat).

D. Key birds – coot, and fish eagle; other piscivorous aquatic birds; other aquatic (in boat)



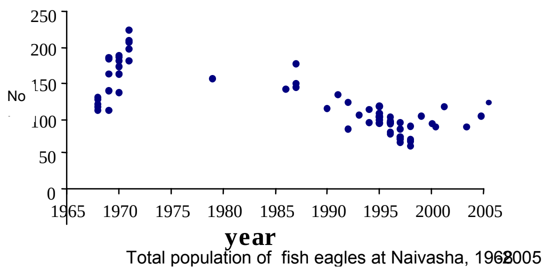


Figure 5. Population of all *Haliaeetus vocifer* (fish eagle) at Lake Naivasha (lower) and percentage of non-adult birds (upper picture) 1965-2005.

Land-Water Ecotone. The basis for this is the annual Earthwatch Institute research teams, which focus upon the following:

A. Biodiversity in Papyrus and Acacia

B. Fish eagle population structure, hippopotamus schools location & approximate size and shore vegetation

C. Riparian vegetation at fixed transect sites – it is suggested that Fishermans' Camp, Manera, KWS Annexe, and Oloidien lake provide a suitable range of stages of papyrus degradation.

D. Fish population structure (see Britton et al, this volume)

These should be superimposed upon the economic measures of lake health, such as commercial fishery and revenue streams from tourism, horticulture. The whole should be reported in the Annual Reports of the Lake Naivasha Management Committee, which is a legal entity enshrined by the Government of Kenya in November 2004, but at the time of writing not yet in force because of legal wrangling in Nairobi courts caused by few, selfish individuals from the Naivasha community.

References

Adams C.S., R.R.Boar, D. S. Hubble, M. Gikungu, D.M. Harper, P.Hickley & N. Tarras-Wahlberg 2002. The dynamics and ecology of exotic tropical floating plant mats: Lake Naivasha, Kenya. *Hydrobiologia* 488 (*Developments in Hydrobiology* 168): 115-122.

Becht R & Harper D. M., 2002. Towards an understanding of human impact upon the hydrology of lake Naivasha. *Hydrobiologia* 488 (*Developments in Hydrobiology*168): 1-11.

Enniskillen, 2002. The Lake Naivasha Management Plan – consensus-building to conserve an international gem. *Hydrobiologia* 488 (*Developments in Hydrobiology* 168): ix-xii.

Everard, M, Kuria, A., Macharia, M., Vale, J. & Harper, D.M. 2002. Aspects of the biodiversity value of the Lake Naivasha catchment rivers. *Hydrobiologia* 488 (*Developments in Hydrobiology* 168); 13-25.

Gaudet, J. J., 1977. Natural drawdown on Lake Naivasha, Kenya and the formation of Papyrus swamps. *Aquatuc Botany* 3: 1-47.

Gaudet, J. & J. M. Melack, 1981. Major ion chemistry in a tropical African lake basin. *Freshwater Biology* 11, 309-333.

Goldson, J. 1993. *A Three Phase Environmental Impact Study of Recent Developments around Lake Naivasha*. Lake Naivasha Riparian Owners' Association, Naivasha, December 1993: 109 pp.

Harper, D.M., K.M. Mavuti and S.M. Muchiri, 1990. Ecology and management of Lake Naivasha, Kenya, in relation to climatic change, alien species' introductions, and agricultural development. *Environmental Conservation* 17: 328-335.

Harper, D.M., Adams C. & K. Mavuti, 1995. The aquatic plant communities of the Lake Naivasha wetland, Kenya: Pattern, dynamics and conservation. *Wetlands Ecology and Management* 3: 111-123.

Harper D.M., G. Phillips, A. Chilvers, N. Kitaka & K.M. Mavuti, 1993. Eutrophication prognosis for Lake Naivasha. *Verhandlungen Internationale Vereinigung Limnologie* 25; 861-865.

Harper & Zalewski,, 2001. *Ecohydrology: Science and the Sustainable Management of Shallow Tropical Waters.* IHP V Technical Documents in Hydrology 46 UNESCO, Paris. 114 pp. http://webworld.unesco.org/ihp_db/publications/

Harper D.M., R.R. Boar, M. Everard & P. Hickley, eds. 2002a. Lake Naivasha, Kenya. *Hydrobiologia* 488 (*Developments in Hydrobiology* 168) Kluwer, Netherlands, XXX 311 pp

Harper, D.M., M. M. Harper, M. A. Virani, A.C. Smart, R. B. Childress, R. Adatia, I. Henderson, & B. Chege 2002b. Population fluctuations and their causes in the African Fish Eagle, (*Haliaeetus vocifer* (Daudin)) at Lake Naivasha, Kenya. *Hydrobiologia* 488. *Developments in Hydrobiology* 168; 171-180.

Harper D.M., A. C. Smart, S. Coley, S. Schmitz, R. North, C.Adams, P. Obade & M. Kamau., 2002c. Distribution and abundance of the Louisiana red swamp crayfish Procambarus clarkii Girard at Lake Naivasha, Kenya between 1987 and 1999. *Hydrobiologia* 488 (*Developments in Hydrobiology* 168); 143-151

Harper, D.M., R.B. Childress, M.M. Harper, R.R. Boar, P. Hickley, S.C. Mills, N. Otieno, T. Drane, E. Vareschi, O. Nasirwa, W.E. Mwatha, J.P.E.C. Darlington, X. Escuté-Gasulla, 2003. Aquatic biodiversity and soda lakes: Lake Bogoria National Reserve, Kenya. *Hydrobiologia* 500; 259-276.

Harper, D.M. & K.M. Mavuti, 2004. Lake Naivasha, Kenya: Ecohydrology to guide the management of a tropical protected area. *Ecohydrology & Hydrobiology* 4; 287-305

Hubble D.S. &.Harper D.M,, 2000. Control of primary production in a tropical shallow lake. *Verhandlungen Internationale Vereinigung Limnologie* 27; 920-923.

Hubble & Harper, 2002. Phytoplankton community structure and succession in the water column of a shallow tropical lake (Lake Naivasha, Kenya). *Hydrobiologia* 488 (*Developments in Hydrobiology* 168): 89-98.

Khrodha, G. 1994. *A Three Phase Environmental Impact Study of Recent Developments around Lake Naivasha II*. Lake Naivasha Riparian Owners' Association, Naivasha, P.O. Box 1011 Naivasha, Kenya.

Kitaka, N., Harper, D.M. & Mavuti, K.M., 2002. Phosphorus inputs to Lake Naivasha, Kenya, from its catchment and the trophic state of the lake. *Hydrobiologia* 488 (*Developments in Hydrobiology* 168): 73-80.

Kitaka, N. D. M. Harper N. Pacini & K. M. Mavuti, 2002 Chemical characteristics, with particular reference to phosphorus, of the rivers draining into Lake Naivasha, Kenya. *Hydrobiologia* 488 (*Developments in Hydrobiology* 168): 57-71.

Muchane, 2001. The conservation value of low intensity land use to maintain biodiversity. In Harper & Zalewski, 2001. *Ecohydrology: Science and the Sustainable Management of Shallow Tropical Waters.* IHP V Technical Documents in Hydrology 46; page 58. UNESCO, Paris.

LNRA, 1999. *Lake Naivasha Management Plan.* Lake Naivasha Riparian Association, P.O. Box 1011 Naivasha, Kenya.77pp

A new look at an experiment in developing and managing the French Jura department's lakes: the example of Le Frasnois district

Nedjai R.[1] and J. Demusy[2]

[1]Institut de géographie alpine – université joseph fourier grenoble france
[2]DESS eau Université Lyon 2 France

Abstract

For almost a century French lake systems have been subject to such pressure that most of them are now showing alarming signs of degraded quality and even of water reserves. These impacts are the result of profound changes in their processes, through either direct action such as drawing off drinking water, overfishing, and unsuitable development, local indirect action like road salting, and even regional and global factors of which air pollution is the main one. To deal with such a worrying situation a strong legal framework has been introduced and local management procedures set up. These involve a central role for local stakeholders, while a number of actions have been undertaken to reduce the pressure on lakes and drainage basins exerted by tourism, the exploitation of water resources, pollution due to farming and industry, etc. Many sites have been placed on the list of significant natural areas, programs like Natura 2000 and LIFE set up, and local protection zones delineated. Despite all this, there has been little noticeable improvement in the lakes, resulting in feelings of disquiet among managers who feel under-informed. The decline in benthic fauna and fish numbers in certain supposedly-protected lakes is the first sign that disturbance phenomena have intensified in the ecosystems. Slow, steady de-oxygenation is taking place, while the rates of certain harmful chemicals such as mercury and lead are increasing. This paper provides answers to managers' queries concerning the increase in such substances and describes the state of the physical and chemical quality of the lakes studied.

Key words: lake, eutrophication, pollution, benthic, ecosystem

Introduction

Efforts to develop and manage lakes have accelerated and intensified since the Water Act was passed on January 3 1992; the latter has since been reinforced by amendments in 2001 and the European Water Framework Directive. Aquatic ecosystems have since been made a priority in the management and protection of natural environments, and considerable means have been devoted to assisting managers. The strong legal framework has reinforced the prerogatives of local stakeholders and placed water use and environmental protection at the centre of discussion. Due to their strongly-felt heritage character, lakes have obviously featured in public debate. The search for balanced solutions between using them directly for drinking water supply, opening them up freely to leisure and tourism, and protecting them completely has therefore mostly led to serious conflict. In most cases, the inclusion of lakes in economic processes has been accepted in return for severe restrictions on farming and industrial activities in the river basins supplying the lakes.

The fragility of this type of environmental system is clear, and recent research by public and private organizations has brought to light symptoms of real degradation (Winiarski, 2000, Roos-Barraclough, F. and al 2003, Shotyk et al, 2001, C. Martin, 1992, Nedjai, 2004). Even high-altitude mountain lakes, hitherto considered as being far removed from potentially-harmful human activity, show traces of metals and pesticides. These findings argue in favour of the introduction of regional, and in some cases global, policies to reduce such pressure and initiate measures to rehabilitate lakes.

Many medium-altitude lakes, such as the Jura ones, have entered a stage of increased eutrophication, despite several years of development measures aimed at reducing the pressures they have been subjected to. A number of scientific studies have shown a clear degradation of the lakes and impoverished flora and fauna. This situation is causing problems for managers who are increasingly calling into question the work undertaken on lakes and river basins.

It was in order to understand the physical and chemical processes which control the overall functioning of lakes that a research project was initiated in 2000. The hydrological processes of the lakes of the Jura high plateau were first characterised and the degree of disturbance quantified by analysing the physical and chemical nature of the water and sediments. The results were then compared with the actions undertaken by managers from the 1970s onwards on the basis of scientific recommendations in order to assess their degree of success or failure. The lakes studied are part of the Hérisson catchment area and include Petit Maclu, Grand Maclu, Ilay and Bonlieu in the High Jura area, and Le Val downstream near the outlet.

Previous studies and the current state of the lakes

A number of different types of study over the last thirty years give a clear picture of the state of the lakes. A total of around thirty studies were carried out within the framework of research missions and direct requests by state agencies or simply at the request of local bodies concerned about their lakes. Most of the lakes suffer from severe oxygen depletion, especially of the lakebed. This situation, which was long attributed to the runoff from farming, has continued to worsen in spite of the efforts made to reduce the sources of pollution. There have been direct effects on the fauna as shown by the reduced density of certain fish species (the larger, carnivorous ones in particular) and a fall in fish catches over the last twenty years. Analysis of the phytoplankton shows a clear loss of abundance and diversity in certain parts of the lakes (DIREN Fr-Comté, 2004; Cemagref, 2004; et Verneaux, 2001). Similar declines have also been observed among the benthic fauna (e.g., worms and molluscs), whose diversity is now extremely low. The indentification of certain species in zones near the shore and their complete absence in deeper waters are signs of distressed systems which worry managers. Such an impoverished biological situation is indicative of considerable changes in the physical and chemical quality of the system. The lakes' reactions to various modifications made in the catchment areas over the last few decades have led to a decline in the physical and chemical quality of the water which has in turn ended up by impacting on the flora and fauna. Signs of chemical release and close interaction between water and sediment have been clearly demonstrated. This situation of purer water and the release of excess chemicals probably explains recent changes in the quality of the lakes, but has caused confusion among managers and decision-makers. (phrase originale pas claire)

The lakes' geography and geology

The district has a total of about ten lakes spread over an area of 20 km^2 (Figure N° 1), at altitudes ranging from 650m to 850m. It is located between the towns of Clairvaux to the west, Champagnole to the north and Saint-Laurent-en-Grandveaux to the south, and is bordered by national highways A39 and Y, which are the main thoroughfares linking the region's major metropolitan areas: Geneva to the South-East, and Dijon to the North and Bourg-en-Bresse to the south-West. Its geographical location makes the district a much-sought-after tourist site in the summer season. Accomodation facilities (mainly camping grounds) have developed in most of the villages near the lakes, resulting in population peaks of over 800 % in summer (e.g., Le Frasnois has 127 off-season inhabitants compared to 3,000 in summer).

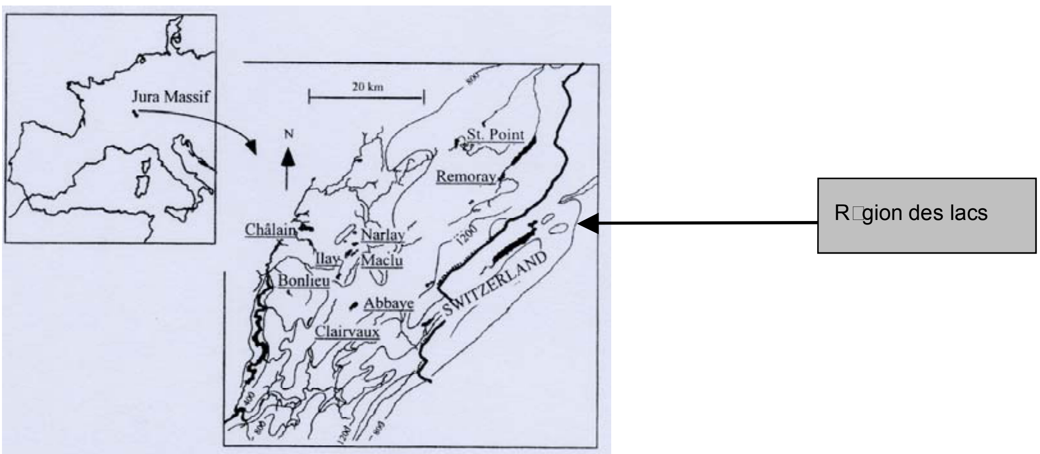


Figure 1. French Jura lake localisation.

The area's economy is agricultural, based mainly on animal husbandry and timber. The primary sector's contribution has declined considerably over the last ten years in favour of mass "green" tourism. The drop in the number of farms has been beneficial for the lakes because of the reduction of the amount of pesticides, phosphates and nitrates, but has also created problems because land is no longer properly looked after and there is consequent overproduction of organic matter.

Geologically, the district around the lakes is located in the centre of the Siam faultline bundle, which is predominantly carbonate (limestone and calcareous marl). It runs SW-NE in the lake area, is highly rumpled in parts and is bordered by plurikilometric features. The Jurassic formations are overlaid by chalky Triassic ones, which are in turn covered by formations from the Tertiary era (Oligocene and Miocene), consisting primarily of sandstone.

The structural basins resulting from tectonic movements and glaciation provided the conditions necessary for the formation of lakes whose water was blocked up by moraine belts (Campy, 1974).

The lakes' morphology

The lakes of Bonlieu, Ilay, Grand Maclu, Petit Maclu and Le Val are all long and narrow, and run in two main directions, NS and EW. They are located in basins, and preserve traces of the last glacial era. Each is surrounded by deep moraine landforms upstream and downstream functioning as large natural dams.

Bathymetric charts show the lakes are relatively shallow with depths of up to 31m for Lake Ilay and 8-9 meters for Petit Maclu.

The Jurassic and Cretaceous limestone formations are highly fractured and karstified; they allow underground watercourses to flow between lakes which are at considerable distances from each other. This form of water supply is reinforced in places by direct surface contributions flowing down from surrounding hillslopes, as well as by springs both on the slopes and underneath the lakebeds. The delineation of the lakes' catchment areas has made it possible to draw up the following table:

Table 1. The main morphological characteristics of the Hérisson lakes.

Lake Parameters	Long.	Larg.	Lake Surface	Surface BV	Max. Depth	Av. Depth	Slac/Sbv	Ind. Creux
Ilay	1900	400	72	5.5	31	11	0.001	3.77
Petit Maclu	500	120	4.5	1.085	11	7.6	0.0004	5.19
Grand Maclu	1120	300	21	1.79	24	12.7	0.001	5.24
Bonlieu	725	300	17.4	1.826	15	11	0.0009	3.12
Val	1375	425	51	n.c*	25	15.4	n.c	3.5

The lake district's climate and the lakes' hydrological processes

Climate

Data from the DIREN, plus measurements by Le Frasnois weather station and from our own weather station set up in 2002, show that the lake district has a highly contrasting oceanic climate with cold, dry winters and hot summers. Mean temperatures are around 10°C with a mean maximum of about 25-30°C. The minimum ranges from -25°C to -15°C. The variability of temperatures means that some of the Jura lakes (Petit Maclu and Grand Maclu) freeze in winter for 10 to 30 days p.a., and the other lakes from time to time. The ice varies from 10 to 40 centimetres in thickness, depending on the winter and factors such as the steepness of the surrounding slopes, the amount of shade, etc. Water supply to the lakes is cut off during this period, considerably reducing oxygen content.

Rainfall is abundant, reaching 2000 mm with close to two thirds in liquid form. The months of winter and early spring have the most snow (1 to 2 m of accumulated snow).

The prevailing winds, as recorded by the Lons-le-Saulnier station about 30 kilometers from the lake district, are SW-NE with mean velocities of about 10-18 kph.

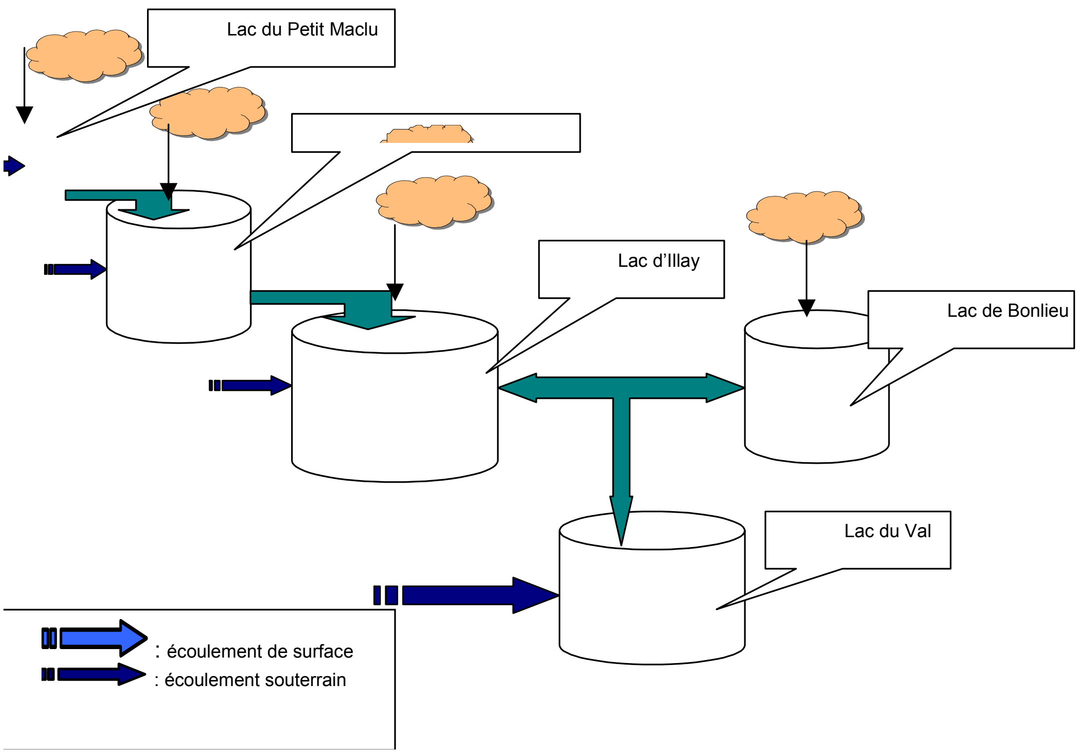


Figure 2. The hydrological organization of the Herisson watershed lake area

The lakes react almost immediately to each hydrological incident; surface levels vary between a minimum of a few dozen centimetres to 1.20 meters over the whole lake system. The difference is due to the fact that most of the basin hillslopes are relatively low and to the nature of the rocks. Most of the slopes consist of limestone landforms and the degree of fracturing facilitates water infiltration flowing towards the lakes.

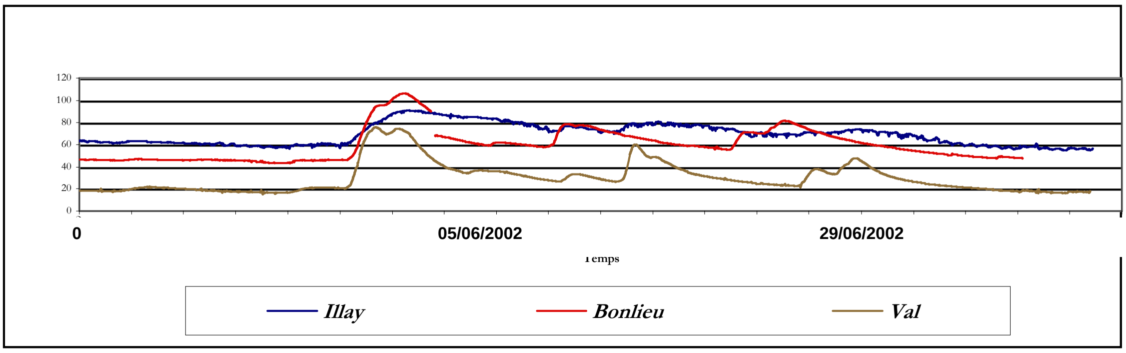


Figure 5. Electric conductivity of the studied lakes.

The system is characterized by complex, highly-diversified hydrological functions (Figure 2). The large number of caves indicates circulatory diversity, while the interconnections between surface and underground flow makes it difficult to assess patterns at lake and catchment-area scales. The levels of the biggest lakes fluctuate in similar ways (Figure N° 3). Lake Le Val receives all the water from the higher lakes, the surface water from its own catchment area and part of the underground water from the karstic system. Numerous springs on the slopes and under the lakes show there is considerable underground hydrological activity.

Physico-chemical measurements and the state of the lakes

The major physico-chemical characteristics (T°C, pH, conductivity, eh) were monitored on a monthly basis from 2000 onwards to determine their evolution and each system's reaction to the changes occurring within its catchment area.

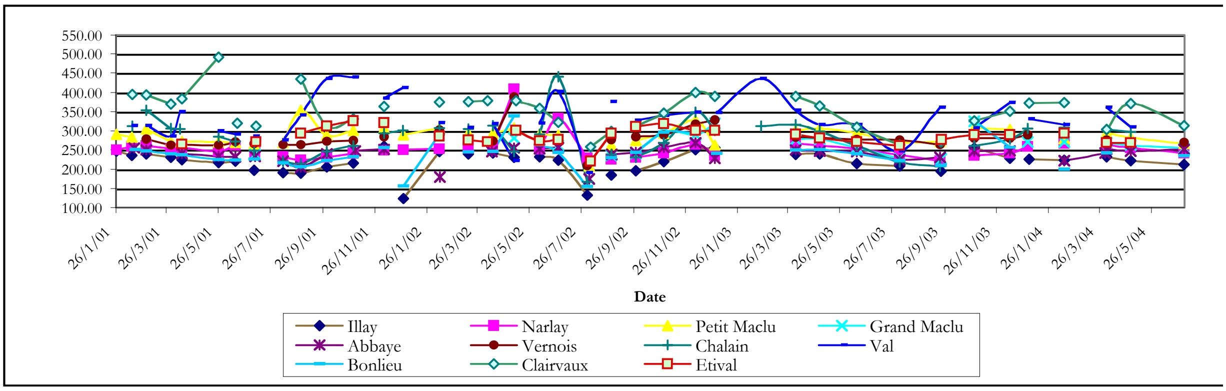


Figure 4. Evolution of the electrical conductivity of the Jura lakes' water.

As can be observed on the graphs, the Hérisson lakes go through three distinct thermal phases. Thermal inversion in winter, the mixing of all the water in spring (homthermy), and relatively clear thermal stratification from late spring on, lasting until early autumn. The lakes come into the category of warm, dimictic lakes (T°C of the bottom >4°C). In summer, the mesothermal layer thickens and sinks, hermetically separating the lake in two. The water's conductivity varies between a minimum of 130 µS/cm for the surface of Lake Ilay and a maximum of 500 µS/cm for Lake Clairvaux (Figure N°4). It is during the summer the extreme values were recorded, marking the salting-out phases clearly.

Oxygen levels are also highly variable, with clear stratification in summer During this period the oxycline temporarily settles at greater depths in the lakes. Oxygen rates are relatively low in deep water in Lakes Ilay and Bonlieu, reaching 2 mg/l in the deepest layer in summer. This tends to confirm the de-oxygenation of the lakes, especially near the lakebeds. The calculation of the $eh_{20\%}$ coefficient of the five lakes of the Hérisson basin shows that de-oxygenation is gaining ground, with the lakes becoming oxygen-starved at upper levels.

The pH is relatively low in most of the lakes, going from a maximum of 8.4 at the surface to a minimum of 6 at the water-sediment interface; this indicates that an increasingly oxygen-depleted environment is developing.

The chemical characteristics of the water and sediments of the Hérisson watershed lakes

The water of the lakes has a calcium bicarbonate facies typical of limestone and dolomitic limestone environments. The average calcium content is around 68 mg/l, with a minimum of 60 mg/l in Lake

Grand Maclu and a maximum of 74 mg/l in Lake Le Val. The water is slightly magnesian with rates of between 1.2 and 3 mg/l.

Analysis of the main trace elements shows fairly high levels of lead in lower water levels near the water-sediment interface. This could be explained by the sediments releasing lead during purging phases. Ten-centimetre-deep layers of sediment were subjected to comparative analysis. The sediment carrot was sampled and 10 one-cm-thick samples analysed for heavy-metal content. The analysis gave the following results:

Lead concentrations are low at the surface (40 µg/l for Ilay), showing a steady return to normal, and rather high at a depth of 6 cm (with a peak of 160 µg/l) (Figure N°5).

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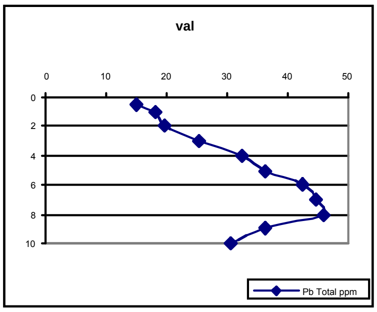


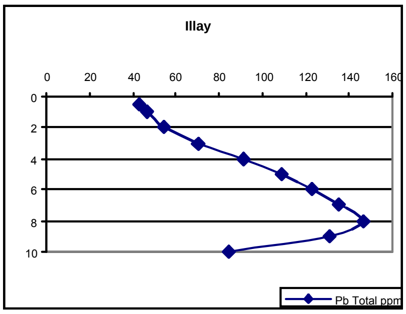


Figure 5. Content of lead in the Ilay and Val lakes.

Isotopic analysis of the lead in Lakes Le Val and Ilay gave $^{206}Pb/^{207}Pb$ ratios of 1.16 and 1.14 respectively, and 2.085 and 2.11 for $^{208}Pb/^{206}Pb$ ratios. The same ratios found in French mountain streams vary between 1.206 > $^{206}Pb/^{207}Pb$ > 1.197, and 2.066> $^{208}Pb/^{206}Pb$ > 2.053. For the Swiss Jura peatlands, Shotyk *et al.* (2000) gives the following values: 1.206 > $^{206}Pb/^{207}Pb$ > 1.194 and 2.064> $^{208}Pb/^{206}Pb$ > 2.053, which is pretty comparable. The same $^{206}Pb/^{207}Pb$ ratio for lead aerosols is around 1.162 (ElBaz-Poulichet). This reflects the fall in the lead content of gasoline. In view of our analyses and Figure N°BBB, the lead content falls within the variation interval of the gasoline lead spectrum, thus confirming its local atmospheric origin. All the other elements (Zn, Cr, Cu, Co) show practically the same tendencies, thus confirming the regional origin of the pollution.

Although the measures carried out in the catchment areas of the main lakes (e.g., the construction of water treatment facilities, reduced road salting, the elimination of the most polluting industries - “mainly pottery” –, less dung spreading and fewer farms) have shown their effectiveness in lowering the rates of certain nitrates and phosphates, they have not been sufficient to reduce other forms of pollution. Many of the suggested remedial measures seem of doubtful value to say the least, while the results of scientific research tend to confirm the alarming state of the lakes.

Conclusion

Monitoring and observation clearly show that the Le Frasnois district lakes are in trouble. Although development and protection measures carried out over the last thirty years have improved water quality with respect to certain fertilizers, quality remains below the acceptability threshold for other harmful substances such as heavy metals. These findings, extremely constraining for managers, have reopened debate on the need for a spatial approach to managing such fragile natural ecosystems. Most previous studies focused on the lakes without situating them in the wider context of their entire drainage basin. Several recent studies have shown the lakes’ functional complexity and the difficulty of determining the sources of pollution.

The interest of an approach involving several partners at different scales is that it opens up the possibility of co-operating at different management levels and thus of undertaking more effective actions. The case of the Jura lakes is an example of management difficulties where feelings of failure are prevalent among managers. The chemical analysis and monitoring of the water quality clearly show that certain essential elements are depleted and that there have been resulting impacts on flora and fauna. These disturbances are the result of the numerous local changes to the systems caused by an increasing number of farms, the opening of fragile natural areas to tourism and leisure, and ever-longer legal fishing seasons. To this first level of disturbance can be added a second, more regional one, which local stakeholders have little control over. Pollutant emissions from large metropolitan areas and industries convey increasingly large quantities of harmful substances such as lead and other heavy metals. This second category of disturbance can only be tackled by involving decision-makers at the highest levels of government.

References

Agence de l'eau, 2004. les lacs du frasnois : ilay, Narlay, Petit et Grand Maclu, Synthèse des données. Décembre 2004, pages 109.

Aries S., 2001. Mise en évidence de contaminations métalliques historiques à partir de l'étude d'enregistrements sédimentaires de lacs de haute montagne. Thèse de doctorat de l'Université de Toulouse III, pages 243.

Campy, M., 1982. Le quaternaire franc-comtois. Essai de chronologie et paléoclimatique. Thèse de Doctorat d'Etat. Faculté des sciences et des techniques de l' Université de Franche-Comté. p 575.

Fédération de pêche de Franche-comté, 2005. Etude du taux de contamination métallique du lac de Clairvaux. rapport interne. Pages 19

Roos-Barraclough, F., 2003. Millenal-Scale Records of atmospheric mercury deposition obtained from ombrothrophic and minerotrophic peatlands in the Swiss Jura mountains. Environ. Sci. Technol. 2003, N° 37, pp : 235-244.

Martin C., 1992. Les lacs de Franche-Comte (massif du jura). Thèse de Doctorat d'Etat, pages 205.

Nedjai, R., Rovéra, G, Bonnet, M. P., 2003, Les effets des épandages de sel sur la tourbière lacustre du Luitel (massif de Belledonne, France) : la dérivation des eaux du ruisseau du Rambert comme remède à la pollution. Revue de Géographie Alpine, 2003, N°1.

Shtoyk W., Weiss D., Kramers J.D., Frei R., Cheburkin A.K., Gloor M., Reese S., 2001. Geochimestry of the peat bog at Etang de la Gruère, Jura mountains, Switzerland, and its record of atmospheric Pb and lithogenic trace metals. (Sc, Ti, Y, and REE) since 12.370 14 C BP. Geochimica et Cosmochimica acta, 65, 2337-2360.

Winiarski, T., 2000. Les lacs montagnards : indicateurs de la qualité du milieu. Application aux lacs d'altitude des réserves de Haute-Savoie. Revue de Géographie Alpine, N° 3, p 10-21.

The potential role of constructed wetlands in protection and sustainable management of lake catchments in Kenya

Oketch, M. A.

Department of Environmental Science, Egerton University, P. O. Box 536, Njoro, Kenya

Email: awinooketch2002@yahoo.com

Abstract

Constructed wetlands are man-made ecosystems, which are specially designed for treatment of wastewater and non-point source pollution. Constructed wetland technology is gaining popularity due to its economically and environmentally sound attributes as a wastewater management option. Freshwater lakes in Kenya are under threat from degradation as human populations in the catchment areas increase, demands for natural resource extraction grow and urbanization and industrialization intensify. Consequently, there is increased surface runoff, causing sedimentation and eutrophication and ultimately deterioration of water quality, habitats and biodiversity. This trend requires integrated and sustainable water resource management strategies. Strategies for integrated and sustainable lake catchment management should target alternatives that are attractive to communities and other stakeholders in a lake basin. However, in developing countries like Kenya, innovative technologies like constructed wetlands have not been widely adopted for wastewater improvement. Although substantial progress has been made in the provision of services including collection, treatment and disposal of wastewater, much still remains to be done to ensure sustainable wastewater management within lake catchments. This paper presents a synthesis of the potential and challenges with respect to two case studies in Lake catchments: The Splash and Chemelil constructed wetlands. An attempt is made to explore their design, performance and limitations to their use in sustainable management of lake catchments. A number of challenges are threatening sustainable lake catchments management goals achievable through wastewater management and pollution control using constructed wetlands. Such challenges, which include lack of awareness, legislation, financial, technical and institutional resources as well as the poor understanding of constructed wetland potential, may limit the potential use of constructed wetlands for sustainable lake catchment management. There is also lack of crucial data and information about the state of constructed wetlands in Kenya. The paper presents recommendations for wider adoption of constructed wetlands use in Kenya.

Key words: Constructed Wetlands; Kenya; Lake Catchments; Protection; Sustainable Wastewater Management

Introduction

Globally, aquatic ecosystems and their associated catchments have played an important role for humankind on all continents. For instance, the African lakes have great significance to their riparian populations and the nations within which they lay for providing protein from fisheries, water for agricultural, domestic and drinking purposes, transportation and recreational use (Hecky *et al*., 2005). In addition, these lakes have also evolved remarkable endemic biodiversity with almost 10% of the earth's freshwater fish species occurring within them, making them globally significant gene banks (Ibid). On the other hand, such ecosystems have been regarded since time immemorial as cleansing media and sink for pollutants. During the past few decades, rapid population growth along the catchments areas of aquatic ecosystems in the tropics have resulted in an intensification of human activities such as industrialization, agriculture and urbanization. Consequently, there is increased deforestation, surface runoff leading to sedimentation and increased nutrient input, causing eutrophication, and ultimate deterioration of water quality, habitats and biodiversity.

Freshwater lakes in Kenya namely Lake Victoria, Nakuru and Baringo basins are threatened due to increased encroachment of human population on the catchment areas and the associated socio-economic activities. This threat is worsened by the fact that almost all the mushrooming townships, urban centers, institutions and industries in the catchments have inadequate or non-functioning waste treatment facilities. As a result, there is increased discharge of untreated or partially treated wastes into these ecosystems. In order for lake catchments to sustain their function in support of environmental integrity, riparian populations and production of ecosystems in which they lie, there is need for reduction of non-point source pollution and treatment of wastewater that drain into them. This effort will also sustain the evolving endemic biodiversity that these lake catchments host.

The multiple functions and values of wetlands in achieving these goals have been recognized worldwide (Hammer, 1989; Kadlec & Brix, 1995). The use of technologies that are economical and environmental friendly such as wastewater treatment constructed wetlands (CWs) is becoming popular and effective around the world for removal of various pollutants (Coleman *et al*., 2001; Dewardar & Bahgat, 1995; Gersberg *et al*., 1984; Rogers *et al.*, 1991; Vymazal, 1996; 2002; Zuidervaart *et al*, 1999). Constructed wetland treatment systems are engineered systems that have been designed and constructed to utilise the natural processes involving wetland vegetation, soil and their associated microbial assemblages to help in wastewater treatment. Compared with technology-based wastewater treatment systems, they require no machinery, chemicals, anthropogenic energy inputs and result in modest operation and maintenance requirements.

In the last years, several wetlands for treatment of wastewater have been constructed in East Africa, e.g. in Uganda for treating municipal wastewater (Okurut *et al*., 1999; Kyambadde *et al*., 2004) and in Tanzania for treating wastewater from the waste stabilization pond at the University of Dar es Salaam (Mashauri *et al*., 2000). These studies have indicated that CWs can be very suitable for treatment of wastewater in tropical climates (Table 1). In Kenya, the efficacy of CWs in purification of domestic wastewater (Nyakango & Van Bruggen, 2001; Nzengya & Witshitemi, 2001), industrial wastewater from pulp and paper effluents (Abira *et al*, 2003), sugar milling effluents (e.g. Opaa & Raburu, 2003, unpublished data) has been investigated. There are currently about seven operational CWs in Kenya. Their use for wastewater treatment is relatively new approach to waste disposal, which is not well exploited in Kenya.

The principal aim of this paper is to discuss the potential application and the challenges facing the use of constructed wetlands in the protection and sustainable management of Lake catchments in Kenya. The paper begins with a presentation of theoretical principles and concepts of wetlands and waste management by use of constructed wetlands. The performance in their application are highlighted. Based on lessons learnt from the case studies, the paper further presents challenges and constraints as well as recommendations for wider adoption of CW use in Kenya.

Theoretical background

Wetlands have been referred to as “kidneys of our environment by Wallace (1998), “living machines” MacDonald (1994), and “… one of nature's most effective ways of cleansing polluted water” by Rocky Mountain Institute (1998). They have been termed "kidneys of the planet" because of the natural filtration processes that occur as water passes through. As defined broadly by Ramsar Convention, they incorporate a wide variety of habitats including “areas of marsh, fen, peatland or water whether artificial or natural, permanent or temporary with water that is static or flowing, fresh or brackish or salty including areas of marine water the depth of which at low level does not exceed six metres”.

Hammer (1989) defines a constructed wetland as a designed and manmade complex of saturated substrates, emergent and submergent vegetation, animal life and water for human use and benefits. Constructed wetlands are human made, engineered areas specifically designed for the purpose of treating wastewater by establishing optimal physical, chemical and biological conditions that occur in natural wetland ecosystems. The main wastewater treatment mechanisms in treatment wetlands are sedimentation, filtration, chemical precipitation, and adsorption, microbial interaction and nutrient uptake by plants with plant harvest (Watson *et al*., 1989).

CWs are classified into three main groups; Free Water Surface (FWS), Subsurface Flow (SSF) and composite CWs, which are briefly discussed here below.

Free Water Surface (FWS) systems: These systems typically consist of basins or channels, within a natural or constructed subsurface barrier of clay or impervious geotechnical material to prevent seepage and water at relatively shallow depth flows over the soil saturated system (Figure 1).

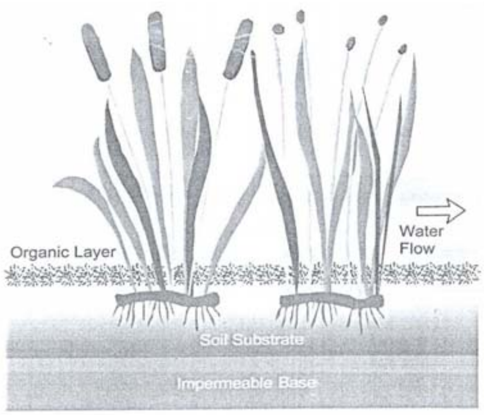


Figure 1. Free Water Surface Constructed Wetland (Source: EPA, 1988).

Subsurface Flow (SSF) systems: These typically consist of a trench or shallow basins underlain by impermeable material to prevent seepage and contains substrate medium (gravel 10-15 mm diameter) which supports the growth of emergent macrophytes such as bulrushes, cattails and reeds and permit long-term subsurface flow without clogging (Figure 2). The subsurface zone is continuously saturated and therefore generally anoxic or/and anaerobic.

Hybrid wetlands: Various types of constructed wetlands may be combined in order to achieve higher treatment effect, especially for nitrogen. In these systems, the advantages of the HSF and VF systems can be combined to complement each other.

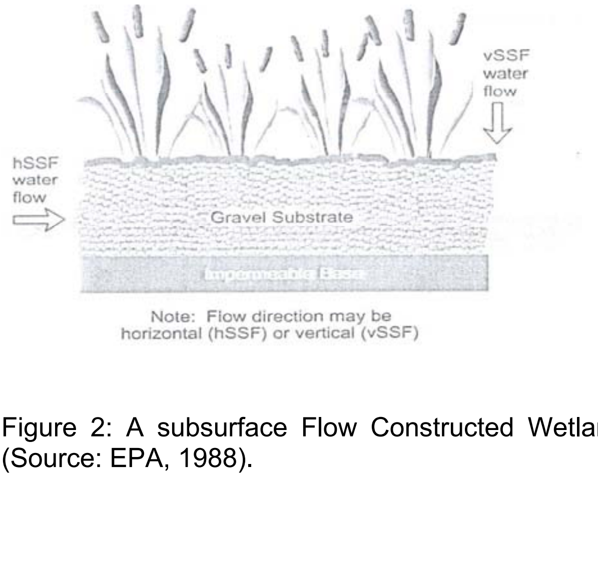


Figure 2: A subsurface Flow Constructed Wetland (Source: EPA, 1988).

Table 1: Treatment efficiencies in relative (%) of some constructed wetlands located in tropical and subtropical regions concerning a selected number of water quality parameters (HLR =hydraulic loading rate; HRT = hydraulic retention time).

CW type	*Country*	*Area (m^2)*	*HLR (m/d)*	*HRT (d)*	*Mean influent concentration (mg/L)*		*% reduction in concentration*		*Reference*
FWS	Uganda	300	-	7-12	COD	155.2	C.P	63	Okurut *et al*., (1999)
							P.M	43	
					PO_4	3.71	C.P	16	
							P.M	37	
HSSF	Tanzania	n.i	6.48	n.i	SS	104.8	80		Mashauri *et al*., (2000)
					COD	100.8	66		
			55.2	n.i	SS	101.8	50		
					COD	125.8	50		
FWS & HSSF	Tanzania	10	0.018	12.8	PO_4-P	2.39	69		Lin *et al*., (2002)
			0.034	6.8		10.45	45		
			0.135	1.7		5.19	35		
VSSF	Uganda	9.36		5	TRP	15,5	C.P	83	Kyambadde *et al*., (2004)
							M.V	48	

n.i = no information; C.P = Cyperus papyrus; P.M = Phragmites mauritianus; M.V =Miscanthidium violaceum

Constructed wetlands for wastewater treatment in Kenya

The Chemelil constructed wetland system for industrial waste treatment

A case in point is the management and purification of industrial wastewater using constructed wetlands in Chemelil sugar factory. The sugar factory is located in the vicinity of a small tributary to the Nyando River, in North-west Kenya. It produces a BOD_5 load of approximately 2000 kg per day placing it among the top polluting industries near the Winam Gulf (Calamari *et al*., 1995). In order to enhance wastewater quality, 12 oxidation ponds each measuring 100 x 35 m have been used. However, these systems were not functioning satisfactorily. As a result, a horizontal subsurface flow constructed wetland system was built downstream of the oxidation ponds in 2002.

Design

The wetland system consists of 8 wetland cells of rectangular shape (10 X 3 X1 m). The cells are of rectangular shape (10 X 3 X1 m). The beds were packed with a wetland medium (sand). Wetland cells 1, 3, 5 and 7 were planted with *Cyperus papyrus* whereas wetland cells 2, 4, 6 and 8 were planted with *Echinocloa pyramidalis* that are used for fodder and fibre in the region. The wastewater is first transported through a ditch to a sedimentation trap. From there it flows to stabilisation ponds. The wastewater enters the wetland system from pond number 9 through an inlet channel, which distributes the wastewater to the 8 constructed wetland cells.

Performance of the Chemelil constructed wetland

The effectiveness of the CWs in treating industrial effluent, at Company during a three-month period (March, 2003–May, 2003) is given in Table 2.

Table 2. Treatment efficiencies in relative (%) of Chemelil constructed wetland from March 2003 to May 2003

Wastewater parameter	*Mean influent concentration (mg/L)*	*Mean effluent concentration (mg/L)*	*% Reduction in concentration*
$BOD_5(mg^{-l})$	-	-	n.i
$COD(mg^{-l})$	166.38 ± 33.50	105.93± 32.63	36.7 %
TSS (mg^{-l})	1.10 ± 0.04	0.04 ± 0.01	96.4%
Organic nitrogen	0.997 ± 0.17	0.945± 0.22	28.6%
Phosphates	1.21 ± 0.23	0.27 ± 0.17	74%

(Source: After Opaa, B. O and Raburu, P. O, 2003) n.i = no information

Lessons learnt

This case study demonstrates that the constructed wetland decreased the loads of SS, COD and nutrients from industrial wastewater. Reductions of upto 36.7% (COD), 96.4% (TSS), 28.6% (organic nitrogen), 74% (Phosphates) were obtained from the treatment cells, which were lower than the splash wetland. The system entailed monocultures of the plant species and not polycultures. However, data collection begun two months after planting which did not give vegetation (macrophytes) and bio-film sufficient time to establish itself as well as time for development of litter and standing dead compartments.

The splash constructed wetland for domestic wastewater treatment

The Splash constructed wetland is located in Langata area, south of Nairobi, Kenya. The site lies at an average elevation of 1680m a.s. l and on latitude 1° 19'south and longitude 36° 59' East. Precipitation is bimodal, with an annual mean of 1680mm. The wetland occurs on black cotton soil, which is rich in montmorillonites and clays, thereby promoting extensive cracking during dry period and water logging during wet season. The Splash wetland was constructed in 1994 in order to purify the wastewater from the carnivore restaurant and the swimming pool resort.

Design

The constructed wetland has a surface area of 0.5ha and was designed for a 1200 Population equivalent (P.E). The wetland was constructed on a small naturally waterlogged depression. It consists of a combination of a subsurface horizontal flow system followed by three surface flow wetland cells in a series adjacent to it. It is vegetated with *Typha* sp., *Papyrus* sp, *Phragmites* and a variety of ornamental plants such as *Hydrocotyle, Hydrocleis* and *Pontederia*. Pretreatment consist of septic tanks before discharging into the gravel bed hydroponics and ultimately into the wetland cells.

Performance

The efficiency of the CW in improving wastewater quality is given in Table 3.

Table 3. Treatment efficiencies in relative (%) of splash constructed wetland from Dec 1995 to March 1996.

Wastewater parameter	***Mean influent concentration (mg/L)***	***Mean effluent concentration (mg/L)***	***% Reduction in concentration***
$BOD_5(mg^{-l})$	1603.0 ± 397.6	15.1 ± 2.5	99.1%
$COD(mg^{-l})$	3449±206.8	95.5 ± 7.2	96.9%
TSS (mg^{-l})	195.4±58.7	4.7 ± 1.9	97.6
TDS (mg^{-l})	816.5 ± 84.2	857.4 ± 62.6	-
$NH4^+$ (mg^{-l})	14.6 ±4.1	Undetected	-

(Source: After, Nzengya & Witshitemi, 2001).

Lessons learnt

The wetland system significantly improved the wastewater quality. This is attributed to the diverse physical and chemical process occurring in the different components as well as to the diversity of wetland plants in the respective wetland cells. Nyakango and Van Bruggen (1999) also reported high treatment efficiencies in terms of BOD_5 (98%), SS (85%), COD (96%), TKN (90%), NH_4 (92%) and 0-$P0_4$ (88%) in the same wetland. Due to its large size, the system also provided a large surface area for the microbial decomposition to occur. The high treatment efficiencies can also be attributed the presence of multiple plant species in the system. Multiple species within the system maximized root biomass in the wetland substrate resulting in more efficient treatment.

Challenges and constraints to be overcome for adoption of constructed wetlands for waste management

A number of challenges have to be overcome in order to achieve sustainable lake catchments management goals through wastewater management and pollution control using constructed wetlands:

1. **Lack of awareness.** Awareness on the multiple functions and values of wetlands are a major hindrance to the management and sustainable use of wetlands. In much of the country, wetland awareness is still very low.
2. **Institutional, financial and technical constraints**: The lack of positive measures such as subsidies for pollution control equipment and funding for introducing low-cost waste technologies continue to hinder the effective implementation of pollution control measures.
3. **Environmental legislation:** With regard to the present legislation on water pollution is absence of concrete regulatory measures and enforcement mechanisms. A situation that does not motivate polluters to search for cheap ways of controlling pollution.
4. **Poor understanding of constructed wetland potential**: Little effort has been made to investigate the effectiveness of CWs in treating various types of effluents despite suitable climatic conditions in Kenya. This is due to lack of financial resources to undertake extensive and comprehensive surveys and research.
5. **Lack of clear policies on public participation in waste management**: An important challenge is the existence of few policies that promote public participation and sustainable wetland management and conservation in Kenya
6. **Lack of training of wetland experts**: As a result any innovative approaches that would encourage the increased use of such technologies have been hampered. There is presently no central government agency promoting constructed wetlands as a sustainable means of managing wastes in Kenya.

Recommendations for a wider promotion and acceptability of Constructed Wetlands in Kenya

The following recommendations are suggested based on the case studies and for wider promotion and acceptability of constructed wetlands in Kenya:

1. These case studies recommend the use of polycultures (mixed plant species) of the plant species in constructed wetlands in order to yield greater removal efficiencies than monoculture systems.
2. There should be an advocacy of methods of waste disposal that are not only cheap and highly effective like CWs but also which promote resource conservation and environmental protection.
3. Education and heightened awareness promotion programmes concerning both natural and constructed wetlands should be implemented at all levels.
4. Increased financial support for research and training of wetland specialists aimed at understanding the complex processes occurring in existing CW in Kenya.
5. Government and planners have to develop motivating mechanisms that include environmental concerns - it is better to use treated wastewater for economic purposes rather, than directly discharging it to waterways and decreasing the waste assimilative capacity of the water courses, economic concerns - reuse of wastewater for aquaculture and irrigation can help reduce the pressure for public investment in large (and costly) water resources development projects; and legal concerns - regulatory and economic instruments can provide direct incentives to polluters to use treated wastewater for aquaculture and farm purposes.
6. Government regulations and legislation need to be enforced in order to ensure that polluters meet environmental standards of waste discharge into water systems.
7. Constructed wetlands may and should be integrated into management plans for conservation of both soil and water resources of a watershed, or integrated as parts of major restoration projects.

Conclusions

Constructed wetlands have a potential to play in wastewater management and non-point source pollution control in Lake catchments in Kenya. However their design, performance and potential for wastewater reuse is critical to the overall realization of this potential. The case of splash wetland reveals a thrilling prospect in use of polyculture systems in wastewater treatment. Larger and well-established systems have higher treatment abilities than new systems. In both case studies, effluent at the final discharge (outlets) met admissible standards set by various bodies such as WHO and European community, for discharge into surface water masses. As a result, the water can be re-used for various purposes. Their use can thus help alleviate the problem of discharging untreated or partially treated wastewater into aquatic systems. This makes them part of the sustainable development approach to waste management.

The great interest displayed by government organizations and other polluting agencies should be encouraged as they have been proven to improve significantly the quality of water effluents for some industries. If Kenya is to meet wastewater treatment requirements of the future and achieve the Millennium Development Goal of halving the number of people with out clean safe water and basic sanitation by 2025, then integrated and sustainable water resource management through treatment systems that are not only effective and reliable, but also simple and inexpensive to construct, operate and maintain should be promoted. The management and conservation of wetlands for their long-term sustainable use should be a priority in Kenya's conservation efforts.

References

Abira M.A, Ngirigacha, H. W & Van Bruggen, 2003. Preliminary investigation of the potential of four emergent macrophytes for treatment of pre-treated pulp and paper mill wastewater in Kenya. *J. Wat. Sci. Tech. 48 (5), 223-231.*

Calamari, D., Akech, M. O. & Ochumba, P. B. O., 1995. Pollution of Winam Gulf, Lake Victoria, Kenya: a case study for preliminary assessment. *J. Lakes & Reservoirs: Research and Management 1, 89-106*.

Coleman,J.,K.Hench,K.Garbutt,A.Sexstone,G.Bissonnette,J.,andSkousen,J., 2001. Treatment of domestic wastewater by three plant species in constructed wetlands. *J. Water Air Soil Pollut.128, 283 –295*.

Dewardar, A & Bahgat, M., 1995. Fate of faecal coliform bacteria in wastewater reservoir containing *Lemna gibba L*. *J. Wat. Res. 29, 2598-2600*.

Gersberg, R. M., Brenner, R., Lyon, S. R., & Elkins, B. V., 1984. Survival of bacteria and viruses in municipal wastewaters applied to artificial wetlands. *In.* Reddy, R. R. and Smith, W. H. eds. *Aquatic Plants for Water Treatment and Resource Recovery*. Magnolia Publishing pp. 237-245.

Hammer, D. A., 1989. *Constructed wetlands for wastewater treatmen*t. Lewis Publishers, Inc. Michigan.

Hecky, R.E, Bootsma, H.A & Odada, E., 2005. African Lake Management Initiatives: The Global Connection Paper presented at WORLD BANK/GEF Atmospheric deposition Workshop, 23-25 Nairobi, Kenya May 2005.

Kadlec, R. H & Brix, H., 1995. Wetlands systems in water pollution control. *J. Wat. Sci. Tech., 32: (3).*

Kadlec, R. H. & Knight, R. L., 1996. *Treatment wetlands*. CRC Press Inc., Boca Raton, Florida.

Kyambadde, J., Kansiime, F., Gumaelius, L. & Dalhammar, G., 2004. A comparative study of *Cyperus papyrus* and *Miscanthidium violaceum*-based constructed wetlands for wastewater treatment in a tropical country. *J. Wat. Res.* 38,475-485.

MacDonald, L. , 1994. Pollution Solution, *American Forests.* Vol.100, No.7/8, pp 26-30.

Maehlum, T., Warner, W. S., Stalnacke, P. & Jensen, P. D., 1999. Leachate treatment in extended aeration lagoons and constructed wetlands in Norway. *In.* Mulamoottil, G., McBean, E. A. and Rovers, F. eds. *Constructed wetlands for the Treatment of Landfill Leachate,*. CRC Press/Lewis Publishers, Boca Raton, Florida.

Mashauri, D. A., Mulungu, D. M. M. & Abdulhussein, B. S., 2000. Constructed wetland at the University of Dar es Salaam. *J. Water Research 34 (4), 1135-1144*.

Nzengy'a, D. M & Wishitemi, B. E. L., 2001. The performance of constructed wetlands for wastewater treatment: a case study of splash wetland in Nairobi, Kenya. *Hydrol. Process. 15, 3239-3247.*

Nyakango, J. B & J. J. A. Van Bruggen, 1999. Combination of a well Functioning Constructed wetland with a pleasing Landscape Design in Nairobi, Kenya. *J. Wat. Sci.Tech. 40 (3), 247-249.*

Okurut, T. O., Rijs, G. B. J., & van Bruggen, J. J. A., 1999. Design and performance of experimental constructed wetlands in Uganda, planted with *Cyperus papyrus* and *Phragmites mauritianus*. *J. Wat. Sci.Tech. 40 (3), 265-271*.

Rocky Mountain Institute,1998. *Green Development: Integrating Ecology and Real Estate*. John Wiley and Sons, Inc. Toronto, Ontario. pp 146-150.

Rogers, K. H; Breen, P. F. & Chick, A. J., 1991. Nitrogen removal in experimental wetland treatment systems. *Evidence for the role of aquatic plants.* WPCF, 63 (7), 934-941.

Vymazal, J., 1996. The use of subsurface flow constructed horizontal systems for wastewater treatment in the Czech Republic. *J. Ecological Engineering 7, 1-14.*

Vymazal, J., 2002. The use of subsurface flow constructed wetlands for wastewater treatment in the Czech Republic: 10 years of experience. J. *Ecological Engineering 18, 633-646.*

Wallace, S., 1998. "Putting wetlands to work" *J. Civil Engineering,* 98-007-0057 American Society f Civil Engineers, New York.

Watson, J.J, Reed, S.C, Kadlec, R.H, Knight, R.C & Whitehouse, A. E., 1989. Performance expectations and Loading Rates for constructed wetlands. *In.* Hammer, D. A. ed. *Constructed wetlands for wastewater Treatment*. Lewis publishers, Chelsea,

Zuidervaart, I., Tichy, R., Kvet, J & Hezina, F. 1999. Distribution of toxic metals in constructed wetlands treating municipal wastewater in the Czech Republic. *In.* Vymazal, J., ed. *Nutrient Cycling and Retention In Natural and Constructed Wetlands*. Backhuys Publishers, Leiden, the Netherlands.

Water balance for Lake Victoria

Okonga J. R. [1], S. M. Sewagudde[1] R. J. Mngodo[2,] F. D. Sangale[3], F. L. Mwanuzi[4] and R. E. Hecky[5]

[1] Water Resources Management Department, Directorate of Water Development, P.O. Box 19 Entebbe, Uganda.
[2]Water Resources Division, Ministry of Water and Livestock Development, P.O. Box 35066 Dar es Salaam, Tanzania.
[3]Ministry of Water and Irrigation, P. O. Box 1922, Kisumu, Kenya
[4]University of Dar es Salaam, Water Resources, P.O. Box 35131 Dar es Salaam, Tanzania
[5]University of Waterloo, Biology Department, 200 Waterloo, Ontario N2L3GI, Canada

Abstract

One of the principal objectives of the Water Quality and Quantity Components is to find the reasons for the changes observed in the lake water quality, quantity and ecosystem, and to identify remedial measures. To identify the reasons for the changes one requires a knowledge of the changes in the pollution loadings to the lake, which, in turn, depends on the discharges into the lake from the catchments and the atmosphere and lake outflow to River Nile i.e. hydrology and meteorological characteristics in and around the lake.

Hydro-metrological data for the period running 1950-2004 was analysed which form the basis for computing the pollution loadings (catchment and atmospheric) into the lake and lake water balance. Continuous rainfall and evaporation records were generated. Full records of land discharges were obtained through modelling using the NAM model. Model performance was evaluated on the ability to simulate the total flow rather than the peak and minimum flows for pollution estimation.

Results indicate that Tanzania's unshared land catchment annual discharge contribution to Lake Victoria is approximately 5.430 BCM (Billion Cubic Metres), while Uganda's one is approximately 1.062 BCM and Kenya's one is approximately 9.271 BCM which in percentage are 21.4%, 4.2% and 37% respectively. Rivers Mara and Kagera that are shared between Kenya and Tanzania and Tanzania, Rwanda, Burundi and Uganda respectively had total flows of 1.151 BCM and 8.215 BCM accordingly representing 4.6% and 32.7% of the total catchment discharges. The mean annual rainfall over the Ugandan side of the lake is about 62,539 BCM, Tanzania is about 60.682 BCM and Kenya is about 4.541 BCM. These forms 48.9%, 47.5%, and 3.6% respectively of the total mean annual lake rainfall into Lake Victoria. There was a 10.7% decrease in rainfall over the Lake in the period 1972-1993. However in the period 1994-2004 there was a 2.2% increase in the amount However there was a 14.7% decrease in catchment inflows into the lake and a 1.64m drop in water level in the period 1998-2004.

Introduction

There are concerns that the ecosystem and water quality of Lake Victoria has changed and more changes have been observed in recent years. The knowledge of the water budget changes is essential in the estimation of pollution loading into the lake in order to identify the root cause of this deterioration. But pollution depends on the discharges to the lake from the catchment and the atmosphere. An estimate of the total water balance for the lake over the period 1950-2004 is considered in order to establish the water budget of Lake Victoria basin, i.e. all discharges to and from the lake on a daily basis. The estimates are required for (i) rainfall onto, and evaporation from the lake surface. (ii) discharges into the lake from all rivers and catchments around it, and (iii) flow from the lake into the River Victoria Nile.

The objectives of this paper are therefore to:

- determine the water balance of Lake Victoria over the above period, and
- attempt to explain the current drop in levels of Lake Victoria.

Hydrology of Lake Victoria

The key elements that explain the hydrology of Lake Victoria are: rainfall, evaporation, discharges from surrounding catchments and outflow through the Nile. This paper looks at the relative quantitative regimes of these processes.

Estimation of lake rainfall

In the estimation of rainfall over the lake, the lake area was divided into polygons (boxes) representing rainfall influences from the stations selected. In this method, each rain polygon (box) had a reference rainfall station. Table 3.2 shows the rainfall boxes for the Lake Victoria. The mean annual rainfall in each box was computed using the rainfall isohyetal curves derived by drawing curves that link stations with similar average rainfall totals. Table 3.1 summarises the contribution of Lake Rainfall to the storage of Lake Victoria : a total of about 127.762 BCM with the relative country contributions indicated in Table 1.

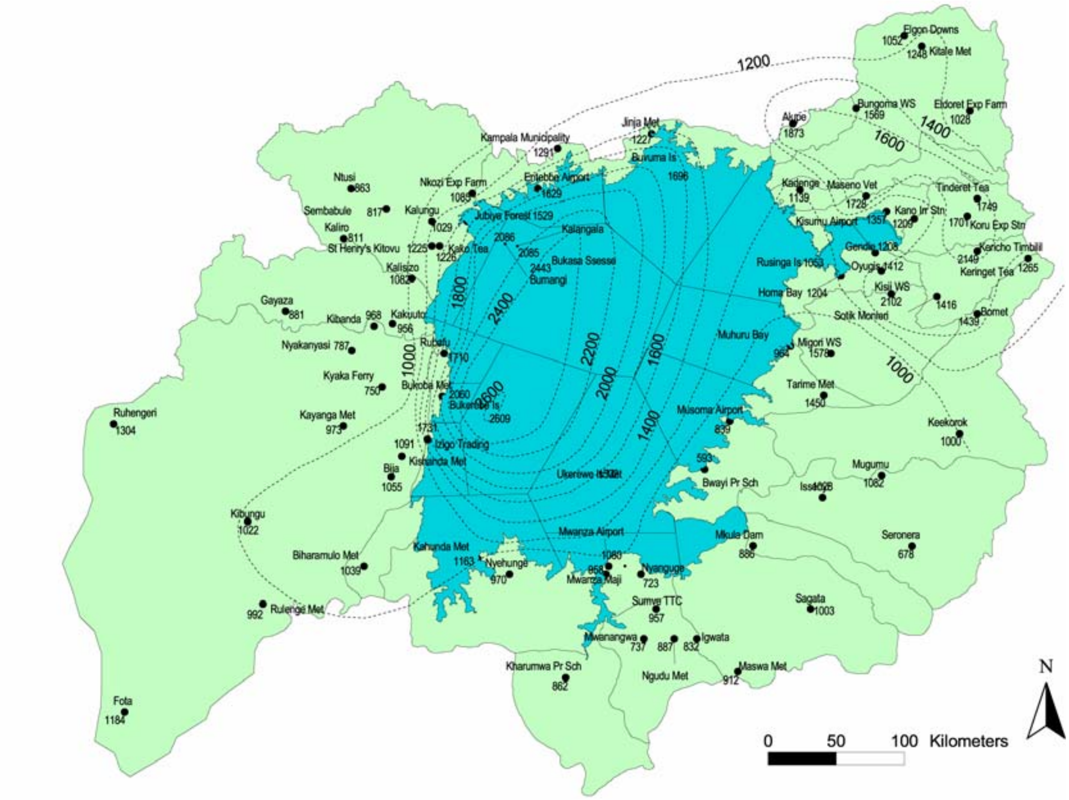


Figure 1. Spatial Rainfall over Lake Victoria.

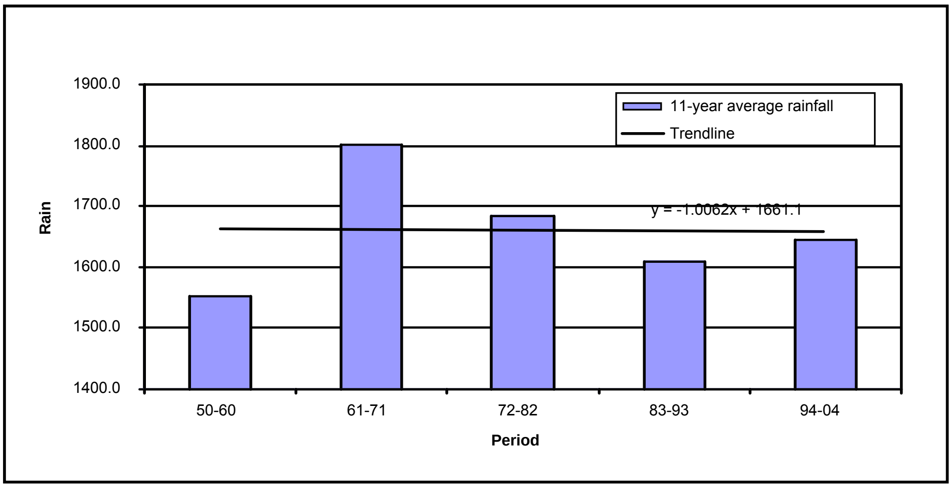


Figure 2. long term rainfall trend over Lake Victoria.

Table 1. Rainfall input into the lake by country.

Country	Rainfall M	% Area	Lake area km^2	Rainfall BCM	%
Uganda	2.02	45	30,960	62,539	48.9
Tanzania	1.8	49	33,712	60,682	47.5
Kenya	1.1	6	4,128	4,541	3.6
		100	**68,800**	**127,762**	**100.0**

Rainfall regimes over Lake Victoria

The convergence of southeasterly and southwesterly winds in this period account for the heavy rainfall amounts in the western and northern shores of the lake. This influence extends in Uganda from the Ssesse Islands to Katonga Catchment leaving Bukora Catchment a bit arid. The highest rainfall in Uganda is received around the Ssesse

Islands. This reaches totals of about 2,400 mm annually, while in Tanzania Bukerebe receives an average of 2447mm while Bukoba has 2020mm of rainfall. In Kisumu the average maximum rainfall is around 1358mm. In the Northern Shores, the lacustrine effects do not extend for more than 40 km in most places where arid climate typical of the cattle corridor sets in.

Figure 1 shows annual rainfall data represented in five blocks of 11-year average annual rainfall total with the aim of determining whether there is any temporal change in rainfall pattern. Results show that there was a 10.7% decrease in rainfall amounts over Lake Victoria in the period 1972-1993. However in the period 1994-2004 there was a 2.2% increase in the amount.

In general there is an increment in annual total rainfall by 0.9% for the LVEMP period 2001-2004 compared to the previous period of 1950-2000, this make the average annual rainfall for the long-term period 1950-2004 to be increased by 0.065%. This drop is probably attributed to the changes in seasonal rainfall pattern. When considering three seasons in a year, February to May (FMAM) accounts for 48.6% of annual rainfall for the LVEMP period (2001-2004) while the period June to September (JJAS) accounts for only 16.7% of annual rainfall and the period October to January (ONDJ) accounts for 35% of annual rainfall. There is an increase of 5% seasonal rainfall for FMAM, decrease of 13.8% rainfall for JJAS and an increase of 2.2% in ONDJ.

Estimation of lake evaporation

A number of stations within the basin have been selected for estimation of evaporation. Data gaps were identified and gap filling was necessary to create continuous record data sets that could be used to derive statistical and analytical values.

A similar approach as for rainfall was used for quality control and gap filling of evaporation time series. Firstly, the use of correlation is limited by the fact that evaporation is mainly measured at synoptic meteorological stations, which are few, and as a result the stations are scattered far apart. Secondly, there is no well-defined method for assertion of an average, dry or wet year for evaporation.

Pan Evaporation data was used in all the three countries for the pre-LVEMP period. Whereas for major part of LVEMP period, Uganda used data from Automatic Weather Stations (AWS), the other two countries (Kenya and Tanzania) continued to use pan evaporation data except in a few cases where the AWS are in operation.

Estimation of evaporation over the lake

A similar approach as that used for estimating lake rainfall was applied in the estimation of the lake evaporation. To calculate evaporation over the lake therefore, boxes similar to those used for estimation of total rainfall over the lake are used. Each box has a reference evaporation station where mean evaporation is estimated on the basis of isohyets curves respectively. Each box also has a weight attached to it according to the size of the lake area it represents.

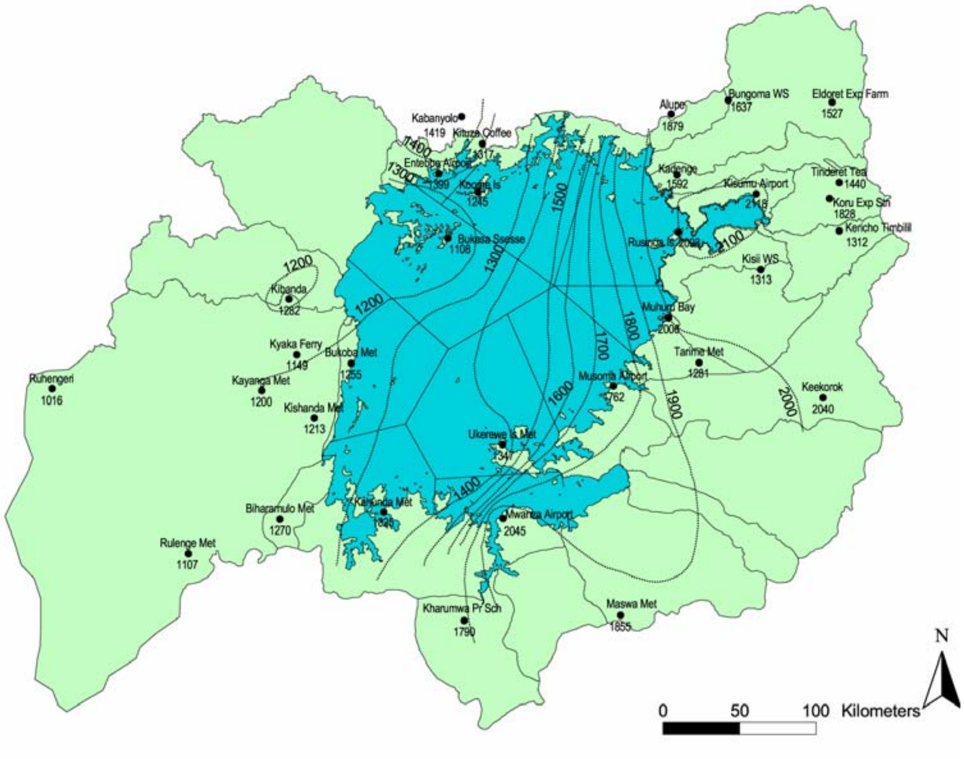


Figure 3. Spatial evaporation over Lake Victoria.

Comparatively, the highest Lake evaporation (figure 3.12) is recorded on the eastern and northeastern shores of the lake (i.e. Mwanza, Musoma, Muhuru, Rusinga and Kisumu), while the islands, western and southwestern shores have the lowest evaporation rates (i.e. Ukerewe, Kahunda, Bukoba, Bukasa, Entebbe and Koome).

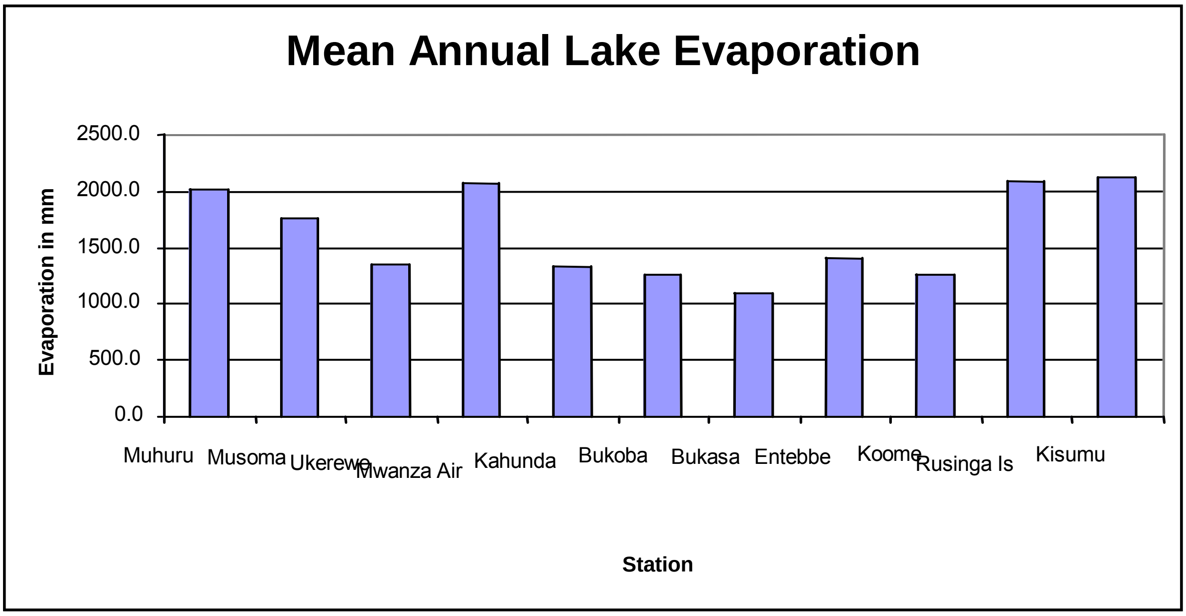


Figure 4. Mean annual lake evaporation.

The relative pattern of evaporation by country portion of the lake is illustrated below:

Tanzania

During the LVEMP period of 2001-2004, the average annual evaporation for Musoma Met station increased slightly by 1% and this value was further smoothened in the combined long-term period of 1950-2004 to a mere 0.1%. As for Mwanza Airport evaporation decreased by 8.3% for the LVEMP period but overall the increment was reduced to 0.6% for the period 1950-2004. Although there was an increase of 13.3% in evaporation for Bukoba Airport, the long-term period of 1950-2004 shows a decrease in evaporation by only 1% due to smoothening effect.

Uganda

Computation of lake evaporation suggests that evaporation tendencies are relatively homogeneous compared to rainfall. The deviation from the mean annual evaporation boxes is 155mm as compared to 270mm for the mean annual rainfall boxes. For that reason the effect of the box size influences the total lake evaporation to a small extent. Results show that evaporation from the Uganda part is less than rainfall by a factor of 0.66 and accounts for 29.9% of the lake evaporation. Since the estimated mean annual evaporation is far less than the mean annual rainfall and considering that evaporation and, it can be deduced that the Ugandan portion plays an important part in determining the positive tendency in the net basin supply for the lake.

Kenya

Kenya had to rely on long-term daily average evaporation figures, as much of the required observed data for the reference stations was not available. According to the data gathered, average annual evaporation from the catchment is 1751.4 mm compared to that from lake surface, which is 2072.1 mm.

In the catchment, highest evaporation rates are recorded in the lower basin stations (i.e.Kano, Kisumu, Muhuru and Rusinga), medium evaporation in the middle zone (i.e. Alupe, Chemelil, Bungoma, Eldoret) and lowest in the highland areas (i.e. Kericho and Kisii)

Flows into Lake Victoria

Time series of aggregate catchment discharges

Because there is an apparent seasonal spatial homogeneity in the data sets depending on the runoff-generating pattern, catchments were grouped into 5 zones (Table 2).

Table 2.

(A) Northeastern Zone	(B) Southeastern zone	(C) South western shores	(D) North western shores	(E) Kagera
Sio Yala North Awach South Awach Nyando Sondu Nzoia and Gucha-Migori	Mara Eastern Shores Grumeti Mbalageti Simiyu Magogo Isanga	Southern shores Biharamulo Western shores	Bukora Katonga and Northern shores.	

Monthly flow data from each zone were summed up and averaged over the years to get mean aggregate flows (figure 3.12), which was subjected to 6 month moving average in order to remove noise. The aim of this approach is to examine the average flow trends over the years in order to detect any significant variations in the time series.

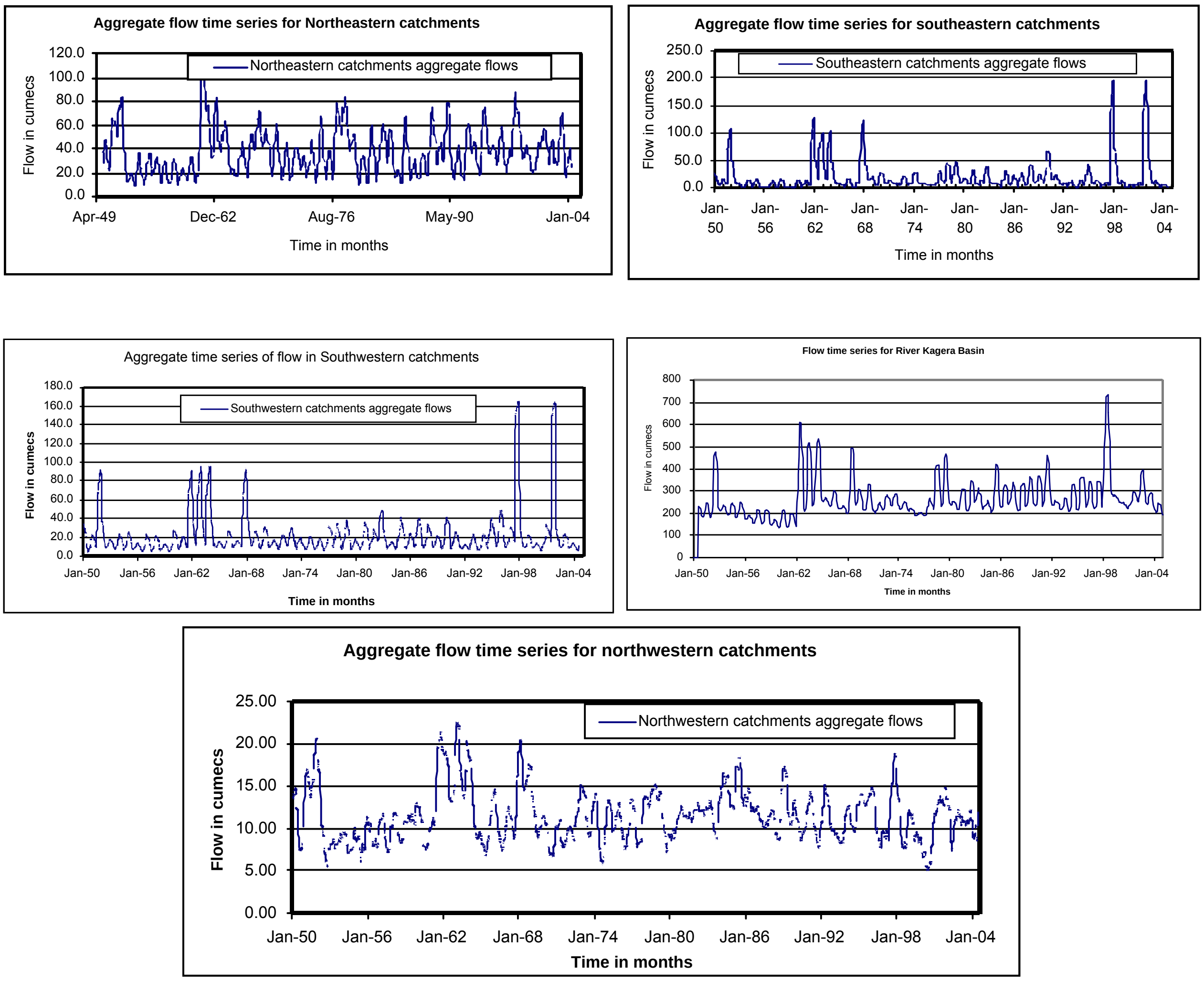


Figure 5. Mean aggregate flows.

The catchment time series of the flow, show striking revelations about some periodic events. Not only does the time series exhibit nearly all the peak annual events associated with the *El Niño* event years, but also does correspond to periods of low flows associated with below normal rainfall performance. The *El Niño* rains of the 1961-62 and the 1998-99 are well replicated in zonal charts (figure 3.12). All the time series have shown declines in catchment discharges in the last 5 years from 2000. The Kagera basin yield has too, declined from the 33.5% of all catchment discharges to only 30.7% in the last 5 years giving a total decline of 8.4 % of relative catchment contribution. Generally it is observed that there is a slight increase in catchment yields in the north eastern catchments.

Mean flows from individual catchments into Lake Victoria

An examination of the mean flows was made by splitting the data sets for catchment inflows into 3 periods as follows: 1950–2000 period during which an Integrated Water Quality and Limnology Study of Lake Victoria was done, 2001–2004 period data collection was done under LVEMP; and 1950–2004 period showing long term averages. Results show that on average, there was a significant decline in catchment inflows into Lake Victoria of 14.7% (Table 3.) for the LVEMP period relative to the 1950-2000 period or record. Table 3.2 shows a summary of the mean flows in the above categories including the proportion of total basin inflow from each river for each time period

Table 3. Mean flows from catchments around the lake.

Country	**Drainage Basin**	**LVEMP Study (1950-2000)**		**LVEMP (2001-2004)**		**Long term 1950-2004**	
		Flow in Cumecs	%	Flow in Cumecs	%	Flow in Cumecs	%
Kenya	Sio	11.4	1.4	9.8	1.4	11.3	1.4
	Nzoia	116.7	14.5	107.4	15.7	116.1	14.6
	Yala	37.7	4.7	47.9	7.0	38.4	4.8
	Nyando	18.5	2.3	41.9	6.1	20.3	2.6
	North Awach	3.8	0.5	3.3	0.5	3.7	0.5
	South Awach	5.9	0.7	5.5	0.8	5.9	0.7
	Sondu	42.2	5.2	43.9	6.4	42.4	5.3
	Gucha-Migori	58.0	7.2	39.9	5.8	56.6	7.1
Kenya and Tanzania	Mara	37.5	4.7	23.1	3.4	36.5	4.6
Tanzania	Grumeti	11.5	1.4	4.6	0.7	11.0	1.4
	Mbalageti	4.3	0.5	3.5	0.5	4.2	0.5
	E. Shore Streams	18.6	2.3	11.3	1.6	18.1	2.3
	Simiyu	39.0	4.8	12.2	1.8	37.0	4.6
	Magogo-Maome	8.4	1.0	1.6	0.2	7.8	1.0
	Nyashishi	1.6	0.2	0.3	0.0	1.5	0.2
	Issanga	31.0	3.9	4.3	0.6	29.0	3.6
	S. Shore Streams	25.7	3.2	3.5	0.5	24.1	3.0
	Biharamulo	17.8	2.2	18.3	2.7	17.9	2.2
	W. Shore Streams	20.7	2.6	18.9	2.7	20.6	2.6
Burundi, Rwanda, Tanzania & Uganda	Kagera	261.1	32.4	252.5	36.8	260.5	32.7
Uganda	Bukora	3.1	0.4	2.0	0.3	3.0	0.4
	Katonga	5.1	0.6	2.1	0.3	4.9	0.6
	N. Shore Streams	25.6	3.2	28.2	4.1	25.8	3.2
	Total	**805.3**	**100**	**686.2**	**100**	**796.6**	**100**

Results show that on average, there is a significant decline in catchment inflows into Lake Victoria to the tune of 14.8% for the 2001-2004 period compared to the long term mean period of 1950-2000.

River Nile outflow

During the period 1950-1954, River Nile outflow was naturally occurring until the commissioning of the Owen Falls Dam in 1954. The Dam was built to operate on the “Agreed Curve” Policy that determines the amount of water to be released by using the prevailing water levels in order to maintain natural flow. The operationalisation of this policy maintained a natural pattern up to 2000. During the period 2001-2004, disparities began to occur between lake levels and Nile outflow. The hydrograph in figure 6. for the period 2001-2004, show that Nile outflows have increased while lake levels have fallen. This can partly be attributed to increasing outflow at Jinja and other climatic factors, e.g. periods of lower rainfall than has occurred over the historic period.

The summary of flow characteristics for R. Nile outflow in Table 3.3 indicate an increase in average flow out of the lake by 15% to 1057.6 Cumecs in the period 2001-2004 as compared with the long term average of 1046 Cumecs in the period 1950-2000 including the per cent of all losses with the remaining loss being evaporation. Although the statistics has fewer samples than the long term, period, it nevertheless gives a general pointer to the new hydrologic trend that may emerge.

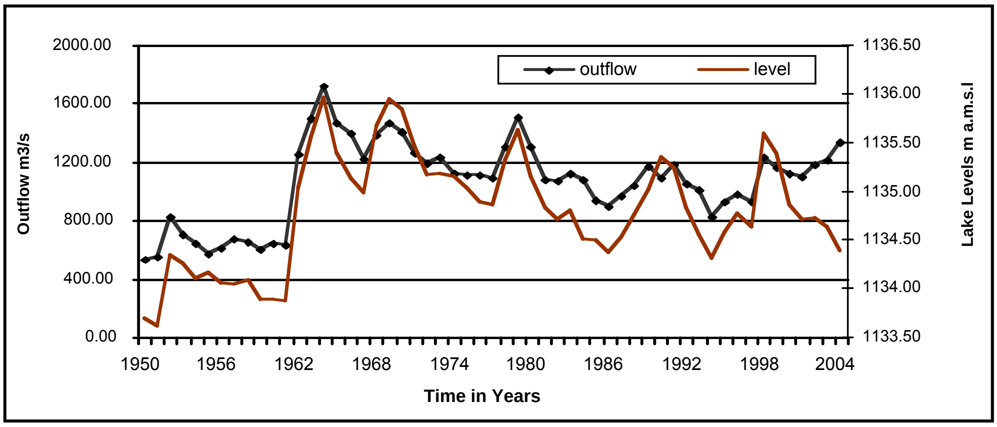


Figure 6. comparison of Nile outflow and lake level hydrograph.

Table 4. Nile outflow statistics summary.

	LVEMP Study (1950-2000)		LVEMP (2001-2004)		Long term (1950-2004)	
	Flow in Cumecs	% of Output	Flow in Cumecs	% of Output	Flow in Cumecs	% of Output
Nile Outflow	1,046	24	1,201.9	26.5	1,057.6	24.1

In the long term, the hydrologic balance was shifted by 1.11% from 1950-2004. This shift is masked when long statistical series are considered.

Water levels of Lake Victoria

The water levels of Lake Victoria over the last 104 years have exhibited striking changes in regimes that have baffled people so much. From 1900 to 1961, the lake was at a different regime that was different from the 1961-2002 regimes. The post 2002 regime has tendencies towards the pre 1961 regime. What further baffles people is which direction the trend will head for!

Data on the levels of Lake Victoria has been collected consistently since 1896. But the focus of this study is from 1950 to 2004. During this period the levels have fluctuated in response to natural processes of input and output in and out of the lake. Of all the above, it is evaporation over Lake Victoria, that is generally assumed to vary the least because the lake lies astride the equator. However even though evaporation varies with temperature, wind speed, humidity, sunshine, etc., it can also be affected by climatic variation.

Rainfall is generally recognized as the most variable component in the water budget with the further assumption that changes in rain over the lake will be similar to changes in rain falling on the land catchment. With stable climate lake levels can be stable year to year as the lake level adjusts to the balance of those inputs and outputs. However the long term record shows variability from year to year and particularly higher levels in the latter part of the twentieth century compared to the earlier half.

Lake levels have followed a general but variable downward trend since 1964's May 12th historic peak. But that long term trend has reversed several times over the last half century, e.g. the later 1970's, the early 1990's and the E*l Nino* rains of 1997 when periods of high rainfall occurred. Since the *El Nino* rains of 1997/98 when lake levels peaked in April of 1998 at 1135.77 metres above mean sea level (mamsl), the trend of the lake level has been dropping steadily.

The 2004 levels were the lowest experience since the flood of 1961-62 but the current low level condition is well above the recorded historic low level of the lake in March 1923 (Table 5) shows some key low flow periods in order of ascent (but the lowest ever recorded level was in March 1923 followed by the September 2004 lows).

Table 5. Historical low flow scenarios.

No.	Year	Month/Day	Level in m.a.m.s.l.
1	1923	March	1133.19
2	1961 (before the famous flood)	January	1133.7
3	2004	September	1133.99
4	1994	February	1134.18
5	1997	October	1134.21
6	1986	September	1134.26

High and low levels recur in an approximately cyclic manner both in the post-1961 level and earlier in the century. However, the most recent drop in level is a record for the post-1961 period.

Reasons for the current drop

To understand this, cumulative deviations from the normal regimes of rainfall, Nile outflow, evaporation and catchment discharges were developed. It was found that of all the processes above, rainfall and Nile outflow varied significantly to warrant their use in explaining the drop in levels. The following summarize the situation.

1. The lake system is such that the inputs (rainfall + river inflows+ groundwater input) and output (Nile outflows + Evaporation+ groundwater) affect the lake level (lake storage). Increase / decrease of any of the components of the system, specifically, rainfall, river inflows or Nile outflows either raises or lowers the lake level
2. After analysis it can be concluded that the observed fall in lake level is a result of a combination of two factors (a) reduced input in terms of rain and inflows into the lake system and (b) Increased outflows caused by excess releases at Jinja.
3. General absence/ limited rains on the lake in recent years resulted in falling of lake levels by 1.64m from 1998 to November 2004 with the year 2004 having been severely hit by this shortage of input.
4. Increased outflows for power generation resulted in a further fall in lake levels by 0.34m for the period June 2001, when the lake was in balance, to 3rd November 2004, when the lake was at its lowest (refer to Figure 3.14).
5. Excess releases accounted for 45% of the total fall in the period 2001-2004. Years 2003 & 2004 accounted for 77% of the extra lake drop with over 50% occurring in 2004 alone.

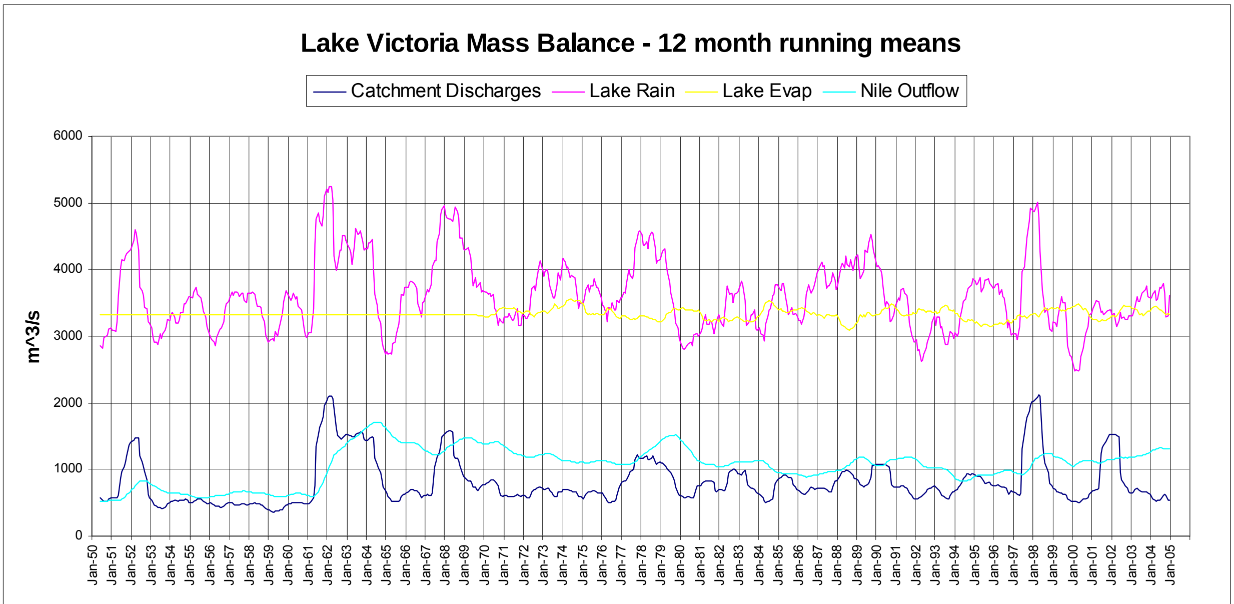


Figure 7. Long term series generated by the water balance model.

Future outlook

In summary, the drop being witnessed now is a result of below normal rainfall for most parts of 2004 and an increase in flow out of the lake at Jinja. It is likely that if the releases are reduced and with above-normal rainfall, the level will pick up. Alternatively, it can also be said that since the current levels are within the flow regimes of the pre 1961 flood, the lake could have completed the dissipation of the great flood waters and resuming its normal regime. Closer monitoring is required in order to understand this issue further.

Water balance for Lake Victoria

The Lake Victoria water balance was done by getting annual averages of rainfall, evaporation, catchment inflows and Nile Outflow at Jinja. The annual summaries were fed in the equation below:

$$\pm\Delta H = P + Q_{in} - E - Q_{nile} \qquad (1)$$

where ΔH is change in water level;

P is rainfall over the lake;

Q_{in} is catchment inflow;

E is evaporation from the lake; and

Q_{nile} is Nile outflow from Jinja.

When the ΔH is positive, water levels in the lake rises over the period, and when it is negative, the opposite is true. For purposes of comparison, rainfall and evaporation that are normally expressed in millimetres were converted to flow units that is expressed as the amount of water that wound ordinarily flow into, for rainfall and out of the lake, for evaporation. Table 3.6 summarizes the water balance for Lake Victoria in 3 different periods as indicated in the table.

Table 6. Summary of water balance for Lake Victoria.

Process	1950-2000 Flow m^3/s	%	2001-2004 Flow m^3/s	%	1950-2004 Flow m^3/s	%
Inflow						
Rainfall	3611.5	81.8	3644.0	84.2	3613.8	81.9
Basin discharge	805.3	18.2	686.2	15.8	796.6	18.1
Outflow						
Evaporation from lake	3329.8	76.1	3337.5	73.5	3330.3	75.9
Victoria Nile	1046.2	23.9	1,201.9	26.5	1057.6	24.1
Sum	**40.77402**		**-209.24**		**22.59122**	

The above table show that in the period 2001-2002, the lake lost on average -209.2 Cumecs from its storage. This accounts for the fall in levels in the same period amounting to 0.38m, as the model predicts and 0.38m, as from measured data. The long term water balance shows also a decline in the net storage as compared to the 1950-2000 period. This decline is caused by the negative net storage for the 2001-2004 period that is mentioned above.

Table 7. Model evaluation.

	1950-00	2001-04	1950-04
Change in Lake level – mass balance (m)	0.95	-0.38	0.57
Change in Lake level - actual Measurement (m)	1.06	-0.39	0.61

The accuracy of the water balance model was tested with the objective function being how best the model can reproduce measured data. The ΔH which was originally expressed as flows was expressed as depth over the lake area. The outcome for both model and measured data were correlated. A very high correlation coefficient of 0.999 shows that the model is accurate and can reproduce the levels of Lake Victoria, once given correct values of the input and output processes.

The model generated data that expressed variations over the long term of rainfall, catchment inflows, evaporation and R. Nile outflow. It can be clearly seen that in the last 4 years, rainfall over the lake did not change much, Nile outflow increased, catchment inflows decreased and water levels fell.

Conclusions

Although the outcome of the water balance in the long term period is positive, that is why current levels are still above the pre 1961 mean levels, the severity of the 2001-2004 trends may tilt the balance to an all time low in the next few years. This calls for a closer interest on the lake level trends by all the riparian states and address the causes of the levels fall. This will require concerted efforts to reverse the factors causing the decline like, better watershed management to increase flow from catchments, resumption of the natural flow practice at Jinja and a political will to conserve Lake Victoria. Therefore if the above are adopted, may be the levels can pick up.

It can also be concluded that the water budget method is a very important tool for monitoring and revealing the changes that are occurring in the water quantities of the lake. This means that there should be continuous and increased monitoring of the lakes hydrologic processes in order to address the management challenges of the lake basin.

Recommendations

1. There should be increased and consistent relevant data collection in all the three countries.
2. There is need for the three countries to continue cooperating in updating the water budget for Lake Victoria.
3. Efforts should be made to determine the role played by groundwater in the water budget of Lake Victoria although work to date including isotopic analysis suggest the role of ground water is minimal in the budget.
4. Data collection equipment and instruments should be standardized so that uniform data can be collected and used.
5. Stations should be established in un gauged catchments so that actual data is used in the balance other than estimates.
6. Undertake intensive studies on the possibility of regulating the lake to optimize its multiple but potentially conflicting uses to achieve maximum and sustainable socio-economic benefits for the riparian states.

References

Baker, D., 1969. *The Water Balance of Lake Victoria. Water Development Department.* Government of Uganda.

Baunly, H. L. and Baker, D., 1970. T*he estimation of discharges into Lake Victoria. Water Development Department*. Government of Uganda.

LVEMP, 2001. *Integrated Water Quality and Limnology Study of Lake Victoria.*

Ministry of Water, Lands and Environment. National Synthesis Reports, Kenya, Tanzania and Uganda

Long-term trends in water quality, water quantity and biodiversity at Lake Nakuru, Kenya

Jackson A. Raini

Fmr Project Ecologist, World Wide Fund for Nature, Lake Nakuru Project, P.O Box 33 Nakuru Kenya

Email: jraini2002@yahoo.com

Abstract

Routine *in situ* and laboratory measurements of pH, Conductivity, DO, temperature, alkalinity, salinity, TSS, TDS, Secchi depth and nutrient levels have been carried out at 13 sampling sites in Lake Nakuru, influent streams and urban effluent since 1993. Mean values for 8 lake sampling sites are: pH (10.14), Temperature (27.3^{o}C), Conductivity (38.8mS/Cm), DO (9.21mg/l, 83% saturation), Secchi depth (10.3Cm). Mid-lake sampling at 19 sites showed the presence of a thermocline of upto 4 ^{o}C gradient (surface to bottom). DO levels (>2m/l) remained within the tolerance thresholds for *A. fusiformis* and the fish (*S. grahami*). Trace metal concentrations in Lake Nakuru water and sediment as determined by X-ray fluorescence spectroscopy, varied over an order of magnitude for each of the metals for dry sediment and water samples. In 1995, results (ppm) ranged as follows: chromium (8.6-155), zinc (44-629), lead (4-102), copper (4.5-94.8), arsenic (1.5-3) and Nickel (1.5-16.5), mercury (1.5-3), selenium (1.5-8.8), titanium (748-14,367). In 1995 pesticide residues by Gas Liquid Chromatography (ppb) were: DDT (3.3-19.2), DDD (5.2-9.6), DDE (7.8-8.6), g-BHC (3.4-5,470) and dieldrin (1-29.1). River and sewage channel sediment revealed presence of heavy metal and pesticide residues.

Mean Lake Depth (1992-2003) was 1.01m (range <0.1 to 4.5m). Lake volume ranged from 1.0 m^3x10^7 to 18 m^3x10^7. The long-term areal precipitation over the catchment is 940mm/annum. Isohyetal analyses show a decrease of rainfall towards the lake. The proportion of annual precipitation over each sub-basin was as follows: Njoro: 29%, Nderit: 23%, Makalia: 17%, Ngosur: 17%, Larmudiac: 6%, and Lion Hill: 8%. Annual runoff volume from each sub-basin is as follows: Njoro: 39%, Ngosur: 23%, Makalia: 21%, Nderit: 13%, Lion Hill 3%, and Larmudiac, 1%. Only 3.2% of the annual rainfall is available to recharge the basin. The highest evaporation coincides with periods of highest radiation and temperature. Evaporation is always higher than precipitation.

Key words: Water Quality trends, Water Quantity, Contaminants, Ecosystem health, Biodiversity

Introduction

Lake Nakuru is one of several shallow, alkaline-saline lakes lying in closed hydrologic basins in the eastern African Rift Valley that stretches from northern Tanzania through Kenya to Ethiopia (Livingstone & Melack, 1984). It occupies an area of 44 Km^2at an altitude of 1,759masl. It is a shallow lake and an extreme ecosystem in which intervals of complete dryness alternate with periods of flooding and high water. The biota of the lake is depauperate, although persistent populations of the cyanobacteria, 'Spirulina' (Ridley *et al*., 1955), *Arthrospira fusiformis* (Voronichin) Komárek (Hindák, 1985) occur, forming the base of the food chain and supporting large concentrations of lesser flamingos (*Phoeniconaias minor*). The only fish species in Lake Nakuru is the filter feeding cichlid, *Sarotherodon alcalicum grahami,* introduced from Lake Magadi between 1953 and 1962 to combat mosquito breeding. The introduction of fish substantially increased the diversity of the lake ecosystem by extending the food chain to over 30 species of fish-eating birds. Other organisms, found in varying abundance, include a copepod (*Lovenula africana*), Cladocerans, chironomid larvae (*Leptochironimus deribae, Tanytarsus spp.*), three species of rotifers (*Brachionus dimidiatus, B. plicatilis*, and *Hexartha jenkinae*), and three species of water bug (*Micronecta jenkinae, M. scutellaris* and *Sigara hieroglyphica kilimanjaronis).*

As is typical of shallow, saline lakes, worldwide, climatic variations have caused large changes in depth and salinity on annual, decadal and longer time scales and has had major consequences for the ecology of the lake. Daily fluctuations in heating and cooling result in strong diel cycles of stratification and mixing (Melack & Kilham, 1974). High insolation and adequate supply of nutrients usually support abundant phytoplankton (Vareschi, 1982). Supersaturation of dissolved oxygen in the upper waters during the day often results from the high rates of photosynthesis (Melack & Kilham, 1974; Vareschi, 1982). Alkaline-saline lakes rich in bicarbonate and carbonate and usually called soda lakes, such as Lake Nakuru, are among the world's most productive, natural ecosystems (Livingstone & Melack, 1984). A conspicuous feature of these lakes is often the presence of over a million Lesser Flamingos (*Phoeniconaias minor* Geoffroy) grazing on thick suspensions of phytoplankton. Each day a population of this size is estimated to harvest about 180 tonnes of the cyanobacteria from the lake and within 24 hours this harvest is replaced (Vareschi, 1982). Under optimal conditions, the fish *S. grahami* at the lake can achieve a total icthyomass of about 400 tonnes (Vareschi, 1982). Low species diversity but abundant populations of aquatic organisms make soda lakes especially suitable for the study of trophic dynamics and ecosystem processes (Vareschi & Jacobs, 1985).

Efficient management of Lake Nakuru entails taking into cognizance of processes occurring beyond its littoral fringe. Strong evidence exists to demonstrate that the ecological integrity of Lake Nakuru is dependent on the rational and sustainable utilization of resources within its catchment basin. Water balance and water quality are two critical aspects of Lake Management that must be monitored and managed. Management of the lake can no longer be reactive. Reliable information on the status of the catchment and its water resources needs to be made available to decision makers and the community on a timely and regular basis. Such is the objective of the Lake Nakuru Water Quality Monitoring Programme.

Materials and methods

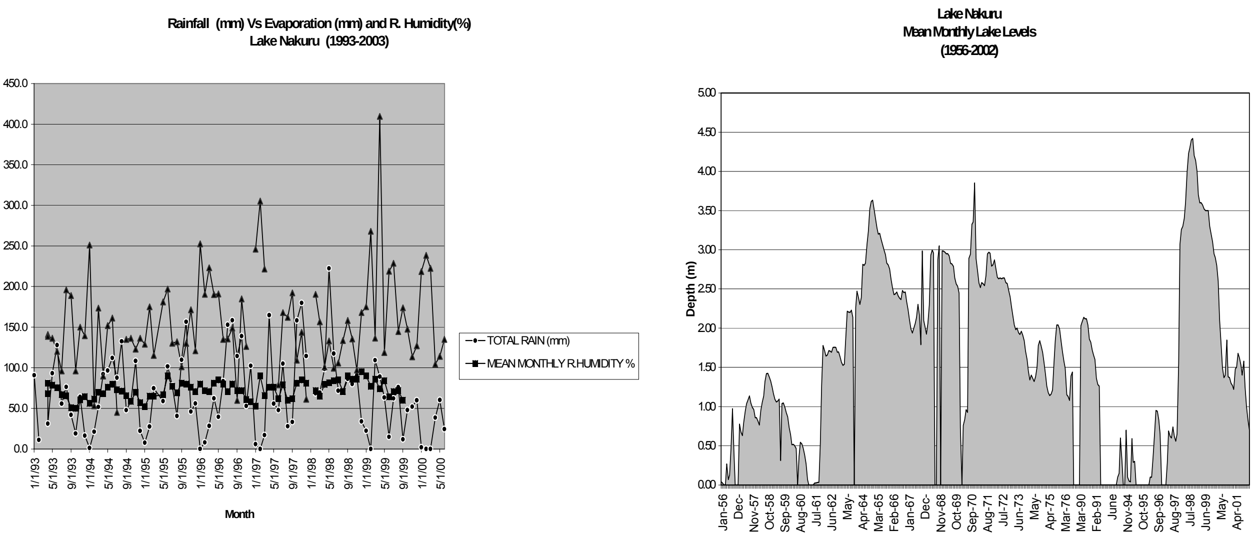
Weather data was recorded at the Lake Nakuru National Park Weather Station. The weather station is registered with the Kenya Meteorological Department and is equipped with a class A evaporation pan, standard rain gauge and autographic rainfall recorder, Gunni bellani radiometer, hygrometer, maximum and minimum thermometer and a hand held anemometer. Parameters monitored at the station include the rainfall (duration and intensity), evaporation rates, ambient temperature, relative humidity, wind speed and solar radiation. A network of 20 rainfall stations was sited at different altitudes and within identified isohyets at various river basins. Lake levels were monitored weekly using metric staff gauges located at the northwestern and southwestern shores of the lake. River depth was measured using metric staff gauges and automatic flow recorders located at 3 major rivers while river discharge volume was measured using a Bargo calibrated OTT C 2 Small Current Meter (Ref: # 10.150.005.B.E).

Water quality monitoring was carried out weekly at 8 sampling sites in Lake Nakuru and at 5 sites along influent streams and sewage channels. Sampling locations were outlined from the 1:50,000 topo maps of Survey of Kenya and geo-referenced using a Gemini GPS and were chosen on the basis accessibility and suitability as baseline, impact or recovery stations. In 1998, the frequency was decreased to once every fortnight. Whenever lake levels permitted (1996 to 2003), a rubber dingy or a fiber glass boat and a 15/40 HP Enduro outboard engine were used for sampling 18 mid lake sites located along six established transects. Mid-lake water samples were collected using a plastic Van Dorn-type messenger closed sample bottle with a capacity of 1.25lts. During periods of low lake levels, duplicate grab samples were collected using 1000ml plastic beakers. Between 1993 to 1995 a portable Jenway Electro-chemical analyzer (Model 3405) was used for *in-situ* pH, Temperature (^oC), Dissolved Oxygen (DO), Electrical Conductivity (mS), ORP (mV) and Salinity (‰) measurements. In August 1995 the parameters were measured using WTW Multiline P4 portable meter (Wisesens-chaftlich Technische Werkstätten Weiheim, Germany). Water transparency was measured using a Secchi disc (Ø 20cm). Standard methods (16) were used for determination of total suspended solids (TSS, method 2540D), and alkalinity (method 2320 B). Filtered water was tested for total nitrates (TN) total phosphates (TP) and total kjedhal nitrogen (TKN) using a HACH DR/2010 Spectrophotometer. Algal biomass was estimated by measuring chlorophyll a Spectrophotometrically (standard methods 10200H (16). In April 1994, 22 samples of deep sediment from Lake Nakuru were collected using a soil auger and submitted to the Kenya Bureau of Standards for heavy metal and pesticide residue analysis. The sediments were collected at a depth of 0.5m and analysed using Gas Liquid Chromatography (GC). Between February and May 1995, 18 samples of surficial sediment (the top 10cms) were collected from 10 sites around Lake Nakuru using a 15x15cm Ekman Grab Sampler. Analysis for heavy metals was done by the Institute of Nuclear science Techniques, University of Nairobi using X-ray fluorescence spectroscopy. Pesticide residue analysis was carried out by the Pesticide Chemistry Laboratory of the Kenya Agricultural Research Institute using Gas Liquid Chromatography. A total of 7 sediment samples from river and sewage channels were also analysed in an attempt to determine the portals of entry of contaminants into the lake. In 1996 a total of 80 sediment samples were collected at 100m intervals along two transects running across the lake. Granulometric analysis was done to determine sediment particle size. All the 80 sediment samples were wet sieved using an electric soil shaker in a set of five sieves of 800µm, 400µm 100µm 50µm and 25µm.

Samples of phytoplankton and zooplankton were collected using a 50µm open net and 100µm plankton nets respectively and preserved using 4% formaline. Identification was done by using 40x Olympus microscope. Flamingo numbers were counted using a 20-60x-zoom telescope. In April 1999, several lake transects were trawled for fish. Net held over the side of a fast moving boat was used to trawl the transects. A total of 675 fish were caught, weighed and measured.

Results and discussion

The basic measurements required to determine the water balance of the lake include surface flow into the lake, rainfall over the lake and its catchment basin, ground water input into the lake, the lake levels, water loss through seepage, evaporation as well as climatological parameters such as solar radiation, temperature, relative humidity, wind speed and sunshine hours. The lake had a mean depth of 2.5m, a maximum depth of 4.5m and a water volume that is about $92 \times 10^6 m^3$ (long-term means 1925-1979 cited by Vareschi, 1982). The recent mean monthly lake depth (1992-2002) is 1.01m (range <0.1 to 4.4meters). The water volume fluctuated between $1.0\ m^3 \times 10^7$- when the lake dried almost completely to about $18\ m^3 \times 10^7$ during the 1997/1998 *El niño* floods. The Baharini springs and other springs along the eastern shoreline are perennial and contribute about 0.6 m^3s^{-1}.



(a) (b)

Figure 1. (a) Rainfall (mm) Vs Evaporation and Humidity (%) at Lake Nakuru (1993-2003) (b) Mean Monthly Lake Levels, 1956-2003.

The long-term areal precipitation over the catchment is 940mm/yr. Rainfall exhibits a strong seasonality with trimodal distribution. Peaks are centered around April, August and November. The respective percentages accounted by the peaks are 17%, 13% and 8%. Isohyetal analyses show a general decrease of rainfall towards the center of the lake, which is located in a rain, shadow. The proportion of annual precipitation over each subbasin within the catchment is as follows: Njoro: 29%, Nderit: 23%, Makalia: 17%, Ngosur: 17%, Larmudiac: 6%, and Lion Hill: 8%.

The proportion of annual runoff volume from each subbasin is as follows: Njoro: 39%, Ngosur: 23%, Makalia: 21%, Nderit: 13%, Lion Hill 3%, and Larmudiac, 1%. During the period 1993-2000, annual mean temperature was 18.2 ± 0.9 ^{o}C, mean annual evaporation (1539.4± 59mm), mean daily solar radiation on the lake surface was 421.7 ± 92 Langleys. Evapotranspiration from the basin accounts for 96.8% of the annual basin rainfall. Only 3.2% of the annual rainfall is available to recharge the basin. The highest evaporation coincides with periods of highest radiation and temperature. Evaporation is always higher than precipitation.

Monitoring water quality

A water quality monitoring programme has been in place for the last 10 years. Since September, 1993, regular measurements of pH, conductivity, dissolved oxygen, temperature, alkalinity, salinity, suspended solids, dissolved solids, turbidity and nutrient levels have been carried out at 13 sampling sites on the lake and along its influent water channels (Tables 2 and 3). These parameters are considered to be fundamental for evaluating water quality. The long-term water quality trends are presented below.

Table 1: Summary of mean values for 8 lake sampling sites at Lake Nakuru (1993-2003).

	pH	Temperature	Conductivity	Salinity	DO	DO	ORP	Secchi Depth
		(oC)	(mS/cm)	(‰)	(mg/l)	(%)	(mV)	(cm)
	(n=1013)	(n=1115)	(n=1074)	(n=235)	(n=1106)	(n=504)	(n=426)	(n=624)
Mean	10.14	27.30	38.81	15.37	9.21	83.25	-69.16	10.26
Median	10.21	27.3	35.6	15.5	7.3	36.95	-145	9
Range	7.8-12.3	14-49.7	0.1-180	0-34.1	0-55.5	0.3-406	-345	0-83

Table 2: Mean values for 8 lake sampling sites at Lake Nakuru (1993-2003).

Sampling site	pH	Temperat.	Conductivity	DO	DO	ORP	Secchi Depth	Wind speed
		(oC)	(mS/cm)	(mg/l)	(%)	(mV)	(cm)	(m/sec)
	n=150	(n=162)	(n=155)	(n=160)	(n=71)	(n=61)	(n=88)	(n=85)

Lake shore gauge	10.0	25.9	30.2	7.2	62.9	-55.0	11	1.3
R. Njoro Mouth	9.8	26.9	33.0	6.9	57	-54.5	11	1.7
Lakeshore Drums	10.2	27.6	44.0	9.4	93.4	-93.6	10.5	3.1
President pavilion	10.3	28.6	54.0	11.7	85.6	-59.7	11.2	3.6
R. Makalia Mouth	10.1	27.4	32.1	10.2	94.0	-83.3	8.1	
Btn Makalia/ Nderit	10.2	27.8	39.5	10.6	109.0	-74.8	10.0	4.8
R. Nderit Mouth	10.0	27.2	27.6	8.0	102.0	-90.8	9.2	4.6
Kampi ya Nyati	10.3	26.9	44.7	8.7	86.2	-59.3	9.1	2.9

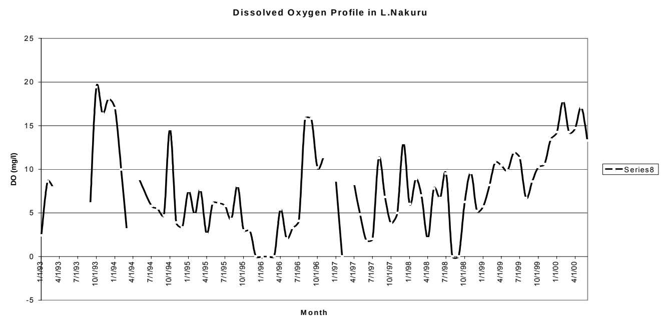
Table 3: Mean values of Water Quality variations at influent streams to Lake Nakuru.

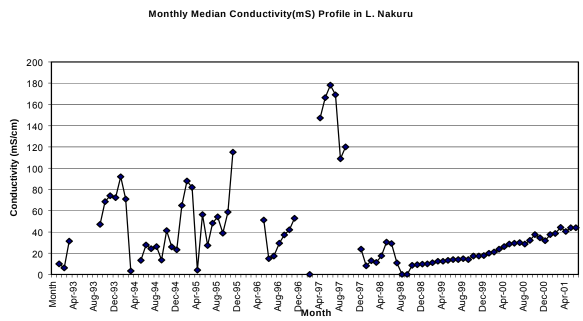
Influent	pH	Temperat (oC)	Conduct (mS)	Salinity (‰)	DO (mg/l)	DO (%)	ORP (mV)	Secchi (cm)
Baharini spring n=168	9.0	25.6	0.5	0.6	7.6	58.1	28.4	32.2
River Njoro (n=153)	8.1	20.2	0.7	0.1	9.3	28.2	35.1	33.6
River Makalia (n=121)	8.2	20.3	2.6	0.0	7.3	21.6		
River Nderit (n=58)	8.0	22.4	0.5	0.0	7.5	27.0	42.2	12.2
Town Sewage (n=123)	8.0	23.0	1.1	2.9	11.2	49.4	-15.6	20.8
Njoro Sewage (n=55)	8.8	21.7	1.9	0.5	7.5	33.7	-70.8	11.7

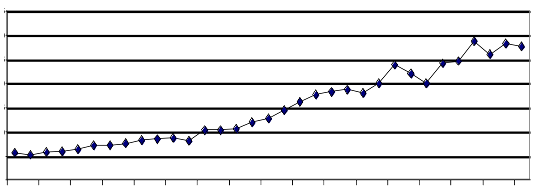
Table 4: Values for Nutrients, Alkalinity, Solids, Chlorophyll *a* and BOD at Lake Nakuru.

	TDS	TSS	T.Alkalinity	OP	TP	NO2-N	NO3-N	NH4-N	NH3	TKN	Chl a	BOD
	g/l n=313	g/l n=313	Mg /l Caco3 n=313	Mg/l n=37	Mg/l n=37	Mg/l n=37	Mg/l n=33	Mg/l n=33	Mg/l n=15	Mg/l n=33	Mg/l n=15	Mg/l n=33
Mean	16.4	2.1	3,526	2.4	0.1	1.2	3.6	153.8	274.9	4.5	35.1	1653.3
median	8.1	0.5	1,750	2.3	0.0	1.3	0.8	162.0	268.0	4.0	32.6	368.0
Max	109.0	21.5	33,000	5.1	1.8	1.9	54.0	260.0	436.0	9.3	42.8	4400.0
Min	0.1	0.0	50	1.1	0.0	0.3	0.4	18.0	23.0	2.2	24.6	192.0

Below are charts showing the long-term trends (monthly median values) for water temperature, conductivity, dissolved oxygen, and salinity at Lake Nakuru between 1993-2003 (Figure 2). Lake Nakuru recorded consistently high water temperature while DO and EC displayed marked temporal and spatial variations. The highest EC level was recorded at the Presidents pavilion-sampling site (Table 2) The gaps indicate periods when the lake was dry.







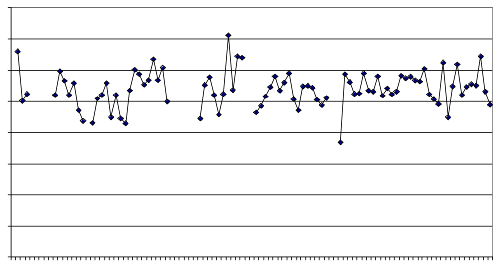


Figure 3. Vertical Profile for Temperature (^{o}C) Vs Depth (m).

In April 1996, lake in-filling began with the onset of the long rains. The lake attained a mean depth of 0.61m in July and a peak depth of 0.95m. in August to September. This offered an opportunity for mid lake sampling. The graphs below (a) show the DO (mg/l) Vs Depth (m) and (b) Temperature (^{o}C) Vs Depth vertical profiles between 1996 –1999.

Temperature of Lake Water varied seasonally and according to the time of the day. Surface water temperature ranged between 20-30^{o}C. Vertical median temperature at 19 sits along 6 transects (November 1998), revealed the presence of a thermocline with a 4 ^{o}C gradient between mean surface and bottom water temperature. This distinct thermocline builds up during the day, and is abolished by wind induced mixing in the late afternoon or early evening.

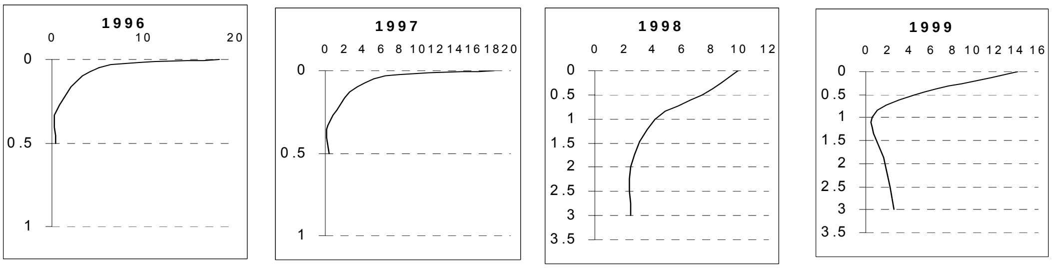


Figure 3.Vertical Profile for Temperature (^{o}C) Vs Depth (m).

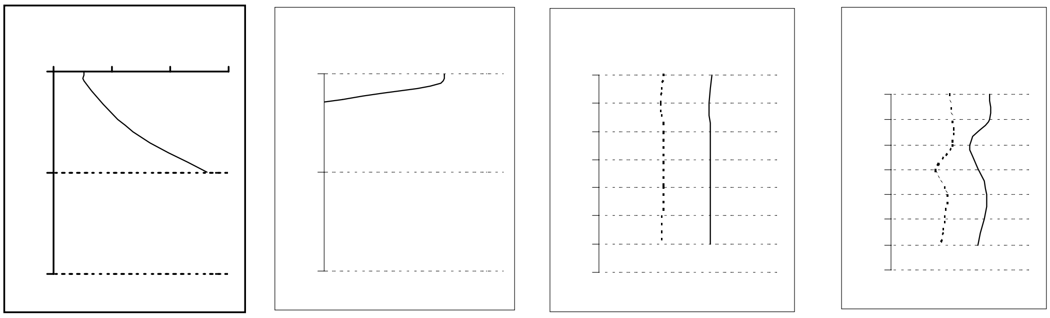


Figure 4. Vertical Profile for Conductivity (mS) and Salinity (‰) Vs Depth (m).

Heavy metals and pesticide residues

Trace metal concentrations in Lake Nakuru water and sediment as determined by X-ray fluorescence spectroscopy, varied over an order of magnitude for each of the metals for dry sediment and water samples. Results of heavy metal and nutrient analysis of urban discharges is presented (table 4). In 1995, results (ppm) ranged as follows: chromium (8.6-155), zinc (44-629), lead (4-102), copper (4.5-94.8), arsenic (1.5-3) and Nickel (1.5-16.5), mercury (1.5-3), selenium (1.5-8.8),titanium (748-14,367). In 1995 pesticide residues by Gas Liquid Chromatography (ppb) were: DDT (3.3-19.2), DDD (5.2-9.6), DDE (7.8-8.6), g-BHC (3.4-5,470) and dieldrin (1-29.1). River and sewage channel sediment revealed presence of heavy metal and pesticide residues. The rivers are a conduit for heavy metals and a variety of pesticide residues whereas the sewage channel is a major conduit for heavy metals and g-BHC.

Table 5. Mean heavy metal, nutrient s and organic pollutant levels in urban effluent.

Parameter	Town Sewage WWF, 1992 (n=4)	Njoro Sewage WWF,1992 (n=4)	Storm Water WWF,1992(n=4)	Town Sewage (JBIC SAPS) 2001
	µg/l	µg/l	µg/l	µg/l
Ti	101	89.3	12.2	-
Cr	<20	55.8	345	3.4
Mn	59.1	37.9	4.28 mg/l	1,350
Fe	3.59 mg/l	3.75	210mg/l	
Ni	12.3	19.0	<10	2.96
Cu	33.1	31.5	289	<5
Zn	218	250	3.7 mg/l	6.6
Hg	<1.0	<1.0	50.1	-
Pb	32.1	22.2	805	<0.1
Mean Nutrients levels (mg/l)				
	Town Sewage WWF, 1992 (n=4)	Njoro Sewage WWF,1992 (n=4)	Storm Water WWF,1992 (n=4)	Town Sewage 2001 (JBIC SAPS I Study)

NH_4	69.8	93	34.9	
TP	33.5	33.7	16.75	
TN	0.001	0.001	0.001	
TSS	60	260	3,540	
BOD	90	325	900	
COD	128	472	1,064	

Biotic response to changes in water quality

Between 1994 and 1997, *A. fusiformis* population made occasional and very transient appearances. In place of *A. fusiformis*, the algal comopsition of the lake has been largely dominated by *Anabaenopsis spp., Anabaena flos aquae* and *Anabaena abijatae, Microsystis spp Coccoids* and various species of diatoms. The lakes' zooplankton population also burgeoned after the El niño rains and were found in varying abundance, These include a copepod (*Lovenula africana*), Cladocerans (daphnia), Beetle larvae, Ceratopoginids, chironomid larvae (*Leptochironimus deribae* and *Tanytarsus spp*.), three species of rotifers (*Brachionus dimidiatus, B. plicatilis*, and *Hexartha jenkinae*), and three species of water bug (*Micronecta jenkinae, M. scutellaris and Sigara hieroglyphica kilimanjaronis*). The fish population, which for several years was confined to rivulets flowing into the lake, re-colonized the lake. Pelicans and other fish eating birds returned. Over 60% of the fish catch was composed of sub-adults with a mean length of 5.8cms 9Range: 2.2-10.9cm) and a mean weight of 4.3g (range: 0.36-22.94g). The rest of the catch consisted of fish with a mean length of 12.07 cm (range: 11.1- 14.2cm) and a mean weight of 25.9g (range: 18.06-41.95g). Mature, fully-grown fish accounted for only 1% of the catch. Young fish were often infested with intestinal nematodes. In September 1993, an epidemic among the lesser flamingos occurred in Lakes Nakuru and Bogoria. An estimated 30,000 flamingos died during this epidemic. 1n 1995, a second epizootic involving the lesser flamingo occurred at Lake Bogoria. Few mortalities occurred at Lake Nakuru, which was practically dry and the flamingo population was small (<10,000). The flamingos dispersed widely, occupying sewage lagoons and dams around L. Nakuru and L. Bogoria and migrating as far as Lake Simbi in western Kenya. Due to this dispersion, it was difficult to estimate the number of flamingos that died in the epidemic. Unlike the 1993 epidemic, the birds were in poor physical condition and were experiencing nutritional stress. Since July 1998,lesser flamingo deaths have occurred sporadically at Lakes Bogoria and Nakuru for several months particularly in the in the years, 2000, 2001 and 2003. The epidemics have raised the specter of the involvement of a toxic substance(s).

Acknowledgements

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References

Vareschi E., 1982. Ecology of Lake Nakuru, Kenya. Abiotic factors and primary production. *Oecologia*, 55, 81-101.

Greichus Yvonne A., A. Greichus, B.Ammann and J. Hopcraft, 1977. *Insecticides, Polychlorinated bi-phenyls and metals in African Lakes ecosystems*. III Lake Nakuru, Kenya.

Livingston Da, Melack J. M., 1984. Some lakes of sub-Saharan Africa. *In.* Tab F. B., ed. *Lakes and Reservoirs*. Elsevier, Amsterdam, The Netherlands.

Mavura W. & Wangila, 2000. *The Status of Lake Nakuru Pollution by Heavy Metals and Pesticide Residues.*

Melack, J.M., P. Kilham, 1974. *Photosynthetic rates of phytoplankton in East African Alkaline, saline lakes*. Limnol. Oceanogr. 19: 743-755.

Vareschi, E. & J. Jacobs, 1985. The ecology of Lake Nakuru. VI. Synopsis of production and energy flow. *Oecologia* 65: 412-424.

American Public Health Association, American water Works Association, and Pollution Control Federation. 1995. *Standard Methods for the Examination of Water and Waste Water,* 19th ed. American Public Health Ass. Washington, DC.

Management implications of the physical limnological studies of Rusinga Channel and Winam Gulf in Lake Victoria

Romero Jose R. [1], Ryan Alexander[1], Jason P. Antenucci[1], Greg Attwater[1], Tom Ewing[1], Sheree Feaver[1], Jorg Imberger[1], Patrick Khisa[2], Carol Lam[1], Henry Njuguna[2], and Kenji Shimizu[1]

[1] Centre for Water Research, Univ. of Western Australia, Crawley, WA, 6009, Australia 35 Stirling Highway Crawley, WA 6009 Australia
Email:romero@cwr.uwa.edu.au
[2] LVEMP Coordination Office, Re-Insurance Plaza, 2nd Floor, Kisumu 40100, Kenya

Abstract

During Apr.-May and Aug. 2005 intensive field investigations were undertaken to characterize the physical limnology of Winam Gulf and Rusinga Channel regions of Lake Victoria. Both the field data and numerical simulations with the three-dimensional (3D) hydrodynamics model, ELCOM (Estuary, Lake and Coastal Ocean Model), during these two field studies led to two major conclusions. First, though water currents in Rusinga Channel can be quite large (ca. 10-50 cm s^{-1}), exchange through the channel is low because the tidal-like oscillations result in low net transport over a 24 hour period. Second, Winam Gulf can be segmented into four regions, two of which have relatively low flushing rates as compared to time scales of biogeochemical processing. Regionally, on the basis of these physical limological findings the implications for water quality management strategies for Winam Gulf are presented. Internationally, the role of Winam Gulf as a net source or sink of nutrients and pollutants to the open waters of Lake Victoria is addressed.

Key words: Hydrodynamic Modeling, Water Quality Management, Winam Gulf

Introduction

Lake Victoria has undergone considerable ecosystemic changes over the past 50 years. Most of the haplochromine cichlids became extinct after the explosive increase of introduced Nile perch (*Lates niloticus*) in the 1980s (Kaufman 1992). The extent of anoxia in the deep portions of offshore waters has increased recently (Hecky 1993) in comparison with past conditions (Talling 1966). Phytoplankton biomass has increased in the Ugandan offshore waters (Muggide 1993, Hecky and Bugenyi 1992) and in the Kenyan inshore waters of Winam Gulf (Ochumba and Kibaara 1989). Further, the occurrence of blue-green algal blooms has become more frequent in Winam Gulf (Ochumba and Kibaara 1989). Three hypotheses are likely to be the causal mechanisms for the eutrophication of Lake Victoria over the past 45 years (Reinthal & Kling 1994). Firstly, the loss of the haplochromine cichlids has resulted in less grazing of the phytoplankton. Secondly, human population growth and land use change has led to increased nutrient loading and greater algal biomass. Lastly, climate change has caused a shallower mixed layer, increased persistence of seasonal stratification, and less ventilation of bottom waters.

The Kenyan portion of Lake Victoria encompasses the largest embayment of the lake, Winam Gulf, and has numerous sources of nutrient and pollutant loading. Recent debates regarding the contribution of Winam Gulf to the eutrophication of the main basin of Lake Victoria has not considered the exchange between the water bodies. In this paper, we build on recent physical limnological investigations of Rusinga Channel from a companion paper (Antenucci *et al.,* 2005) to assess the role of Winam Gulf on the overall eutrophication of Lake Victoria by addressing the exchange dynamics. Secondly, we consider the exchange amongst different regions of the Gulf, and evaluate the scope for catchment management to improve local water quality. Field data from recent surveys of Winam Gulf (Apr.-May and Aug. 2005), three-dimensional hydrodynamic simulations, and a brief literature review serve as the basis to consider possible management strategies to improve the Gulf's water quality.

Study site

The three riparian nations of Kenya, Uganda, and Tanzania have 6%, 45%, and 49% of Lake Victoria (3°S to 0.5°N, 31°40'E to 35°E) as territorial waters, respectively. The lake has a surface elevation at ca. 1134 m, mean surface area of 68,800 km^2, mean depth of 40 m, and maximum depth of ca. 70 m (Figure 1A). The catchment area is approximately 195,000 km^2, which also includes Rwanda and Burundi. Lake Victoria serves as an important water resource for domestic and industrial purposes, supports a valuable fishery, and is used for transportation for regional trade.

The Kenyan waters of Lake Victoria include Winam Gulf (also known as Kavirondo Gulf or Nyanza Gulf) and the northeastern corner of the main lake, a total area of ca. 4200 km^2 (Figure 1B). Winam Gulf has a surface area of ca. 1,800 km^2 and is connected to the open waters by Rusinga Channel. The Channel has a complicated bathymetry with numerous deep holes separated by sills. Several large rivers (Sondu and Nyondo) enter the Gulf in the southeast as major point sources of nutrients and sediments (Okungu & Opanga, 2004) as considerable agricultural and industrial activity occurs in these catchments generally associated with sugar cane (Scheren *et al*., 2000). Several large cities (Kisumu, Homa Bay) discharge domestic waste effluent into the Gulf, resulting in high loads of biological oxygen demand (Scheren *et al.*, 2000).

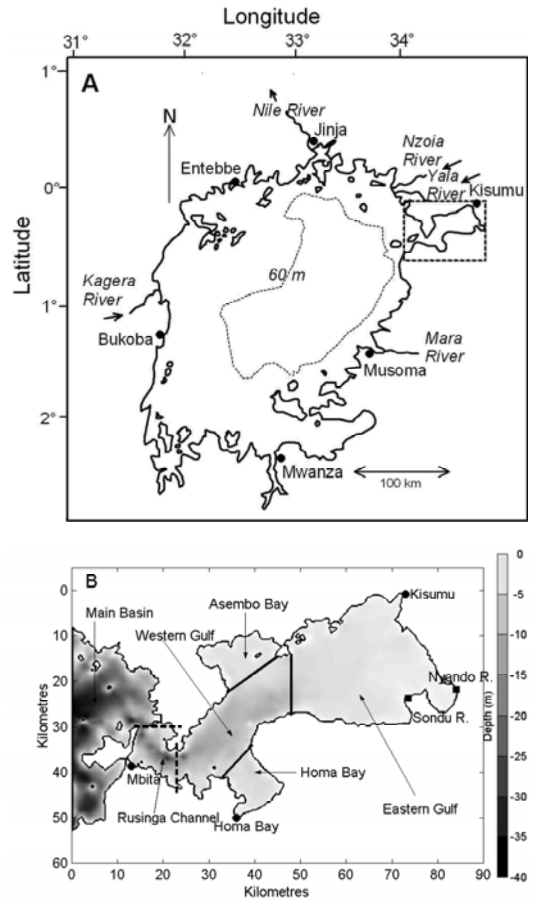


Figure 1. (A) Map showing the shoreline of Lake Victoria, the 60 m isobath, and major rivers and cities where the dashed box delineates the area of this study. (B) Map of the shoreline with shaded bathymetry, major cities (circles), major rivers (squares), hydrodynamic regions outlined by bold lines (Homa Bay, Asembo Bay, Eastern Gulf, Western Gulf), and dashed lines that define boundaries between the Western Gulf and the Channel, and the Channel with the Main Basin where exchange is estimated.

Methods

Therimstor chains were deployed in the offshore waters at T1 and Rusinga Channel at T2 during two field experiments (Apr. 22-May 4 and Aug. 5-16 2005). A full meteorological station (short and total radiation, air temperature, relative humidity, wind speed and direction) at T2 recorded measurements 2 m over the water surface with only wind speed and direction measured at T1. Water currents and directions, and surface level were measured with an acoustic doppler current profiler at T2. During the second field experiment (Aug. 5-16) additional surface level measurements were made at T1 and the Kisumu shoreline. A free-falling profiling instrument, the Finescale Profiler, was used to measure temperature, salinity, dissolved oxygen, pH and turbidity at 1 cm vertical resolution along transects of Rusinga Channel, Homa Bay and Winam Gulf (Gulf transects only during the 2nd study). The Fluoroprobe was attached to the Finescale Profiler and simultaneously recorded the biomass of various phytoplankton groups (diatoms, blue-greens, greens, cryptophytes). Refer to the companion paper by Antenucci *et al.* (2005) for more details on the instrumentation and locations of stations.

Three-dimensional hydrodynamic simulations with the Estuary, Lake, and Coastal Ocean Model, ELCOM (Hodges *et al.,* 2000), were carried out over both field studies to characterize transport in the Gulf and Channel. A uniform numerical grid was used with 250 m horizontal and 1 m vertical resolution (Figure 1B) and a model time step of 6 minutes. Free slip boundaries were used to model the sidewall (land) boundaries with a drag parameterization at the bottom boundaries. Meteorological forcing from T2 was applied over the entire domain except for wind speed, where T1 data was applied over the Main Basin. Water temperatures at the western open boundary were forced with T1 data. The open boundary surface levels were forced with T1 data during the second study, and T2 surface levels shifted back by 3 hours during the first study.

In order to estimate exchange and visualize transport, conservative tracers were employed in the ELCOM simulations. Transport in Rusinga Channel is similar to tidally-driven estuarine systems (Antenucci *et al.,* 2005). A common factor for defining the effectiveness of tidally induced harbor and lagoon flushing is the average per cycle, E, given by Nece and Richey (1975) as, $E=1-(C_i/C_0)^{1/i}$, where C_i is the concentration of a conservative substance after i tidal cycles and C_0 is the initial concentration of the same substance. Here, we defined i as the number of days, so that E was the percent flushing per day. We initialized two tracers over the Gulf and Channel to calculate flushing (dashed lines in Figure 1B) and three tracers to visualize flushing of several regions within the Gulf (bold lines in Figure 1B).

Results

During Aug. 2005 the study region of Winam Gulf, Rusinga Channel, and the northeastern offshore waters of Lake Victoria had gradients in physico-chemical and phytoplankton measurements (Figure 2). The salinity of Winam Gulf was about twice that of the offshore waters because of evaporative concentration during both the wet (Apr.-May, not shown) and dry (Figure 2) seasons. A similar spatial signature was evident in turbidity during both seasons with higher levels in the Gulf. High blue-green biomass occurred in the Eastern Gulf with surface accumulations of *Microcystis* observed during Aug. 2005 (J. Imberger, pers. obs.), presumably because buoyancy regulation by these algae provides a competitive advantage to overcome light limitation. In contrast, diatom levels were elevated primarily in Rusinga Channel where turbidity was lower and bio-available phosphorus from the offshore waters (Gikuma-Njuru and Hecky, 2005) was in closer proximity.

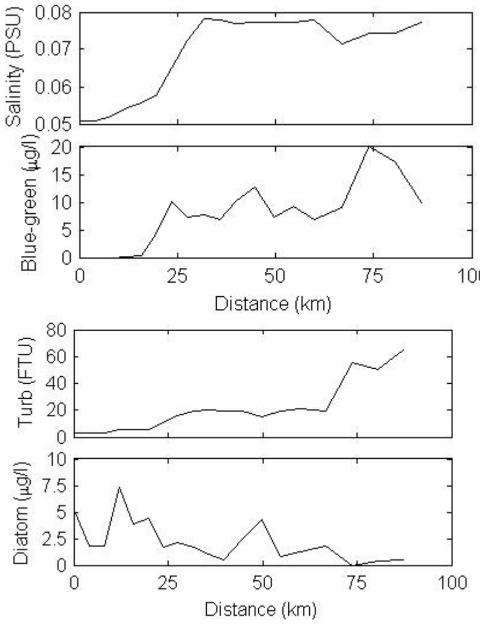


Figure 2. Transect from the Main Lake (0 km) to Kisumu (85 km) of averages over the upper 8 m of the water column of salinity (upper left panel), turbidity (upper right panel), blue-greens (lower left panel), and diatoms (lower right panel) during Aug. 14-16 2005. See Antennuci *et al.* (2005) for station locations

Temperature and current dynamics at T2 have been discussed in the companion paper (Antenucci *et al.* 2005). Here we briefly compare the observations at T2 with the simulation output of Apr.-May 2005. ELCOM modeled the temperature (Figure 3) and currents (Figure 4) well at T2. Diurnal stratification during the first half of the simulation was captured over the upper several meters of the water column, though it persisted longer into the evening than field measurements. The abrupt cooling of the water column mid-way through the simulation was captured by ELCOM, as was diurnal stratification in the upper several meters on Apr. 30, May 1 and May 3. Peak current speeds were generally captured by the simulation in phase with observations and generally of the same velocity. Similarly, current directions were captured well by ELCOM in terms of phase and duration. Favorable comparisons of the temperature and currents at T2 provide confidence in ELCOM simulations of other regions.

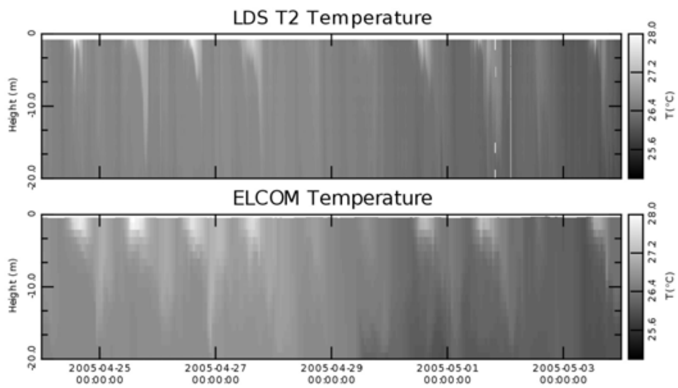
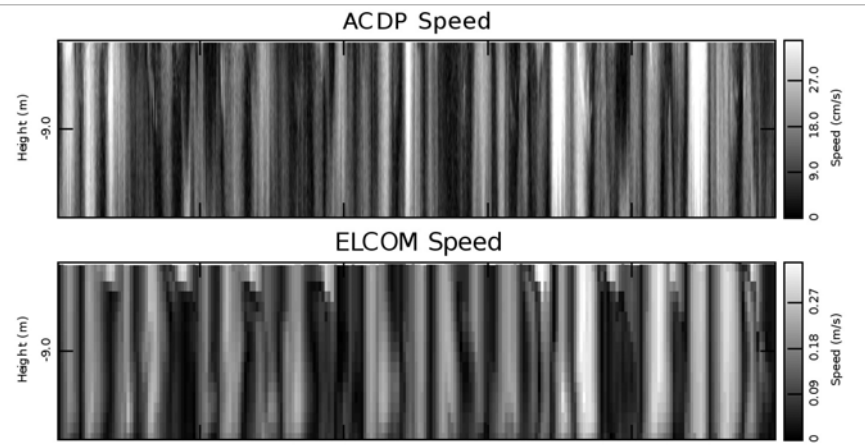


Figure 3. Comparison of observed (upper panel) and modeled (lower panel) temperatures at station T2 from Apr. 24-May 3 2005.



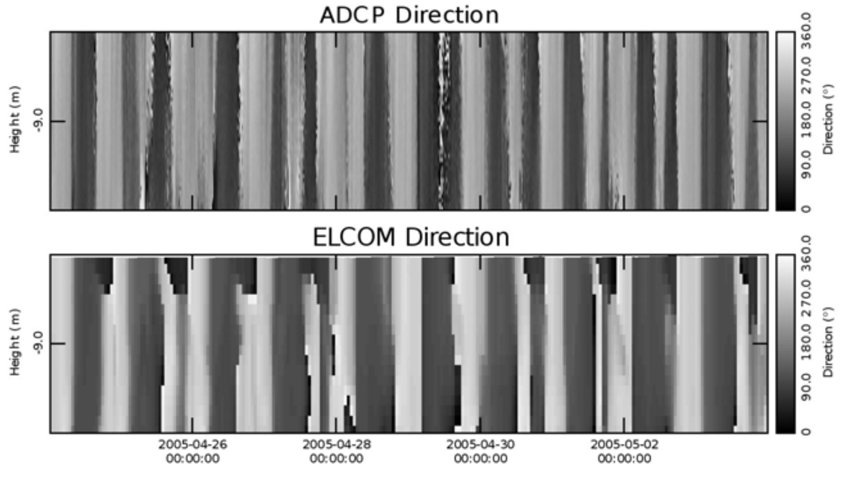


Figure 4. Comparison of observed current speeds (upper panel) and directions (lower middle panel) with modeled current speeds (upper middle panel) and directions (lower panel) at station T2 from Apr. 24-May 3 2005.

Conservative tracers in Asembo Bay, Homa Bay, and the eastern half of Winam Gulf indicate a range of exchange intensity (Figure 5). Over the 10 days the Eastern Gulf was relatively uncoupled from the Western Gulf. The major feature was the development and persistence of a clockwise topographic gyre over the Eastern Gulf, with features similar to patterns from LANDSAT images. In contrast, the tracer in Asembo Bay had a short residence time because of greater flushing with the Western Gulf. Flushing in Homa Bay was intermediate between Asembo Bay and the Eastern Gulf. The city of Homa Bay in the southwestern corner of Homa Bay is located in a region with relatively low flushing rates.

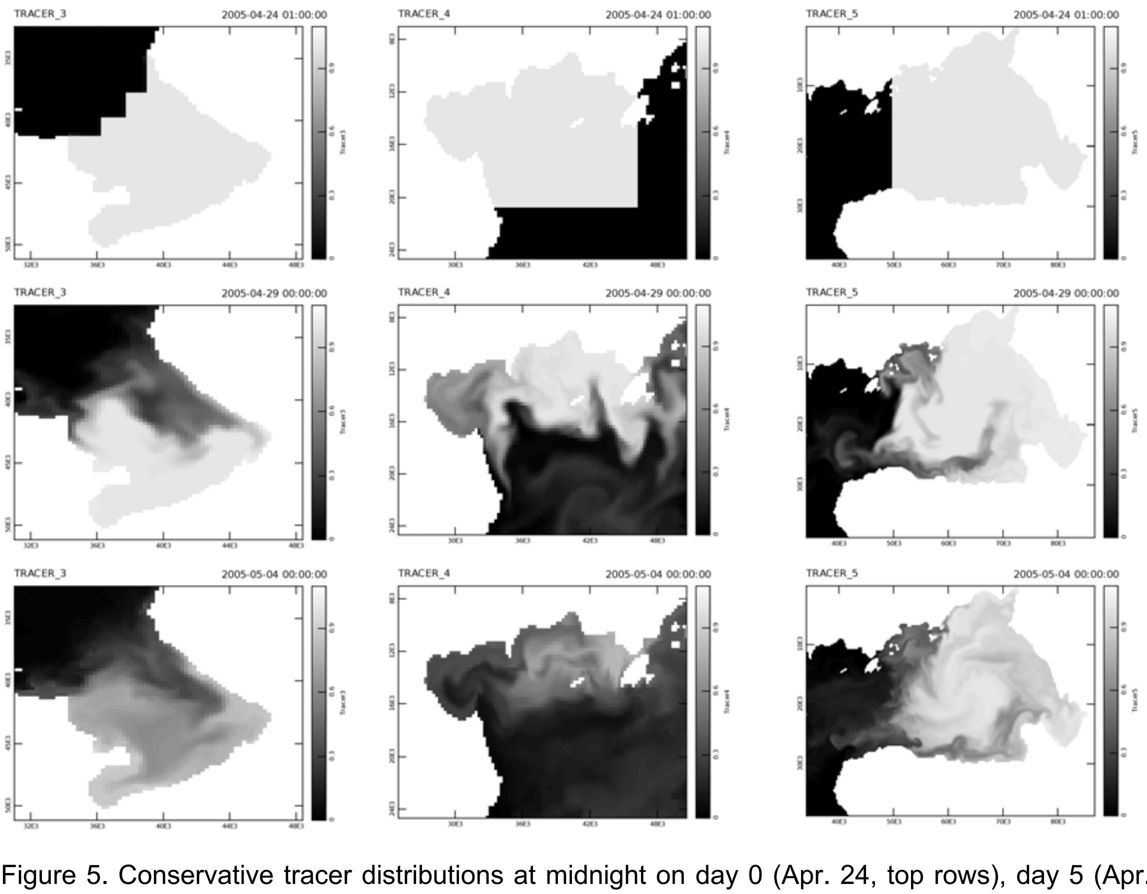


Figure 5. Conservative tracer distributions at midnight on day 0 (Apr. 24, top rows), day 5 (Apr. 29, middle rows) and day 10 (May 4, bottom rows) in Homa Bay (left column), Asembo Bay (middle column), and eastern Winam Gulf (right column).

Exchange estimates were made with ELCOM simulations across the two vertical planes at the Gulf-Channel and Channel-Open Water interfaces (dashed lines in Figure 1). The first field experiment occurred during the wet season and the lake level rose ca. 10 cm. The flushing estimates were between 0.2-0.3% per day over the two periods, which yielded a flushing time of ca. 300-500 days.

Discussion

Even though observed and simulated currents in Rusinga Channel are often in excess of 20 cm s^{-1} (range 0-50 cm s^{-1}, Figure 3), the net exchange between Winam Gulf and the offshore waters is low. These currents are primarily generated by tidal-like variations in surface levels (i.e. barotropic forcing) with 6-12 hour periods (Antenucci *et al.* 2005). Hence, even though the transport distance of the Channel water masses can be in excess of 3-5 km during an 'incoming tide', a similar length scale of movement occurs during the 'outgoing tide', which results in relatively low net transport.

The flushing time scale of Winam Gulf is on the order of 1-1.5 years over the conditions of our two field campaigns. This provides ample duration for fate processes to buffer the effects of external inputs from point sources (rivers and cities) and diffuse shoreline sources. Our current modeling investigations are aimed to improve understanding of the fate of nutrient loading from these point sources.

Unfortunately, our sampling did not coincide with periods of high inflows, which may result in considerably higher flushing and short-circuiting of riverine loads more directly to the Main Lake. A recent high flow event from 2003 with average discharges of 180 m^3 s^{-1} for the Sondu River and 400 m^3 s^{-1} for the Nyondo River was simulated to ascertain short-circuiting. The tracer of the flood inflows was confined primarily to the Eastern Gulf by the topographic gyre (not shown). These results highlight that even during high discharge periods, the region that is most affected by poor water quality from the rivers is the Eastern Gulf.

The tracer simulations (Figure 5) indicate that there may be some scope for catchment and point source management to improve local water quality in the Eastern Gulf and Homa Bay. Catchment management practices to reduce nutrient, pollutant and sediment loading will likely improve the mean water quality of the Eastern Gulf after such flood events. Recent estimates indicate nearly 50% of the BOD load into Winam Gulf is from Kisumu, hence upgrades to wastewater treatment facilities would greatly improve local water quality (Scheren *et al.* 2000). Similarly, the water quality in the region of the city of Homa Bay would also benefit from improved wastewater treatment given the low flushing in the southeast corner of the bay (Figure 5) at the location of this urban center.

Recent bio-available phosphorus surveys consistently record higher levels in the nearby offshore waters than in the Gulf, which has been attributed to adsorption onto the inorganic particles in the turbid Eastern Gulf (Gikuma-Njuru & Hecky 2005). Nutrient sampling during our two field campaigns also revealed these patterns (Gikuma-Njuru, unpubl. data). Similarly, the inshore waters of Uganda near the Nile outflow are comprised mainly of particulate organic phosphorus with low levels of bio-available phosphorus, which suggests nearly all of the phosphorus is converted to organic forms in these inshore regions (Lehman & Branstrator 1994). We hypothesize that the adsorption and biological uptake of bio-available phosphorus in the eastern Gulf occurs because of high levels of turbidity and blue-green biomass (Figure 2). Additionally, the high biomass of cryptophytes (not shown) and diatoms profiled in Rusinga Channel during both field studies (Figure 2, J. Imberger & J. Romero, unpubl. data) suggests that sufficient bio-available phosphorus is transported either from the offshore waters or the particulate bound component from the Gulf becomes bio-available. There is no doubt that bio-available phosphorus concentrations have increased offshore recently (Hecky 1993) relative to the 1960s (Talling 1966).

In short, the role of Winam Gulf as a source or a sink of phosphorus remains uncertain. Currently, we are using ELCOM coupled to the ecological model, CAEDYM (Romero & Imberger 2003), to improve understanding of the fate and transport of particle-bound phosphorus from the Gulf and bio-available phosphorus from the Main Lake. These simulations will provide the first estimates of the fate and transport of nutrients and sediments throughout Winam Gulf and Rusinga Channel so that further refinements can be made to research priorities and to management options of the region.

Acknowledgements

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References

Antenucci J. P., Romero J. R., Imberger J., Ewing T., Khisa P. & Njuguna H., 2005. The Physical Limnology of Winam Gulf and Rusinga Channel of Lake Victoria during Apr.-May and Aug. of 2005. *Proceedings on the 11th World Lakes Conference*, 31 Oct.-4 Nov. 2005, Nairobi, Kenya.

Gikuma-Njuru P. & Hecky R. E., 2005. Nutrient concentrations in Nyanza Gulf, Lake Victoria, Kenya: light limits algal demand and abundance. *Hydrobiologia* 534, 131-140.

Hecky R. E., 1993. The eutrophication of Lake Victoria. *Verh. int. Ver. Limnol.* 25, 39–48.

Hecky R. E. & Bugenyi F. W. B., 1992. Hydrology and chemistry of the African Great Lakes and water quality issues: Problems and solutions. *Mitt. int. Ver. theor. angew. Limnol.* 23, 45–54.

Hecky R. E., Bugenyi F. W. B., Ochumba P., Talling J. F., Mugidde R., Gophen M. & Kaufman L., 1994. Deoxygenation of the deep water of Lake Victoria, East Africa. *Limnol. Oceanogr.* 39, 1476–1481.

Hodges B. R., Imberger J., Saggio A. & Winters K.B., 2000. Modeling basin-scale internal waves in a stratified lake. *Limnol. Oceanogr.* 45, 1603-1620.

Kaufman L., 1992. Catastrophic change in species-rich freshwater ecosystems: the lessons of Lake Victoria. *BioScience* 42, 846–858.

Lehman J. T. & Branstrator D. K., 1994. Nutrient dynamics and turnover rates of phosphate and sulfate in Lake Victoria, East Africa. *Limnol. Oceanogr.* 39, 227-233.

Mugidde R., 1993. The increase in phytoplankton primary productivity and biomass in Lake Victoria (Uganda). *Ver. int. Ver. Limnol.* 25, 846–849.

Ochumba P. B. O. & Kibaara D. I., 1989. Observations on bluegreen algal blooms in the open waters of Lake Victoria, Kenya. *Afr. J. Ecol.* 27, 23–34.

Okungu J. & Opanga P., 2004. *Pollution loading into Lake Victoria from the Kenya Catchment*. Available from URL:http://www.lvemp.org/L_Publications/kenya/Okunguj.pdf. Accessed 15 August 2004.

Nece R. E. & Richey E. P., 1975. Application of physical tidal models in harbor and marina design. *Proceedings of the Symposium on Modelling Techniques*; 10–15 Aug. 1975, San Francisco, California.

Reinthal P. N. & Kling G. W., 1994. Exotic species, trophic interactions and ecosystem dynamics: a case study of Lake Victoria. *In:* D. Stouder, K. Fresh & R. Feller eds. *Theory and Application in Fish Feeding Ecology* pp. 295-313. Univ. of South Carolina Press, Columbia.

Romero J.R. & Imberger J., 2003 Effect of a flood underflow on reservoir water quality: Data and three-dimensional modelling *Arch. Hydrobiol*. 157, 1-25.

Scheren P. A. G. M., Zanting H. A. & Lemmens A. M. C., 2000 Estimation of water pollution sources in Lake Victoria, East Africa: application and elaboration of the rapid assessment methodology. *J. Environ. Manag*. 58, 235-248.

Talling J. F., 1966. The annual cycle of stratification and phytoplankton growth in Lake Victoria (East Africa). *Int. Rev. ges. Hydrobiol.* 51, 545–621.

Dynamic Simulation on Nutrient exchange and release in Lake Water-sediment Interface

Xue Lianqing, Hao zhenchun, Lu xiwu

State Key Laboratory of Hydrology-Water Resources and Hydraulic Engineering; College of Water Resources and Environment, Hohai University, Xikang road 1#; Nanjing, 210098, P.R.China

E-mail: lqxue@hhu.edu.cn

Abstract

The sediment distributed and insolated under lake was collected for experiments. The nutrient layer distribution conditions of sampled sediment and its physical and chemical characteristics were analyzed to simulate and assess the influence degree to lake water quality. Based on the dynamic water exchanging experiments the nutrient release process in sediment and influence mechanism to substance exchanging on water-sediment interface were studied, and the correlation between the changing content of total phosphors and total nitrogen in sediemnt and covered water was analyzed for setting up simulation model. At the same time the influence degree is explained in detail. Through the experiments the following results were concluded that even if clean water without nutrient contents was used for water exchangement in order to decrease pollution or prevent eutrophication, however owing to the vertical nutrient distribution in lake sediment will lead to the increasing release amount greatly especially when the organic nutrient contaned in sediment turns into inorganic status because of insolation. Besides the release process of total phosphate (TP)and total nitrogen (TN) were modeled and each nutrient's exchanging equation at interface caused by covered water nutrient concentration changing was set up. According to the simulating prediction TP and TN content of cover water will also sustain a steady higher level in a long period, then tend to a level. The nutrient release amount of sediment is not noly affected by the covered water concentration but also connects with accumulative time. The experiments provide the fundamental theoretical and practical basis for taking ecological restoration project. And research is helpful to prevent or cure with lake eutrophication.

Key words: phosphate & nitrogen release; sediment experiments; simulation model

Introduction

With the rapid lake eutrophication in the city, some hydraulic engineering or ecological measures are operated to prevent or decrease the degree of eutrophication. In the paper, according to the hydraulic engineering influence made on the lake, the nutrient release process was analyzed and simulated through a great many of nutrient monitoring experiments. Aiming at the sediment dredging up and water renewal engineering the nutrient changing process during the whole project is simulated. The nutrients including phosphate, nitrate, TP and TN content of the insolated sediments and nutrient content and dynamic release curve of lake water after water renewal procedure were studied (Jansson, 1994). The research indicate that a great deal inner inorganic nutrient released into water rapidly owing to the organic nutrient degradation at the early water renewal period and the nutrient concentration would increase greatly and strengthen the lake eutrophication. If at this period a ecological restoration project and hydraulic engineering or scheme were organized and operated in a reasonable way the lake eutrophication would be controlled and managed in an effective way. The research results provide a technical support for put some ecological restoration measures in practice.

Materials and methods

Sediment sampling

Based on the horizontal and vertical distribution characteristics of lake sediments, there are five sampling locations selected including 2 locations C and E in central area, and 3 locations A, B and D in the lakeside (Wang & Sun, 2002). For each sampling location the sediment with different depth is collected for experiment (Yan, 1998). There are total four samples at different layer (0~5cm, 5~10cm, 30~40cm, 50~60cm) for each location point. The whole sediment sampling experiment is finished after the lake drying up for one year. TP, inorganic phosphorus, TN content of the collected sediment samples is determined separately by HClO4-H2SO4-Mo-Sb-Vc colorimetry, Olsen NaHCO3 lixiviation-Mo-Sb-Vc-colorimetry, KHSO5-Mo-heteropoly blue method. The sediment sampling location in the lake is shown as the Figure 1.

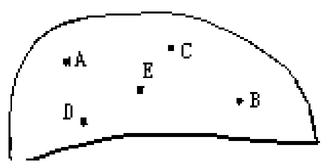


Figure 1 Sampling location Distribution of sediments.

Nutrient distribution characteristics of sediments

Taking the selected sediments in different layer for chemical and physical test, it is founded that the upper layer 0~10 cm sediment shows gray to dust color with loosen homogenized content, and the middle and the lower layer 30~60cm sediment transit from dust color to black color. For sampling location (C and E) in central area of lake, the average TP and TN concentration of each layer is low, separately being 1/2~1/5of monitoring locations (A-B-D) surrounding the lake. The distribution lowest average TP concentration of each layer is 254mg/kg at location E, and the highest is 1387mg/kg in location D. The TP concentration of surface layer at location E attains 341mg/kg, taking as 1/8 of the highest TP concentration of surface layer with 2810mg/kg in location B. Owing to the exterior pollution burden entering into lake and enriching at the inner surroundings of lake, the TP、TN concentration surrounding the lake is greater than that in the local area evidently. Taken the collected sediments as analysis, it is found that the organic matter components decreased 43.8~59.1% after insolating caused by dredging engineering operation compared with the former years monitoring results. Thus the inner inorganic nutrient matter in sediments increased greatly because of the organic matter's inorganication and released rapidly then to influence water quality and arouse the lake eutrophication when implementing water renewal and ecological restoration projects. So to analyze the distribution characteristic of TP, TN and other nutrient concentration in sediments is necessary to simulate the engineering results, the test and monitoring analysis results are shown as the following Fig.2-6. In the figures, it is showed that TP content declines gradually at last with the increasing sediment depth. And to the sediment layer (0~5cm) of location B, TP content is the highest with 2810mg/Kg and with 391mg/kg in layer (50~60cm). For the layer 50~60cm of location A, the TP content is 34.2mg/kg taking 1/12 of B. However the vertical distribution characteristic of TN content was not presented obviously. TN content in each sediment layer varies from 1500mg/kg to 1825mg/kg and inorganic nitrogen content increase with the depth to some degree.

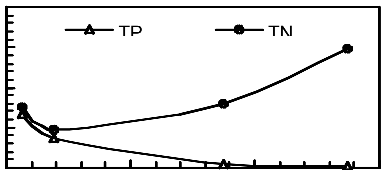


Fig.2 Distribution of TN\TP at site A

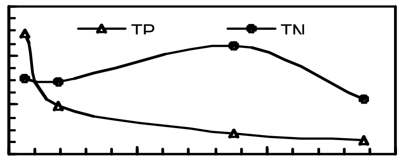


Fig.3 Distribution of TN\TP at site C

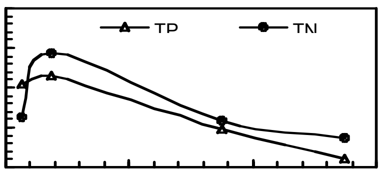


Fig.4 Distribution of TN\TP at site B

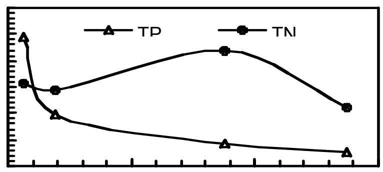


Fig.5 Distribution of TN\TP at site D

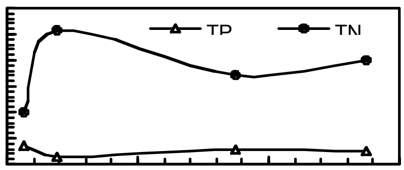


Fig.6 Distribution of TN\TP at site E

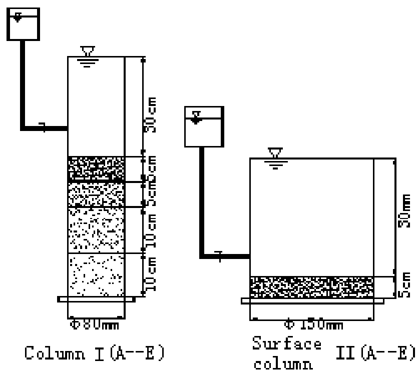


Fig 7 Dynamic Water-exchange Experiment Equipment

Dynamic water renewal experiment simulation

In order to analyze and simulate the nutrient release influence to water caused by sediment and implementing results of water renewal engineering on lake, according to factual engineering circumstance the water renewal experiment is taken. The experiment equipments is shown as Fig.7. In the experiment TP and TN of the covered water over sediment is separately tested by using (NH4)2MO4O13·2H2O spectrophotometry method and Potassium Peraulfate (JENWAY 6505) Spectrophotometry. In Fig.2, the collected sediments with different depth are backfilled into column of 80mm - 600mm in natural turn (5-5-10-10cm) for experiment, the above water depth is remained at 30cm. Meanwhile the other column vessel of 150mm - 350mm is backfilled with top-sediment 0~5cm for experiment and also covered by water with 30cm depth. By exchanging the 10% of covered water each time, the lake nutrient concentration is tested to analyze and simulate the purifying results of engineering.

Simulation experiment

Phosphate release in sediment

During the dynamic water renewal process, the laboratory air temperature varies from 24~27^{o}C every day and decrease to 16~17^{o}C so as to suit with the changing air temperature outside TP and phosphate release process in column and surface layer sediment were simulated as the Fig.8-9. However in the paper only the monitoring results at location B and E was displayed.

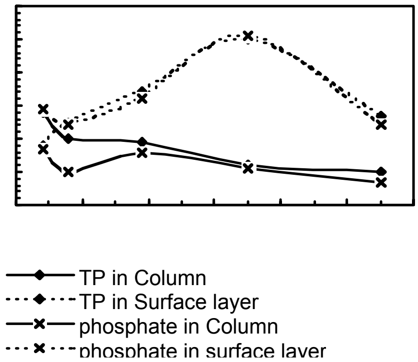
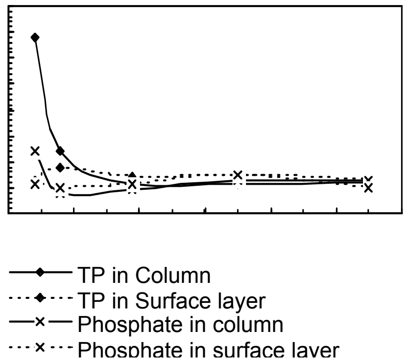


Fig.8 Phosphorus release curve at site B



According to the release experiment, it is analyzed that in the early period the release amount of TP into water in sediment column is more and reach the maximum in the obvious fourth day, the analysis is based on the every four day owing to the obvious characteristi(Lin.& Tao, 1997). However the release amount of TP then decrease gradually with the maximum release at location E and at this time the TP concentration of the above water reaches 0.69mg/L. For the location A, it is typical that the TP concentration of above water reach the maximum 0.59mg/L at the eighth day then decrease gradually. For all the monitoring examples TP concentration of the above water is remained not less than 0.1mg/L after 50days, so the lake water presents eutrophicatio. From the monitoring results of column the released phosphate amount into water decrease steadly form about 0.2mg/L to 0.1mg/L. It is because the released TP in surface layer of sediment presents the obvious rule and the water reneawl make the sediment disordered then promoted the material exchanging and influenced phosphate release. When the accumulation effect making TP and phosphate concentration increasing reaches the equivalence, then the disordering influence decreases gradually and the nutrient concentration will decrease greatly. Meanwhile the TP and phosphate concentration of the above water in surface layer sediment is more than relative concentration of column in the middle and upper period. All these indicate that only the surface layer sediment will produce greater influence to phosphate release when exchanging water.

Nutrient release in sediment

From the release experiments the TN and Ammoniate-Nitrogen are released in the approximate rule in sediment column and surface layer. The monitoring results are shown as the Fig.10-11. Compared with phosphate release, the nitrogen release rate is much lower and reach the maximum through 8-20 days' accumulation then decreased thereafter (Fan & Qin, 1998).

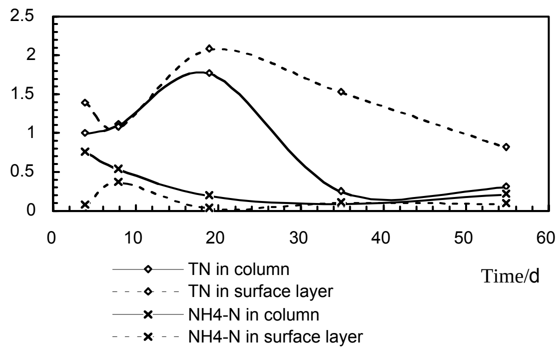


Figure 10. Nitrogen release Curve at site B sediment

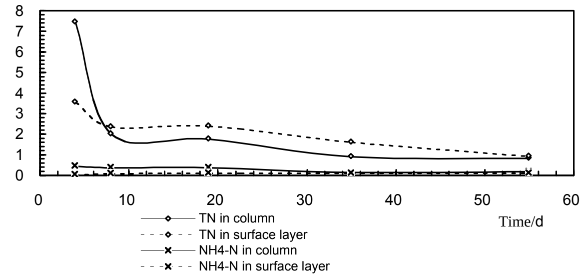


Figure 11. Nitrogen release Curve at site E.

Since the fourth day, TN and amonium-nitrogen concentration at location A, B and D remains between 1.0~2.0mg/L, thereafter increase at 2.0~3.0mg/L. After 50 days they also fall to 0.5mg/L. However for the typical E, the TN and amonium-nitrogen concentration will remains at about 1.0mg/L becuause of the loose and arenaceous structure at location E. For the surface layer sediment, the released TN and amonium-nitrogen changes steadily. They present the similar rule with TP and phosphate release procedure.

Set up source simulation model

There are two methods on determing nutrient exchanging amount in lake water-sediment interface. One is practical mensuration and the other is indoor simulating method. In the paper the above two methods is combined to simulate the dynamic nutrient release and analyze or predict the simulation result. The dilute method by capacity increase is used as the foundation on setting up model. The water exchanging period is taken as 10 days and it is assumed the water body is mixed evenly. The exchanged water used is clean water without nutrient. So the 10% of the total water was limmited and there is 10% is overflowed at the same time. The simulation experiment was finished seperately in column and surface layer of sediment. According to the experiemnt principle the following dynamic water exchanging Equa.1 was set up.

$$C_{t+1} = \frac{VC_t \times 0.9 + q_{t+1}}{V} \qquad (1)$$

Where, C_{t+1} (mg/L) is TP or TN concntration of the above water in time $t+1$, and q_{t+1}(mg) is the total released nutrient amount of sediment from t to $t+1$. V (*L*)is the covered water volume of monitoring sediment column.

According to the Equa.1 and the series of monitoring results, the daily TP and TN release amount and total release amount in sediment when taking the dynamic water exchanging experiment could be obtained by difference method. Therefore the relation between TP/TN concentration and the relative release amount of sediment in covered water is determined. The following Fig.12-13 indicate the relation between TP/TN concentration in the covered water and the TP/TN release amount in sediment will show the direct proportion. It could be conclued that the nutrient concentration of the above water will directly influence the deposited nutrient release of sediemnt especially to the surface layer sediment. So disposing surface layer sediment and diluting the covered water are important and necessary for lake eutrophication prevention.

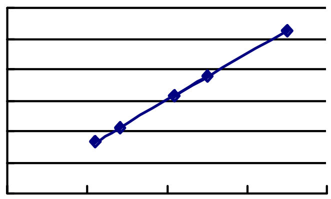


Figure 12. Relation between TP in covered waterand daily sediment release amount.

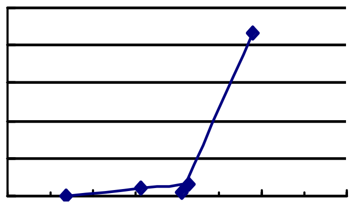
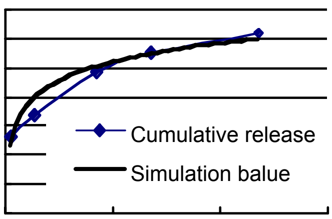
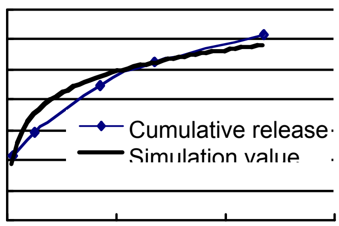


Fig.13 Relation between TN in covered waterand daily sediment release amount.

In order to analyze the contribution degree of the total accumulated released TP/TN amount to the quality of water body during the dynamic water-exchanging experiment. The accumulated nutrient release amount in sediment is simulated and the release curve with time and model is simulated as the following Fig.14-15 and Equa.2-3 separately. The simulation error on TP and TN is 0.81 and 0.85 expressed by square difference.





TP release Equation $y = 0.234\ln(t) + 0.365$ (2)

TN release Equation $y = 2.243\ln(t) + 2.987$ (3)

where *t* is time variable with day as unit and *y* is the accumulated release amount mg of TP or TN.

Results and discussion

Through the practical lake eutrophication engineering, the TP and TN concentration is simulated at the same time the background of sediment and practical water quality is monitored as analysis form Sept.14. The calculated TP/TN concentration in the covered water by experiment and practical monitoring after engineering operation are expressed as E and F seperately in the following Table 1.

Tab.1 Comparision Analysis

Time (d)	Sept. 24		Sept.28		Oct. 9		Nov. 4	
	E	F	E	F	E	F	E	F
TP (mg/L)	0.35	0.32	0.27	0.30	0.11	0.14	0.11	0.15
TN (mg/L)	2.88	2.78	2.05	2.47	1.79	1.87	1.70	1.62

Based on the simulation results of model 1-3, it could be concluded that the accumulative TP and TN release amuout will increase in a much longer time owing to the vertical inner nutrient distribution in sediment. Therefore the vertical nutrient distribution characteristic and simulation analysis provide a feasible and reasonable foundation to implement sludge elimation engineering. And if 40cm of the sediment is elimiated then the 90% of TP will be removed. Besides the nutrient release amount of sediment not noly is affected by the covered water concentration but also connects with accumulative time dispite of the dynamic exchanging water is clean. From the experiemnt simulation, after 50 days the TP and TN concentration of covered water will still remain over 0.1mg/L and 1mg/L seperately. The lake still remains eutrophication level. So the other ecological engineering should be brought into effect so as to prevent or alleviate lake eutrophication even if the sludge engineering is carried out.

Acknowledgements

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References

Fan C.X., Qin B.Q., 1998. The substance exchange in water-sediment interface of Taihu lake. *Lake Science*. 10, 72-75.

Jansson M., 1994. Denitrification and nitrogen retention in a farmland stream in southern Sweden .*AMBIO*. 23,326-331.

Lu M., Zhang Y.G., 2003. The vertical changing characteristics of nutrient and metel element in northern lake Tai and its influence factors analysis. *Lake Science*. 15(3), 213-217.

Lin B., Tao P., 1997. Research on dynamic organic material release in soil and deposit.[J]. *J. Environmental Science*. 17, 8-11.

Wang J.Q., Sun Y. M., 2002. Release simulation modeling of sediment in Chaohu lake. J. Environmental Science.22(6),738-742.

Yan W., 1998 Nutrient retention by multipond systems: mechanisms for the control of nonpoint source pollution. *J Environ Qual*. 27,1009-1017.

The effect of the Mbita Causeway on water currents in the region of Rusinga Channel, Winam Gulf, Lake Victoria: a 3D modelling study with ELCOM

Khisa Patrick [1], Jose R. Romero[2], Jorg Imberger[2] , Tom Ewing[2] , Jason Antenucci[2], Henry Njuguna[1] and John Okungu[1]

[1] LVEMP Coordination Office, Re-Insurance Plaza, 2nd Floor, Kisumu 40100, Kenya
Email: pkhisa@yahoo.com
[2] Centre for Water Research, Univ. of Western Australia, Crawley, WA, 6009, Australia

Abstract

The Mbita Causeway was constructed in the early 1980's to link Rusinga Island to the mainland to facilitate the transport of people, goods, and services to the island. One of the effects of the Causeway was to permanently block a secondary connection between Rusinga Channel and the offshore waters of Lake Victoria. This secondary channel may have served as a substantial pathway of exchange between Winam Gulf and Lake Victoria prior to the Causeway. There has been considerable scientific and political debate as to whether decreased flushing within the two bays on either side of the Causeway has occurred and led to poorer lake water quality since the construction of the barrier.

In Apr.-May 2005 an intensive field investigation was undertaken to determine the exchange dynamics between the northeastern offshore waters of Lake Victoria and Winam Gulf through Rusinga Channel. One of the outcomes of this scientific study was the validation of ELCOM (Estuary, Lake and Coastal Ocean Model), a three-dimensional (3D) hydrodynamics model, over Rusinga Channel.

In this applied modeling study, ELCOM simulations with and without the Mbita Causeway were run. The simulated exchange through the channel without the Causeway was quantified. Simulated differences in the flushing within the two bays with and without the Causeway were evaluated through numerical tracer studies to infer the possible effects on water quality.

Key words: Hydrodynamics, Modeling, Scenarios

Introduction

Engineering works across water bodies often affect water circulation patterns and thereby water quality. One example is the construction of the Mbita Causeway in 1985 that linked the Mbita peninsula to Rusinga Island. This Causeway is located 70 km east of Kisumu, Kenya (Fig. 1). The Causeway essentially blocked a natural channel with dimensions of 250 m in length and 10 m in depth through earth filling and construction of a road to Rusinga Island.

Clearly, the construction of the Causeway resulted in the loss of an alternative exchange pathway between Rusinga Channel and the offshore waters of Lake Victoria. Numerous hypotheses on the effect of the loss of this exchange pathway have been suggested by scientists, policy makers, local community members, and special interest groups. For example, some speculate that the blockage of the Mbita Channel has affected the ecosystem around Mbita and Rusinga area by blocking a fish migration route. One of the effects we consider in this paper is the loss of flushing and water exchange in the regions near the Causeway.

Until recently, the water circulation patterns in the region of Rusinga Channel were poorly understood, but several papers presented at this conference of recent field (Antenucci *et al.,* 2005) and numerical modeling (Romero *et al.,* 2005a) studies have increased understanding of the region's hydrodynamics. During the first of these studies field data was collected over 12 days in Apr.-May 2005 with moored automated instruments at two stations and free-falling instruments that profiled many stations throughout Rusinga Channel. Comparison of these hydrodynamic data with the three-dimensional Estuary, Lake and Coastal Ocean Model (ELCOM) was good (Romero *et al.,* 2005a). In this paper we extend the application of ELCOM to assess the water circulation and exchange patterns prior to the construction of the Causeway.

Methods

Simulations were conducted with the three-dimensional hydrodynamic model, the Estuary and Lake, and Coastal Ocean Model, ELCOM (Hodges *et al.,* 2000). A uniform numerical grid was used. The computational domain was approximately 100km long and the smallest feature to be resolved is the 250m wide Mbita Causeway. On removal of the Causeway, the Mbita Pathway was aligned with the computational grid and was equivalent to one horizontal grid cell of width 250m and 10m adjacent vertical grid cells. Simulations were started from rest, with horizontal free surface and isopycnals. Free slip boundaries were used to model sidewall (land) boundaries with a drag parameterization at the bottom boundaries.

Application of the model ELCOM to compare conditions prior to the construction of the Causeway relied on data from the 12-day field study of Apr.-May 2005. The simulation to validate the model over the field study was the current conditions with the Causeway. The Causeway was removed and replaced with a channel for the scenario prior to 1985. Refer to Antenucci *et al.,* (2005) and Romero *et al.,* (2005a) for details of the field and model methods.

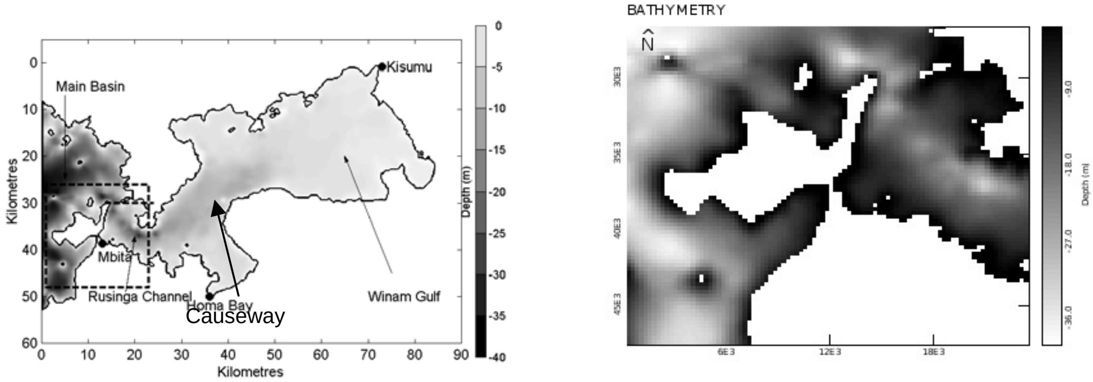

Figure 1. Map of general region (left panel) illustrating northeastern offshore waters of Lake Victoria, Rusinga Channel and Winam Gulf along with the major cities (circles) and the focus region of this study (dotted rectangle). The dotted line from the eastern tip of Rusinga Channel directly to the eastern shoreline at the constriction of Rusinga Channel marks the initialization boundary of conservative Tracer 1. The right panel shows the focus region of this numerical study with the location of the Causeway.

The hourly water volume flux through the 'former' Mbita Channel (now the Causeway) was calculated by applying the method used by Laval *et al.,* (2003) for Lake Maracaibo and the Caribbean Sea. The water volume flux (Q_v) was defined as;

$$Q_v = \int_A V dA \qquad (1)$$

where A is the cross-sectional area and V is the velocity along the cross-section. Q_v is computed discretely for each hour from simulated data as;

$$Q_v = \sum_i v_i \Delta A_i \qquad (2)$$

where *i* is the *ith* depth along the cross section, Ai is the north-south unit area of the face and V is the eastward component of velocity along Mbita Pathway. The eastward velocity (inflow) was considered positive while the westward velocity (outflow) was taken as negative. The total cross-sectional area of Mbita Channel was taken as 9 m depth by 250 m width. The cumulative water volume flux from hourly model output was computed over the 12 day simulation.

Results

The simulated lake level at station T2 reproduced reasonably well the observations recorded by the ADCP (Fig. 2). This lake level validation along with the validation of water temperatures and currents at T2 (see Romero et al., 2005a) provided confidence that ELCOM simulates the Rusinga Channel dynamics reasonably well. Next a comparison of simulations with and without the Causeway is considered.

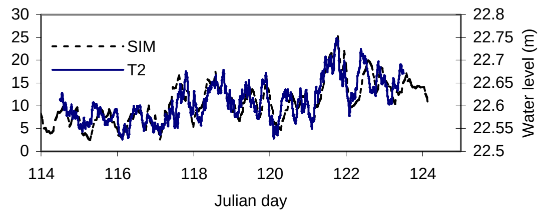

Figure 2. Comparison of ADCP observations (line) and simulation (dashed) of lake level at stations T2.

The simulated currents with and without the Mbita Causeway were similar (Fig. 3). This suggests that the effect of the construction of the Causeway had a minimal affect on the regional hydrodynamics, exchange, and transport of water masses between the offshore and Winam Gulf through Rusinga Channel.

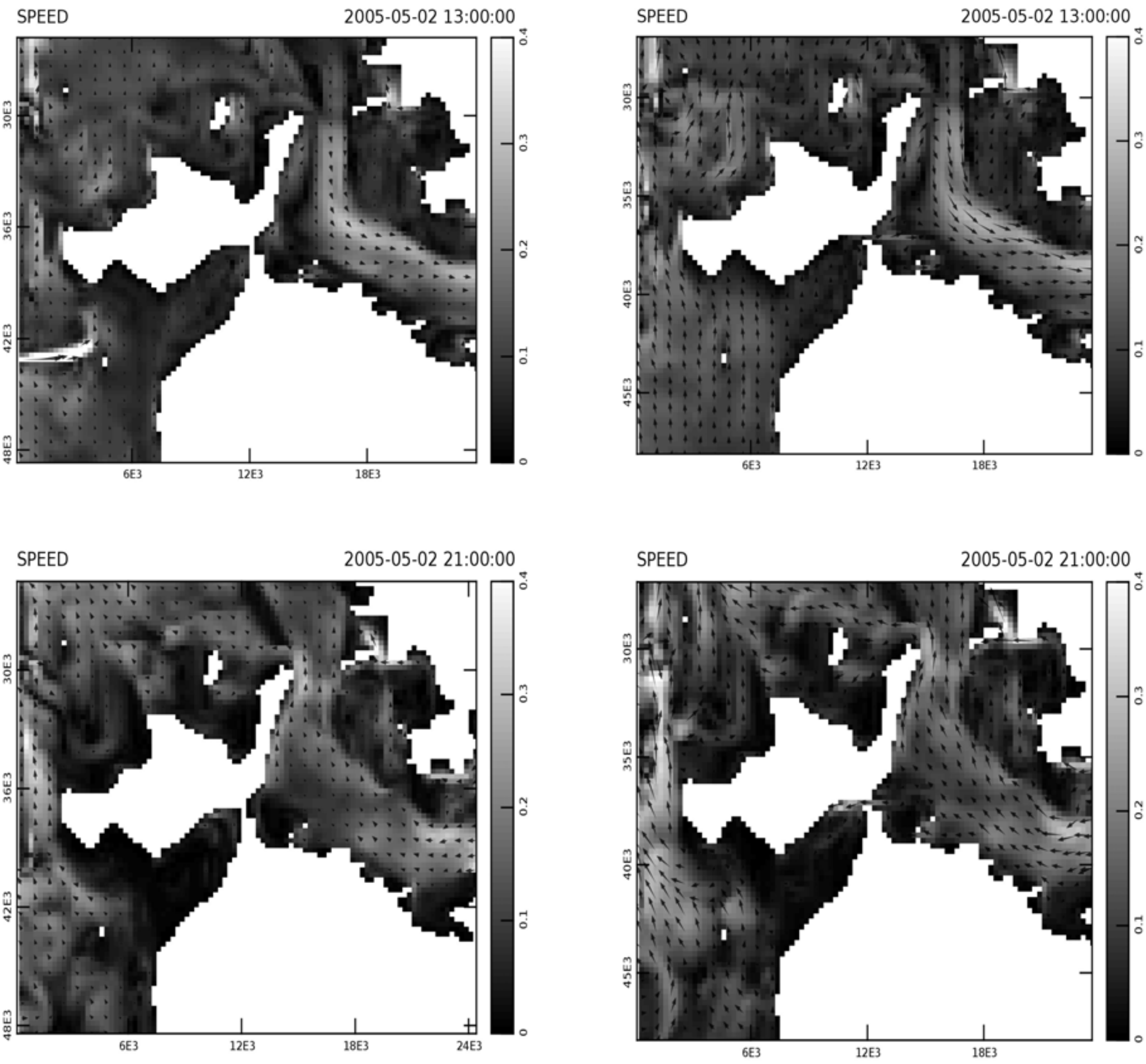

Figure 3. Water current directions (vectors) and speed (shading) in the region of Mbita on May 2 during the afternoon at 1300 when currents are into Winam Gulf (top panels) and in the evening at 2100 when currents are towards the open waters. Left panels are with the Causeway and right panels are without the Causeway. Vectors represent current direction and not the speed.

Conservative tracers were used in the simulations to visualize the differences in transport between the Causeway and Channel scenarios (Fig. 4). Tracer 3 was added to inputs from the open boundary on the western margin of the model domain. Tracer 1 was initialized throughout Rusinga Channel and Winam Gulf as illustrated in Fig. 1. Lower Tracer 3 concentrations were simulated in the region of the small island at the interface of the bay to the west of Mbita and the open waters of Lake Victoria without the Causeway. This suggests that greater flushing of this bay may have occurred without the Causeway. In contrast, Tracer 1 patterns in Rusinga Channel were nearly the same for the two simulations throughout Rusinga Channel and the offshore waters to the north. This again suggests that either condition (i.e. Causeway or Channel) did not have much influence on the bulk exchange and transport patterns between Rusinga Channel and the offshore waters to the north.

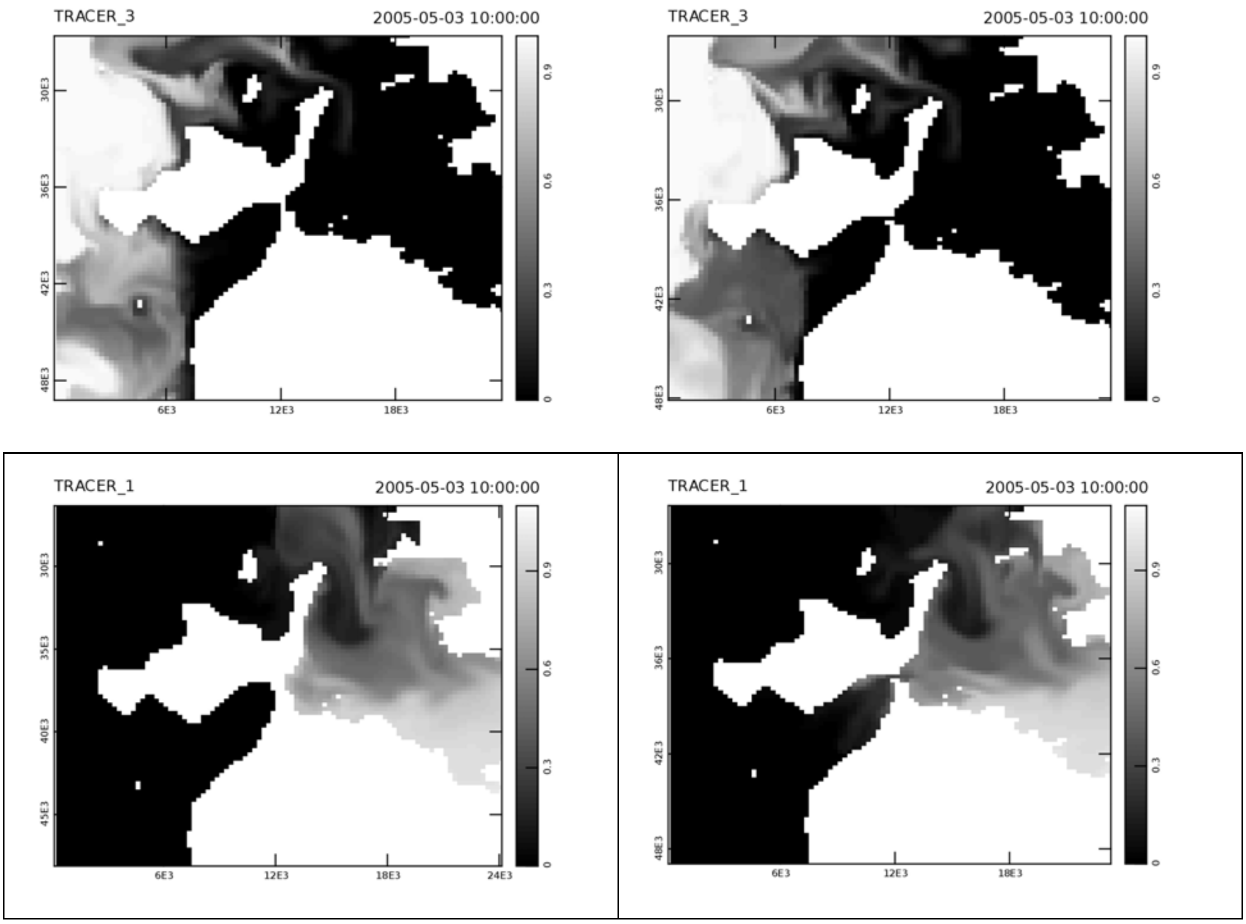

Figure 4. Tracer concentrations in the Mbita region on the late morning of May 3 for the simulations with (left panel) and without (right panel) the Causeway.

The simulation with the Mbita Channel resulted in net transport of Tracer1 through the channel and into the bay to the west of Mbita (Fig. 4, lower right panel). The cumulative volumetric flux of water through the channel indicated net westward transport through the channel (Fig. 5). The net westward flux through the Mbita Channel occurred primarily after the 4th day of the simulation. Over the final 8 days of the simulation ca. 5 x 10^7 m^3 of water flowed westward. This is equivalent to a daily flux of ca. 0.005 km^3 day. Application of a similar methodology across Rusinga Channel yields a net volumetric flux into Rusinga Channel of 0.17 km^3 day over the simulation (Romero *et al.,* 2005b), approximately a factor of 30 greater than the Mbita Channel flux estimate.

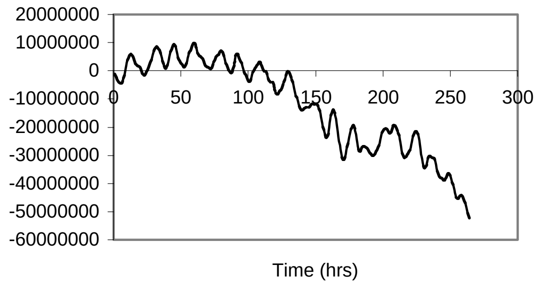

Figure 5. Cumulative volume flux over the 12 day simulation through Mbita Channel.

Discussion

Simulated circulation patterns were only influenced in the immediate vicinity of the Causeway when it was removed. Closer inspection of the simulations revealed high easterly currents in the evenings through the Mbita Channel that persisted approximately 2 km into Rusinga Channel (not shown). Anecdotal evidence from long-time fisherman in the region supports these simulation results (P. Khisa, pers. comm.). For example, fishing accidents in Mbita Channel prior to the Causeway often led to rapid easterly transport of boats and people into Rusinga Channel. Simulated velocities through the Mbita Channel were approximately 50 cm s^{-1} with continued currents into Rusinga Channel of nearly 30 cm s^{-1} (not shown). However, the removal of the Causeway likely did not have a large influence on the overall Rusinga Channel exchange dynamics.

The construction of the Causeway likely had a substantive localized effect on the circulation patterns in the near-region of Mbita and hence water quality in the locale as well. Greater flushing of the bay to the west of Mbita likely occurred with improved water quality relative to the current condition. One of the results of less flushing in the locale of Mbita has been increased sediment deposition in the two bays on either side of the

Causeway, where depths have been reduced by 5 m (LVEMP, unpubl. data). Further, because of the large population density in this region and the associated pollutant loading, the greatest benefit derived from the former Mbita Channel was likely enhanced transport and dilution of these pollutants.

This modeling study represents a first quantitative evaluation of the effect of the construction of the Causeway on the circulation patterns of Rusinga Channel. Again, it is unlikely that the Causeway influenced the general circulation and exchange of Winam Gulf with the open waters of Lake Victoria. However, localized effects on circulation and flushing, and thereby water quality in the Mbita locale likely resulted from its construction. Longer term hydrodynamic modeling and incorporation of biogeochemical models will provide additional insight into the effects of the Causeway on the local water quality.

Acknowledgements

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References

Antenucci J. P., Romero J. R., Imberger J., Ewing T., Khisa P. & Njuguna H., 2005. The Physical Limnology of Winam Gulf and Rusinga Channel of Lake Victoria during Apr.-May and Aug. of 2005. *Proceedings on the 11th World Lakes Conferenc*e, 31 Oct.-4 Nov. 2005, Nairobi, Kenya.

Hodges B. R., Imberger J., Saggio A. & Winters K.B., 2000. Modeling basin-scale internal waves in a stratified lake. *Limnol. Oceanogr.* 45, 1603-1620.

Laval B.E., Imberger J. & Findikakis A.N., 2003. .Mass transport between a semi-enclosed basin and the ocean: Maracaibo system. *J. Geophys. Res. (Oceans)* 108, 3236.

Romero J. R., Imberger J., Antenucci J. P., Ewing T., Khisa P. & Njuguna H., 2005a. Management Implications of the Physical Limnological Studies of Rusinga Channel and Winam Gulf in Lake Victoria, *Proceedings on the 11th World Lakes Conferenc*e, 31 Oct.-4 Nov. 2005, Nairobi, Kenya.

Romero J.R., Feaver S.F., Boroto J., Antenucci J.P. & Imberger J., 2005b. Pilot study of hydraulic conditions over Rusinga Channel and Winam Gulf of Lake Victoria: Report of First Field Study. *Prepared for Kenya Agricultural Research Institute, Lake Victoria Environmental Management Program*. CWR, UWA Reference WP1623JR.

Conservation and management of lakes in urban environment: bioremediation, a new frontier in the control of eutrophication of urban lakes

Kodarkar, M.S.[1] and Sandeep Joshi[2],

[1]Zoology Research Laboratory, P.G. & Research Department of Zoology, Vivek Vardhini College, Hyderabad – 500 095 (Andhra Pradesh)
Email: kodarkar_mohan@yahoo.com (web site : www.lakesmission.net)
[2]Director, Joshi, Sandeep, Shrishti Eco-Research Institute, D-16, Shivsagar Phase I, Manik Baug, Sinahgad Road , Pune – 411 (Maharashtra)

Email: sandeepjoshi-IP@eth.net

Abstract

A number of new ideas are being implemented in the form of decentralized treatment of sewage in the last few years as an alternative to the centralized conventional and capital intensive Sewage Treatment technology operational in the country for almost 100 years; the first Sewage Treatment Plant (STP) was established in India in 1895. The eco-technologies for utilization of sewage as a resource are very significant in the light of the fact that out of enormous quantities of sewage generated in the country hardly 2% is recycled and rest ends up in to water resources without any treatment leading to vicious cycle of pollution (Central Pollution Control Board, 2000).

Key words**:** Urban lakes, Bioremediation, Eco-technologies

Introduction

Water bodies in urban environment are organically enriched by untreated or partly treated domestic sewage and discharges from industries in the catchment. Further, erratic monsoon patterns related with global climatic change, unabated urbanization and an ever increasing volume of water supplied to settlements in the catchment has enormously increased the volume and quality of sewage generated by the cities and maga-cities. As a matter of fact, hydrology of most of the urban lakes is sustained by the sewage. Thus, for effective management of lakes in urban environment, key factor in restoration of water quality is removal of nutrients that will reverse the trend of eutrophication (Kodarkar, 1995).

One of the most effective ways of controlling pollution from sewage is to treat it through Effluent Treatment Plant (ETP) and a well developed technology is available for this purpose. The treatment involves following three steps:

A Primary treatment

In involves settlements of suspended particulate matter in tanks. The Process removes 40 95 % suspended solids.

B Secondary treatment

Essentially it is biological treatment aimed at reduction of BOD. The organic matter in effluents is subjected to microbial degradation. To bring the organic matter in contact with microbes trickling filters is most commonly used. Such a filter consists of a circular tank filled with gravel, crushed stones or slag that provide surfaces for microbial culture. The liquid sludge is sprinkled or spread over the gravel over which a film of microbial culture develops. The sludge is applied intermittently to facilitate maximum degradation. The liquid effluent after passing through biological degradation is collected at the bottom of the tank and carried for further treatment by drainpipes.

The secondary treatment can also be given by activated sludge process. It involves vigorous aeration (48 Hours) of sewage as a result of which it undergoes flocculation. The floc contains actively metabolizing bacteria apart from yeast's, mould and protozoa. Like sprinkler tanks, activated sludge process too effectively reduces BOD.

The liquid sludge after secondary treatment is allowed to stabilize in 2 to 4 feet deep oxidation ponds. The oxygen at this stage is provided by its dissolution from atmosphere and algal photosynthesis.

C Final treatment

The liquid effluents are disinfected to remove microbial pathogens usually by chlorination. Finally, the liquid from settlement tank goes to biological aerator and settled solids are directed to digester . The liquid produced is rich in nutrients and can be effectively used for growing algae and fishes. The algae, in turn, can be used as a poultry feed. The liquid component is finally used to irrigate agricultural land.

As a long-term measure sewage treatment could be a very effective measure for the control of organic pollution in the case of grossly polluted lakes. Nevertheless, in the case of urban lake ecosystems, recovery of water quality is delayed due to non point source of pollution and release of nutrients from sediment. It is essential to understand that sewage is a resource and an integrated approach in waste management needs to be given priority. For example, sewage is very rich source of basic elements N, P, and K and could be a cheaper substitute to synthetic fertilizers in agriculture and aquaculture. It can also be utilized for generation of bio-gas and electricity.

Bioremediation

A number of new ideas are being implemented in the form of decentralized treatment of sewage in the last few years as an alternative to the centralized conventional and capital intensive Sewage Treatment technology operational in the country for almost 100 years; the first Sewage Treatment Plant (STP) was established in India in 1895. The new ideas and technologies for utilization of sewage as a resource are very significant on the background of the fact that out of enormous quantities of sewage hardly 2% is cycled and rest ultimately pollutes the precious aquatic resources of the country. In recent years as an alterative to conventional STP technology, efforts are being made to develop technologies based on ecological principles and holistic ecosystem approach. In this philosophy waste is looked upon as parked resource and emphasis is on its effective recycling by employing eco-technologies.

Bioactive soil treatment for sewage

The Bio-active soil treatment technology is based on the natural ability of soil biota to digest and assimilate the organic matter in the raw sewage thus enabling its effective recycling in controlled manner (Joshi, Sandeep and Joshi, Sayali, 2001). The bioactive soil filter stimulates natural bio-lithosphere condition and acts as natural digester for different biochemical reactions favourable for recycling. Similar in design to simple sand filter the bioactive soil filter is built by using biologically active soil as seed with local soil. The floor of tank is made of impervious layer of cement concrete to avoid infiltration and percolation losses. The bioactive soil is developed through the active assemblage of soil organism like bacteria, fungi, protozoa, arthropods, earthworms and molluscs. Further, algal and macrophytes growth on this soil creates ideal conditions for sewage degradation.

Raw or partly treated sewage from is applied on the bioactive soil in increasing dosage from 100 to 1000 L. Soil bed is continuously monitored during the sewage application. In a pilot study promising result in terms of % biodegradation of sewage have been obtained.

Parameter ppm	In Conc.	Out conc.	%degradation
BOD	220.0	53.5	75.68
COD	504.5	144.5	71.36
NH_3N	86.65	50.4	41.83
NO_3-N	2.5	1.38	44.8
Total-P	9.35	7.05	24.6

The easily operable system which dose not involve much maintenance and skilled man power is ideal for individual house complexes. Such systems can reduce total volume of sewage otherwise polluting the water resources like lakes and rivers.

Sand filter technique

A sand filter is prepared by using a 4' PVC pipe having a diameter of 6" and filled with small stones and pebbles upto 6" and on the top of it column of sand is laid. Further, this layer of sand was covered with a little quantity of activated charcoal to remove the colour and odour from the water sample. In one of such pilot study conducted at household level following results were obtained :

Table 1. Physical Characteristics.

Characteristics	Sample A (Before filtration)	Sample B (After filtration)
Colour (visual appearance)	Light green	Colour less
Turbidity (visual appearance)	Turbid	clear
Total Hardness	485 mg/lit	375 mg/lit
Odour	light fishy	odourless

Table 2. Chemical Characteristics.

Characteristics	Sample A (Before filtration)	Sample B (After filtration)
Total solids	1240 mg/lit	1080 mg/lit
Suspended matter	220 mg/lit	80 mg/lit
Total dissolved solids	1020 mg/lit	1000 mg/lit
Sulphates	288.4 mg/lit	247.2 mg/lit
Dissolved oxygen	1.6 mg/lit	6.46 m/lit
Chlorides	314.5 mg/lit	278.12 mg/lit
Acidity (Ph)	8-9	8-9
COD	150.37 mg/lit	56.3 mg/lit

Table 3. Biological parameters.

Characteristics	Sample A (Before filtration)	Sample B (After filtration)
Bioligical oxygen demand(BOD)	141.1 mg/lit	95.2 mg/lit
Dissolved Oxygen (DO)	2.4 mg/lit	6.46 mg/lit

Macrophytes as bio-cleansers of pollutants and nutrients

The diversity of aquatic weeds reflects limnological status of an ecosystem. Some weed species are bioindicators of aquatic pollution. In urban lakes of India blooms of common aquatic weeds like water hyacinth are a common phenomenon. Manual removal which is a common practice that usually leads to replacement of the weed with blooms of blue greens like *Microcystis.* The algal blooms can also be equally harmful to the health of an ecosystem. In a lake ecosystem if development of weeds is a reality, there is need to look at this as an opportunity. As a conservation strategy, the aquatic weed can be periodically harvested. Such controlled harvesting can effectively remove nutrients and other toxicants.

In one of the pilot study conducted on select weeds from lakes in and around Hyderabad, India, the role of aquatic weeds in removing heavy metals, one of the most toxic contaminants was assessed (Kodarkar, 2005). The results are summarized in Table 4.

Table 4. Analysis results of aquatic macrophytes from three lakes in Hyderabad, India.

Characteristics	Osmansagar				Saroornagar				Hussainsagar			
	A	B	C	D	A	B	C	D	A	B	C	D
% Water content	78.0	72.5	78.9	68.0	72.5	71.8	77.4	66.7	78.9	69.5	75.93	65.0
% Organic matter	12.4	14.2	6.45	4.5	14.2	15.3	7.32	5.3	6.45	17.0	9.50	6.0
% Total ash content	2.94	3.0	2.04	1.89	3.0	4.24	2.93	12.13	2.04	8.3	4.04	2.4
Atomic absorption spectroscopy (AAS) analysis : All values in ppm.												
Copper	-	-	-	-	-	-	-	-	57.03	126.13	148.05	-
Zinc	-	-	-	-	-	-	-	-	311.3	847.3	0.14 %	-
Arsenic	-	-	-	-	-	-	-	-	29.02	14.30	6.18	-
Cadmium	-	-	-	-	-	-	-	-	4.04	21.30	1.00	-
Lead	-	-	-	-	-	-	-	-	8.45	3.69	9.77	-

Sample A - *Cyperus alopecuroides*; Sample B - *Cyperus articulatus*;
Sample C - *Alternanthera philoxeroides*; Sample D - *Polygonum glabrum*

Stream Eco-System (SES) Technology

Natural in lets of lakes have their own in - built purification system that operates through a complex food web effectively removing nutrients from the in-flowing water in to a lake. This ecological principle was effectively employed on pilot scale for treatment of a stream in Pune, Maharashtra, India.

This novel technology uses filtration power of cellulose / fibrous material. A very good filter is developed when the cellulose / fibrous material like coconut coir or dried water hyacinth or aquatic grasses are compacted and woven to form a bridge / porous wall like structure. All the floatable and suspended solids are trapped in the biological bridge and the turbidity of flowing water is reduced.

Green Lake Eco - System (GLES) Technology

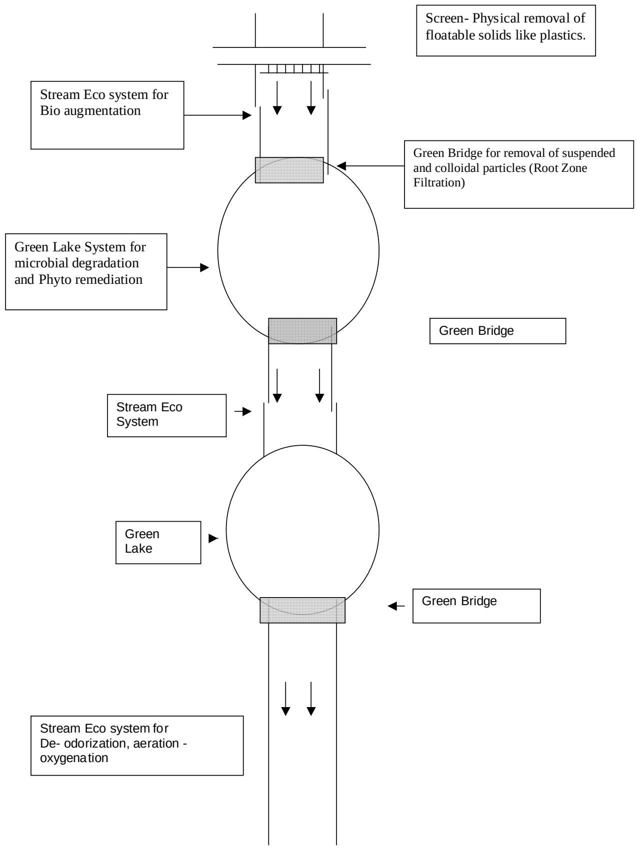
Green Lake uses floating, submerged or emergent aquatic plant species. These can be termed as *macrophyte ponds.* Macrophytes are capable of absorbing large amounts of inorganic nutrients such as N and P, and heavy metals such as Cd, Cu, Hg and Zn etc and to engineer the growth of microbes to facilitate the degradation of organic matter and toxicants.Green Lake system harnesses the power of floating plants like *Salvinia, Spirodella, Lemna* and *Eichornia*, with their enormous root systems, are very efficient at nutrient stripping.

Eichornia crassipes (water hyacinth) has been studied in detail worldwide. In tropical waters, water hyacinth doubles in mass about every 6 days i. e. more than 500 kg biomass / ha - day (dry weight). Nitrogen and phosphorus reductions up to 80% and 50% can be achieved i. e. a maximum storage of 900 kg N / ha and 180 kg P / ha in the form of biomass.

In Tamil Nadu, India, studies have indicated that the coontail, *Ceratophyllum demersum,* a submerged macrophyte, is very efficient at removing ammonia (97%) and phosphorus (96%) from raw sewage. It has a lower growth rate than *Eichornia crassipes,* which reduces the need of frequent harvesting.

In such green lake systems, the aquatic vascular plants serve as living substrates for microbial activity. The basic function of the macrophytes is to assimilate, concentrate and store contaminants on a short-term basis. Subsequent harvesting of such biomass results in permanent removal of stored contaminants from a lake system.

Floating macrophytes can be collected manually or by floating harvesters. The harvested plants can be used as cattle feed. It can also be used as green manure in agriculture as bio-fertilizer or converted into biogas.



Conclusion

Like Ayurveda, the traditional system of Indian medicine, eco-technological approach for mitigation of an ecological problem like eutrophication, a wholestic approach is adapted. In future, eco-tecnologies are bound to play an important role by virtue of their operational simplicity and cost effectiveness.

References

Central Pollution Control Board (CPCB), *2000. Status of Water Supply and Wastewater Generation, Collection, Treatment and Disposal in Class-I Cities*.

Gaudy, Anthony F. Jr. & Gaudy, Elizabeth T., 1980. *Microbiology for Environmental Scientists and Engineers.* McGraw Hill Book Co, New York.

Joshi, Sandeep & Joshi, Sayali, 2000. *Economical treatment of sewage by Soil Scape Process*. Proceedings of Seminar on Management of the existing water and sewerage facilities and services. Case studies and Analysis of proposals for future development organized by Institution of Public Health Engineering, Kolkata, pp. 19 - 21.

Kodarkar, M.S., 1995. Conservation of lakes, case study of five lakes in and around Hyderabad, Andhra Pradesh, India. *Indian Association of Aquatic Biologists (IAAB)*, Publ. 3. Hyderabad. PP : 85.

Kodarkar, M.S., 2005. *Evaluation of role of select weeds in decontamination of heavy metals in the waters of select lakes in and around Hyderabad, Andhra Pradesh, India*.

Lakes water quality monitoring and management programme in developing countries

Mutia, Silas M'Nyiri

Ministry of Water and Irrigation, P.O. Box 49720-00200, Nairobi, Kenya
Email:silasmutia@yahoo.co.uk

ABSTRACT

Developing Countries face an array of traditional and modern lakes water quality problems ranging from faecal contamination to toxic chemicals. Moreover, they do so in an economic environment that is severely restricted, an institutional environment which is often poorly structured, and for which the modern scientific knowledge base is frequently poorly understood and applied. Agencies in many developing countries recognize this as major impediment to sustainable development, especially as water quality has become one of the leading economic issues for the purposes of development and investment. Generally water quality programmes tend to suffer from traditional approaches, both of methodology and legal/administrative. The Consequence is that many programmes on lakes water quality are grossly inefficient; produce often unreliable data and which are not generally useful for making management or investment decisions, and face decreasing economic and political support. Programme modernization is essential to achieve the twin goals of greater efficiency and greater relevance in meeting data needs for contemporary lakes water quality management purposes. Modernization reduces costs, may reduce the amount of equipment and infrastructure required, and more closely matches the abilities of developing countries where, for example, knowledge of advanced environmental chemistry may be limited but where knowledge of biological systems is strong.

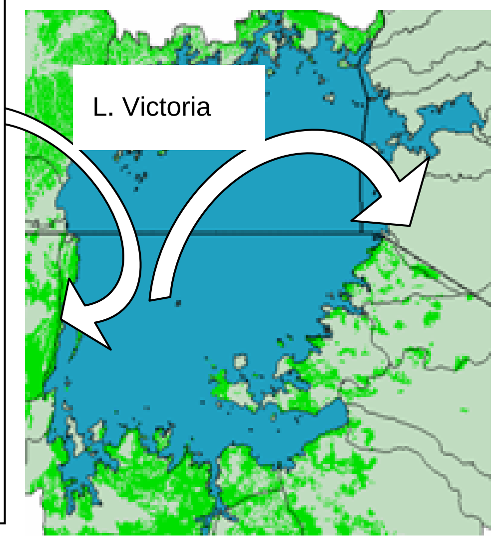
Key words. Management, Monitoring, Programme Modernization.

Introduction

The developed world has had the luxury of facing major problems of environmental degradation sequentially in time. These problems include faecal contamination and water supply, eutrophication, acid rain, toxic chemicals and ecosystem dysfunction, among others. In contrast, developing countries are facing these simultaneously. A typical example of this is demonstrated from the East African region where L. Victoria is simultaneously under threat of all the above pollutants in addition to others not mentioned. See the figure below.

Pollutants entering the Lake

1. Domestic effluent,
2. Nutrients,
3. Atmospheric deposition,
4. Water hyacinth,
5. Solid wastes dumping,
6. Climate changes,
7. Industrial emissions,
8. Industrial effluent,
9. Domestic effluent,
10. Agrochemicals,
11. Over-exploitation of fisheries,
12. Introduction of exotic fish species,
13. Sediments deposition from degraded catchment



Resulting condition

1. Eutrophication of Winam Gulf,
2. Proliferation of water hyacinth,
3. Water turbidity,
4. Loss of water depth at the inlets,
5. Acid rain
6. Ecosystems deterioration, disappearance or extinction (200 species within 50 years) of some fish, reduced the biodiversity of the lake's flora most notably the phytoplankton

This is happening in an economic environment that is severely restricted, an institutional environment which is often poorly structured, and for which the modern scientific knowledge base is poorly understood and applied. Agencies in many developing countries recognize this as a major impediment to sustainable development, especially as water quality has become one of the leading economic issues for the purposes of development and investment. Typically the lakes water quality data programme of developing countries fall into one of two categories: a) programmes that focus on "traditional" parameters such as major ions, nutrients and microbiology; (b) those that attempt to include some toxic chemicals such as metals and organic pollutants. Additionally, those rapidly developing countries with concerns for toxic organic and inorganic chemistry almost always adopt the "chemical list" approach to monitoring. This is equivalent to the 1970's "Priority pollutant" approach of the United States-Environment Protection Agency (US-EPA) which has been demonstrated to be inefficient and costly in the USA and other western countries which are adopting much cheaper, more flexible, and more effective alternatives.

A common observation of lakes water quality programmes is that they tend to be inefficient, the data are of uncertain reality, programme objectives are poorly linked to management needs for data, the analytical technology is often old and inefficient, focus is on water chemistry even though water is known to be a poor monitoring medium for many toxic chemicals, and data bases are incapable of mobilizing for management purposes (Ongley, 1993). The concept of programme efficiency includes consideration of all these factors, ranging from appropriate selection of parameters and sampling medium, to institutional inefficiency. It has legal and regulatory implications, especially where the regulatory framework imposes rigidity and prevents the use of more cost effective field and analytical methods. However, the greatest inefficiency tends to lie in the assumption that conventional water quality programmes produce data that can be used to make managerial decisions on pollution control, water resources planning, and relate to investments decisions. The fact is that such programmes are designed mainly for descriptive rather prescriptive purposes, with the result that nations tend to spend much money producing data that are closely linked to decision making and, not infrequently, not used at all.

Data for decision making

In developing monitoring programme it is necessary to identify the principal reasons for which monitoring data (chemical and biological) are required. Generally these are as follows:

- Description of lakes and river water quality at the regional or national scale including determination of trends in time and space,
- Determination of whether or not lakes water quality meet previously defined water quality of regulatory objectives for designated uses, including public health,
- Managerial resolution of specific pollution management issues, including post audit functions,
- Decision for investment options based on potential benefits from proposed or alternative remediation options,
- Comprehensive assessment of lake basins, especially to determine the relative importance of point versus non-point pollution sources,
- Lake Basin planning, including the development and audit of national/regional policies on land use relative to polluting land use activities and
- Reporting of compliance to international standards or action plans.

The inability of most national monitoring programmes in developing countries to meet many of these objectives requires rethinking of the role and practice of lake water quality monitoring. There is no single type of programme that meets all objectives; therefore the modernization of water data programmes requires a careful evaluation of objectives, knowledge of alternatives methods of achieving the objectives, and the ability to implement change that will enhance efficiency throughout the programme (data collection, laboratory operations, data assessment, and mobilization/use for the client)

Modernization

This paper focuses on the considerations that programme managers must make to implement modernization of Lakes Water Quality Monitoring and Management programmes.

Legal and institutional considerations

Role of Government in lakes water data programme:

The role of government in monitoring is being re-evaluated in many countries. The old model government has been doing and paying for everything. However, this is being replaced by market forces in many countries. This means that some governments, especially in developed countries, are reducing their direct participation in monitoring and enforcement, and are concerned primarily with setting and enforcing rules and standards. Some developing countries are making these changes and a good example can be sighted in Kenya where extensive Water Sector Reforms have been carried out.

Market forces will produce more efficient laboratories and decrease the need for government expenditures. Under market economic conditions, there is greater efficiency and less cost to government if industrial monitoring is carried out by industry with data reported to government agencies. This mechanism is used by USA, Canada and many European countries. A critical role of government is, however, in developing and enforcing national data standards through programme of quality assurance and laboratory accreditation. National standards impose a strict regimen on all aspects of the laboratory operations and, in some cases, field operations, so to ensure the reliability of the final data products. The present situation in some countries where there is devolution of decision making authority to local levels has been the loss of national data standards, which is catastrophic for developing countries that require reliable data for evaluating and deciding upon investment alternatives for remediation and/or development.

Commitment by management to change

The rational and comprehensive management of change is very difficult in large organizations. Senior management often results to budget solutions (reduced budget-reduced number of stations) rather than to a serious examination of how modern monitoring technologies, regulations and institutional structures can introduce greater efficiencies. It is essential that commitment be demonstrated at the most senior levels of an organization. An example of commitment is in the realm of training.

Changing legal and regulatory standards for efficiency

The use by many countries of rigid legal standards both for the parameters used for regulation and for the types of analysis that are permitted, is inflexible and inefficient. The US-EPA has now recognized that the rigid codification of analytical standards has not been as effective as once thought, but is expensive both for government and industry, and does not provide the flexibility to permit new and more efficient regulatory methods. The alternative as used in Canada is “Performance based” techniques which offers simpler and more cost effective bases for attaining programme goals. In “performance based” techniques the method is not rigidly prescribed, however the outcome must meet predetermined requirements of accuracy and precision. An example is the requirement in some countries to use Atomic Absorption Spectrophotometer (AAS) for metals, whereas new techniques using Emission Spectrophotometer reduces costs by one to two orders of magnitude by its ability to perform multitude simultaneous analysis on sample.

Maintenance of rigid legal standards ensures that countries will repeat the costly mistakes of many North American and European water quality programmes as it will not permit use of new and more efficient methods such as screening methods, and use of toxicity as a regulatory mechanism. In some cases, legal restrictions extend to the requirement to use out-dated field methods which precludes cheaper methods such as portable Dissolved Oxygen (DO) meters and proxy data such as turbidity measurement instead of direct measurements of suspended sediment concentration. In China for example, monitoring is restricted to standards and types of analysis identified by Chinese Law. Therefore, more efficient and modern methods, especially those of biological screening are difficult or impossible to use for most purposes. The Consequence is that, without greater flexibility, monitoring in China is unlikely to be able to modernize significantly (except to build bigger and more expensive laboratories).

Technical

Laboratory programmes

Recognizing that there are economic and cultural limitations to issues of efficiency, one of the greatest barrier to modernization is the inefficiency and in some instances, duplication of government laboratories within small geographical areas. Sample volumes are small and therefore preclude economies of scale that are possible with the modern, automated instrumentation. Many laboratories lack auto samplers without which volume analysis is impossible and data accuracy is jeopardized. In some countries, laboratories have approximately 30 staff who handle some 2,000 samples per year, a number that is about what a modern laboratory would process in 2-3 weeks. Gross inefficiency also produces waste in training, equipment procurement and infrastructure costs because of the need to train more operators than are necessary, to purchase equipment that is not efficiently utilized, and to build and maintain redundant facilities.

The objective of laboratory optimization is faster turn-around time at lower cost per sample of water, sediments and tissue, and for any toxicological test that may be included in the analytical programme. Modernization may involve some or all of the following considerations:

- Sample handling, with special consideration for samples that may be used for legal purposes,
- Turn-around time required,
- Reporting requirements,
- Sample tracking – level of automation,
- Client access to data bases,
- Types of analysis that are meaningful – totals/filtered for water; sediment and issue analysis; toxicology testing,
- Detection levels required and appropriate instrumentation,
- Level of Quality Control (QC) required for different types of clients,
- Type of Quality Assurance (QA) required for lab operations and inter-lab comparison,
- Availability of certified pure reagents,
- Personnel availability and/or needed, and training required,
- Operating budget and cost recovery,
- Ambient air quality (site limitations),
- Facility infrastructure, including ambient laboratory temperatures and stable electric power supply and
- Workplace safety and health.

Multiple techniques within monitoring programmes:

Most monitoring programmes tend to be relatively inflexible and are unable to accommodate many of the new approaches to monitoring. In part this tends to reflect old fashioned legislation on water quality and, in part, reluctance on the part of managers to change. Nowadays, monitoring is a comprehensive activity that includes many different approaches to problem solving including fixed site monitoring, field surveys, emergency mobile monitoring, toxicity assessment, and environmental effects monitoring which focus on in-stream biological response to control measures.

Biological assessment and toxicity assessment are increasingly being used as part of modern monitoring and enforcement programmes and, in some countries, are dramatically reducing the need for expensive chemistry. The interpretation of toxicological tests is now well understood and has at least as much, and usually more, relevance to decision-making as chemical data. Toxicity is used by several Canadian provinces, United States and a number of European jurisdictions for industrial effluent testing as a cheap and effective screening method.

New screening and diagnostic tools

There now exist many new and cost effective techniques for lake water quality monitoring which are more cost effective, produce more useful information that is linked directly to decision, which save time, and have reduced instrumentation needs. These methods follow from the recognition that 1970's type chemical monitoring is not very useful for managing lakes and rivers. These "tools" include a wide range of new biological, biochemical, chemical and toxicological techniques. Selection amongst these many new techniques depends very much on the objectives of the programme and the ability to integrate these techniques into the water programme. These new tools produce useful information quickly and cheaply, and can be used to define which parameters need further and detailed analysis. They also include methods which are outright substitutes for conventional data programmes or which are used to determine whether there is reason to continue with a more detailed and expensive analysis.

As an examples, simple cheap field kits are available for screening samples for toxicity. Toxicity Evaluation Identification (TEI) is a procedure which permits accurate determination of those chemicals which cause toxicity to selected organisms and which should be added to the monitoring programme. Indeed "TIE" is usually sufficiently precise that it allows identification of a particular industry group that needs to be studied or regulated first. One could completely replace traditional chemical monitoring of large and polluted lakes and rivers with a combination of benthic surveying (for DO and nutrient conditions), use of sentinel organisms (for target toxic chemicals) and cheap toxicity testing. The data produced by such a programme are cheap, and more relevant for managing the lakes and rivers, especially as chemical data are not directly related either to human health or to the ecological "health" of the lakes and rivers. Also, it is now widely accepted that toxic chemicals of concern are frequently none of those routinely included in chemical analytical protocols in many countries, and exists at concentrations that are known to be of concern, but which are below the detection levels used in many laboratories.

Data quality objective

While much of the monitoring in many countries appears to be established according to legal requirements, monitoring stations should begin the use of Data Quality Objectives (DQO's) to ensure greater communication between the monitoring programme and those that use the data. DQO's are a process through which the laboratory and the client mutually examine the type of data needed for the management issue at hand, the level of reliability required, reporting requirements, etc. The objective is to ensure that the client understands the costs, the limitations, and the uncertainty in the information produced by the monitoring programme. This process is essential when monitoring programmes achieve greater flexibility in the choice of methods, etc. In a market economy, the DQO process, either formal or informal, becomes the basis for a contract between the monitoring station and the paying client.

Optimization of the national network:

Optimization is a complex and unfamiliar task to many agencies. While optimization may mean reduction in the size of the network and rationalization of the types of parameters, it can also include change in monitoring sites and the addition of more relevant parameters or in limited stations, the expansion of the network to include important unmonitored lakes and rivers. Optimization also includes the detailed analysis of historical data sets in order to identify and eliminate data that do not change or that change in very predictable way (e.g. annual cycles) or that consistently report ND's (not-detectable). Optimization includes reconsideration of types of data, including use of biological survey methods and measures of toxicity, plus use of sentinel organisms for determining the presence of important industrial parameters.

Information system

Database and information systems in most monitoring agencies in developing countries are not effectively used for information processing, analysis and visualization of data, and decision support functions. This has two types of implications: a) data are not easily accessible for managing purposes, b) water quality programmes remain largely invisible because of the lack of highly visible data products; the result is that often such programmes fail to win managerial and political support.

Modern information systems includes: database and data archiving, Geographical Information Systems (GIS), analytical tools (statistics, graphics, etc.), decision support capabilities, and visualization (output display) capabilities. They should be capable of being operated by non-specialists with minimum training. One such example is Canada's RAISON system. Great care should be taken to match real GIS requirements with the type of GIS system purchased, especially as GIS is only part of a full information system. Indeed, the GIS requirement of most water agencies is quite small (usually limited to handling georeferenced site information, spatial moping, and limited map overlaying) and which can be met by many cheap GIS software packages. Large Commercial GIS systems should only be used for specialized GIS activities because the learning requirements are substantial, the hardware and software costs are high, and only specialists can efficiently use such systems. Contrary to GIS vendors, GIS is only one part of a complete information system.

In Mexico for example, the Mexican water Commission is a partner in the development process of suitable cheap software based on the RAISON software platform. The main reasons are the high cost of providing commercial packages to all the regional offices, and the need to develop simple applications for routine tasks that can be operated by relatively unskilled operators. These tasks include source inventories, data interpretation, and standard report writing. This approach has integrated with most GIS systems in the Mexican government and linked directly to commercial databases.

Quality control, quality assurance, accreditation and good laboratory practices

Data quality in many developing countries is a serious problem. Increasingly QA, QC and Good Laboratory practices (GLP) are a major part of bilateral assistance programme in developing countries. The problem of reliable data within a laboratory has too many facts to discuss here. These includes not only the normal quality control steps during analysis, but also the difficulty in many countries of obtaining pure reagents or of ensuring that the so called certified reagents are, in fact pure. Site conditions in many developing countries are a major problem, especially the location of laboratories in highly polluted air sheds of major cities. Rarely do laboratories have proper air handling systems that can deal with such problems. This is particularly acute for metals and, given the recent experience with contamination of samples in very well operated North American metals programmes, suggests that metals data are especially unreliable in developing countries. A major advancement and absolute necessity, especially in large countries with many laboratories, is an accreditation programme that establishes common performance criteria for all labs responsible for water quality data used by government. Modernization requires rigorous application of these principles both in government laboratories and in those private laboratories that serve government needs.

Reporting

For descriptive purposes at the national scale, monitoring should increasingly focus and report upon biological conditions and measures of toxicity including the use of sentinel organisms for determining the presence of important industrial parameters. This follows from the recognition that chemical data are difficult to interpret or to relate to actual ecosystem state of the water environment. Data should be integrated into several useful indices that describe the water according to water use. Surface water sites should be reported separately from a national effluent monitoring programme. If there are a large number of river sites, the sites should be integrated (for reporting purposes) into the river reaches.

Capacity building

Training

Contemporary needs of water managers for water quality data and for informed analysis and interpretation, is often beyond the technical ability of staff in many developing countries. Managers of monitoring programmes are often not committed to sufficient training, especially for junior staff. Investment in infrastructure, equipment etc., is not effective or efficient without a commensurate investment in training. In the 1995 redesign of the Mexican national water quality programme, a significant percentage of the programme resources are dedicated to training and professional development. This training includes activity oriented training for the bulk of field and laboratory staff, and advanced training in Mexico and abroad for key staff who will be the leaders of the programme in the future. In contrast, certain other countries are reluctant to invest in training as a matter of programme strategy and tend to rely on training that can be obtained at low or no cost from donors. Unfortunately, experience suggests that while such courses may be individually of merit, collectively they do not comprise a well structured and balanced programme that needs to include consideration of the types, numbers and length of foreign "training" mission undertaken by domestic staff.

There is much use by many countries of "short courses" by foreign professional. Experience indicates that short courses are generally valuable only in two circumstances – is for the raising awareness of new types of monitoring and assessment methodologies; the second is the training in very specific topical areas having limited scope (e.g. training in particular methodology).

Training programmes overseas that involve extensive visits to labs etc., especially for junior staff, are not particularly cost effective. Foreign training should be restricted to highly specialized training (e.g. specialized analysis, computer training, etc.), for extended educational purposes (advanced degrees), and for senior staff who need familiarization with foreign environmental management methods, policies and regulations.

A national strategic training programme should establish training goals of fixed number of PhD and MSC degrees within a given period for the purposes of developing a core group of professionals around which the next generation of monitoring will be built. The principal mechanisms for advancement of monitoring programmes in developing countries include:

- On a competitive basis, identify the best persons for foreign education in environmental chemistry and toxicology, environmental assessment etc.
- Increase the number of persons with advanced degrees within monitoring programmes
- Promote persons with advanced foreign education and suitable experience to positions of management responsibility

Other mechanisms include the development of long term relationships with foreign companies and agencies for importation of appropriate methodologies. This may be done through international aid programmes and in some cases with specialized foreign agencies (e.g. US-EPA). However, the relationship with foreign agencies and companies can increasingly be expected to be of a commercial nature due to the economic change that is occurring in most western countries. Also, for many developing countries, commercial relationships are consistent with developing markets and economies, and after a fast-track solution to training, programme efficiency and financial sustainability of the programme by the development of market based programme in which the government is one of the client.

Institutional development

Institutional development is essential to deal with all aspects of programme management, legal change, etc. Below are two very specific types of institutional development that are essential if modernization of data programmes is to be successful.

(a) Client-Oriented programmes: It is essential that managers of monitoring programmes regard their work as a service to the clients. Clients in most cases are the different levels of government, however as the move to a market economy driven increases, there may develop the need to work with other types of clients. Use of Data Quality Objectives (DQOs) ensures greater communication between the monitoring programme and clients that use the data. The objective is to ensure that the client understands the costs, the limitations, and the uncertainty in the information produced by the monitoring programme.

(b) Revenue generation: Data programmes can be put on a revenue generating basis, especially if the government is considered to be a client. A revenue basis is essential for financial sustainability of programme; it is also a major tool in creating the appropriate business approach to data programmes. A business approach ensures that the data programmes are relevant and efficient, well connected to users, and are operated according to good scientific and business practices. There are many ways to accomplish this; however the main message here is that the government managers of water programmes usually find the concept of revenue very difficult to accept. Hence, institutional change requires education about alternative approaches to the business of water data programmes. Experience shows that modernization creates opportunities for revenue generation for government laboratories and opportunities for creative partnership between government agencies and the private sector. In many developing countries, a revenue base may provide the funds to adequately compensate staff to achieve personnel stability.

Conclusions

Water quality monitoring modernization (e.g. using Emission Spectrophotometer instead of AAS- ability to perform multitude simultaneous analysis on sample)

- Will achieve the twin goals of greater efficiency and greater relevance in meeting data needs for the water quality management purposes,
- Will reduce: costs, the amount of infrastructure required and the amount of data collected.

Water quality monitoring modernization will need the inclusion of legal and institutional considerations**,** technical issues, and a strategic programme of capacity building.

Monitoring should increasingly focus and report upon biological conditions and measures of toxicity including the use of sentinel organisms for determining the presence of important industrial parameters

Serious examination of how modern monitoring technologies, regulations and institutional structures can introduce greater efficiencies rather than result to budget solutions (reduced budget-reduced number of stations) is necessary.

Management of the lakes water quality will require adequate prevention of further pollution and appropriate management of the present pollution.

References

Ongley, E. D., 1997. *GEMS/WATER Programme*. Water Quality Monitoring and Management

Ongley, E. D., 1993. Global Water Pollution: Challenges and Opportunities. *In*. Integrated Measures to Overcome Barriers to Minimize Harmful Fluxes from Land to Water. *Proceedings of the Stockholm Water Symposium*. Publication No. 3, Stockholm Water Company, Sweden

G.J. Mathews, 1994. Linkages between water quality protection, educational renewal, and credit worthiness. *In Integrated Land and Water Management: Challenges and New Opportunities*. Proceedings of the 1994 Stockholm water symposium. Publication No. 4. Stockholm water Company, 63-69

Ministry of Housing, 1995. *Monitoring water quality in the future*. Spatial Planning and the Environment, Department for information and International Relations, the Hague, The Netherlands

The effect of turbid inflows into Winam Gulf, Lake Victoria: a 3D modeling study with ELCOM-CAEDYM

Njuguna Henry[1], Jose R. Romero[2], Patrick Khisa[1], Tom Ewing[2], Jason Antenucci[2], Jorg Imberger[2] and John Okungu[1]

[1] Lake Victoria Environmental Management Program – Water Quality Component, P.O. Box 1922, Kisumu 40100, Kenya
Email: hmnjuguna@yahoo.com.au
[2] Centre for Water Research, University of Western Australia, 35 Stirling Highway, Crawley 6009, Australia

Abstract

The catchment of Winam Gulf supports a population of 4.5 million inhabitants. Much of the population practices subsistence agriculture in the catchment that has led to high rates of sediment erosion and ultimately highly turbid rivers. Hence, every year during the two equatorial wet seasons, large quantities of catchment sediments are deposited into Winam Gulf.

In Apr.-May 2005 an intensive field investigation was undertaken to determine the exchange dynamics between the northeastern offshore waters of Lake Victoria and Winam Gulf through Rusinga Channel. One of the outcomes of this scientific study was the validation of ELCOM (Estuary, Lake and Coastal Ocean Model), a three-dimensional (3D) hydrodynamics model, over Rusinga Channel and Winam Gulf. CAEDYM (Computational Aquatic Ecological DYnamic Model) is a water quality model that can be readily linked to ELCOM.

In this modeling study we linked ELCOM and CAEDYM to evaluate the transport and fate of riverine sediments in Winam Gulf from several of the large rivers. Several size classes of suspended solids were evaluated. The extent and concentrations of sediment plumes were modeled over a range of riverine discharge. Further, simulations were run with reduced riverine sediment concentrations to evaluate the effect of turbidity levels in Winam Gulf if catchment improvement practices were implemented. This study illustrates that application of coupled 3D hydrodynamic and water quality models can serve to improve understanding of sediment plume dynamics in tropical lakes.

Key words: Hydrodynamics, Modeling, Sediments

Introduction

One of the identified causes of eutrophication of Lake Victoria is the nutrient loading by rivers and diffuse runoff (Bootsma & Hecky, 1993). Though lake-wide eutrophication from these sources may likely result after many years or even decades, the effects of localized eutrophication can be much more immediate. Eutrophication has accelerated since the 1960s following conversion of extensive tracts of forests and floodplains to agricultural land. Evidence for the contribution of riverine loading to the eutrophication of the lake is illustrated through a series of satellite images during periods of high inflow discharge (Fig 1). Riverine inputs can lead to the degradation of the water quality of Lake Victoria through nutrient and contaminant loading, much of which is absorbed to the sediments. In shallow environments such as Winam Gulf, winds can cause resuspension and transport of these sediments to a much wider region of the water body. In this study, the spatial distribution of sediments over a several week time scale over Winam Gulf is examined with conservative tracers for a number of rivers and simulated concentrations of suspended solids.

Because Winam Gulf is shallow, we expect the river discharge not to behave as underflows, interflows or surface overflows. Rather, the combination of the kinetic energy of the river entering Winam Gulf, and the action of wind and surface forcing to mix the water column in this shallow environment, is known to create relatively vertically homogeneous conditions. Hence, the transport and settling of inorganic particles from the rivers is expected to be governed by wind driven water currents and the momentum generated by stream discharge.

Method

In this study, four scenarios were considered with a three-dimensional hydrodynamic model coupled to a sediment fate model. The hydrodynamic model ELCOM (Hodges et al. 2000) was used to simulate the spatial and temporal distribution of river water through the simulation of conservative tracers. The fate of inorganic suspended solids through settling was modeled with a ecological model, CAEDYM (Romero and Imberger 2003), coupled to ELCOM, which simulated the transport of the particles. A uniform numerical grid was used with 250 m cells in the horizontal with 1 m layers in the vertical. The model domain included all of Winam Gulf, Rusinga Channel and a portion of the northeastern sector of Lake Victoria. Bathymetry was interpolated from the original 500m data using Landsat satellite image for edge resolution (T. Ewing, pers. comm.). The western boundary of the model domain was an open that required temperature and lake level inputs as described below. Four rivers were configured in the model domain, though we focus on the two large rivers that enter in the southeastern corner of Winam Gulf. Free slip boundaries were used to model sidewall (land) boundaries and no slip boundaries were used for bottom boundaries.

Our ELCOM modeling had an open boundary condition on the western margin of the model domain that was forced by temperature measurements at T1 and lake surface levels measured by an ADCP at T2 (Fig 1). The lake level data at T2 was shifted back in time by 2.4 hours to account for time of wave travel between T1 and T2. Wind and other meteorological parameters on shortwave radiation, total radiation, air temperature and relative humidity measured at T2 were assumed to apply over the entire model domain.

River inputs for ELCOM included discharge of the rivers and for CAEDYM the suspended solid concentrations. Time varying riverine discharges was measured at gauging stations upstream of the confluence with Winam Gulf, and served as inputs in the simulations for the Sondu and Nyando rivers. Inflow data for the period April-May 2003 was used as the high discharge year, and April-May 2000 for the low discharge year (Figure 2). Estimates of suspended solid concentrations were based on measurements during low to medium flows in the Nyando River with the relation, $SS=aQ^b$, where a = 24.74, b = 0.372, Q is the discharge rate in $m^3 s^{-1}$, and SS is the suspended solids concentration in mg L^{-1}. Two discharge regimes reflecting high and low river inflows were simulated to characterize the spatial extent of riverine sediment plumes. Lastly, scenarios with reductions in the sediment concentrations of the rivers were considered to assess the extent catchment management practices to reduce erosion would decrease sediment plume coverage over Winam Gulf. Improved estimates of the suspended solids concentrations are recognized.

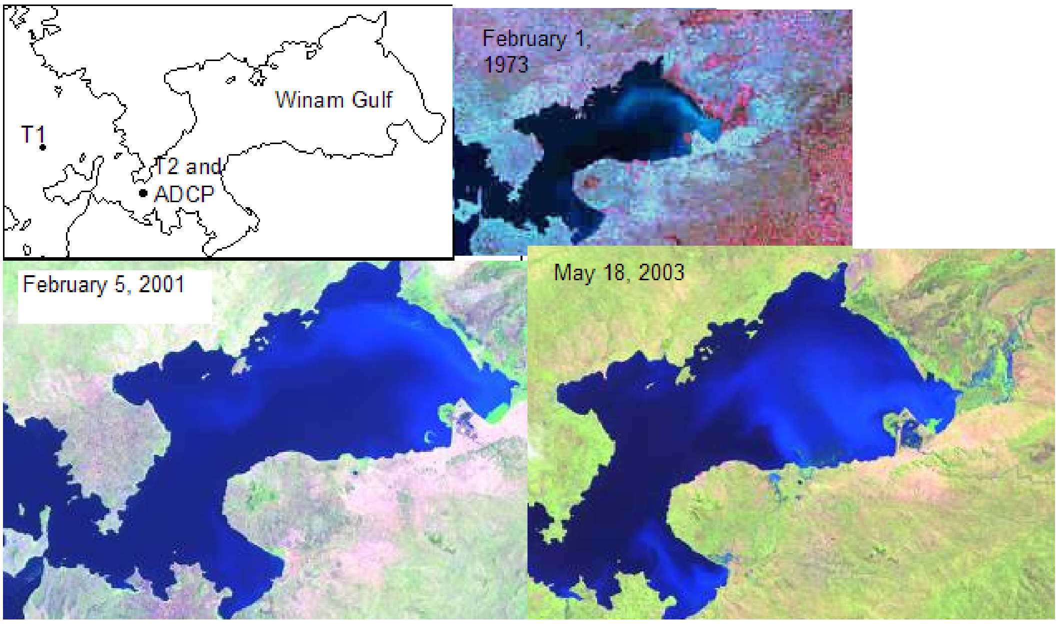


Fig. 1 . Position of meteorological buoys and satellite images of sediment plumes from sediment laden river discharge into Winam Gulf during periods of high flow. Sediment plumes generally cover a large extent of the eastern half of Winam Gulf and evidence for a counterclockwise gyre is suggested by these images.

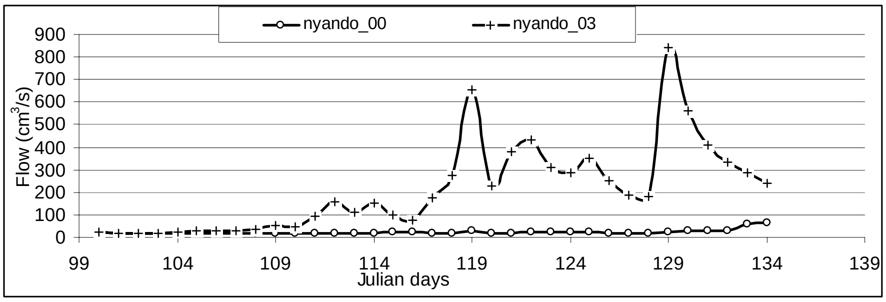


Figure 2. Nyando River during high (2003) and low discharge (2000).

Further, each of the rivers were given a separate tracer so that each stream transport and mixing throughout Winam Gulf could be tracked in the simulations. The two size classes of the suspended solids was set to diameters of 0.7 and 2 µm to evaluate differences in small particle size on the plume coverage (Table 1). A reduction by 75% in the suspended solids levels was simulated to characterize catchment improvement practices. Several outputs formats were configured for the tracers, temperature, suspended solids and currents in ELCOM, including horizontal sheets along the sediments and the surface, and profiles at certain locations. The horizontal sheets were particularly useful for comparisons with the observed satellite data. We also assessed whether the wind data during this period was representative of other years and the results (Figure 3) show a fairly reasonable agreement with other years.

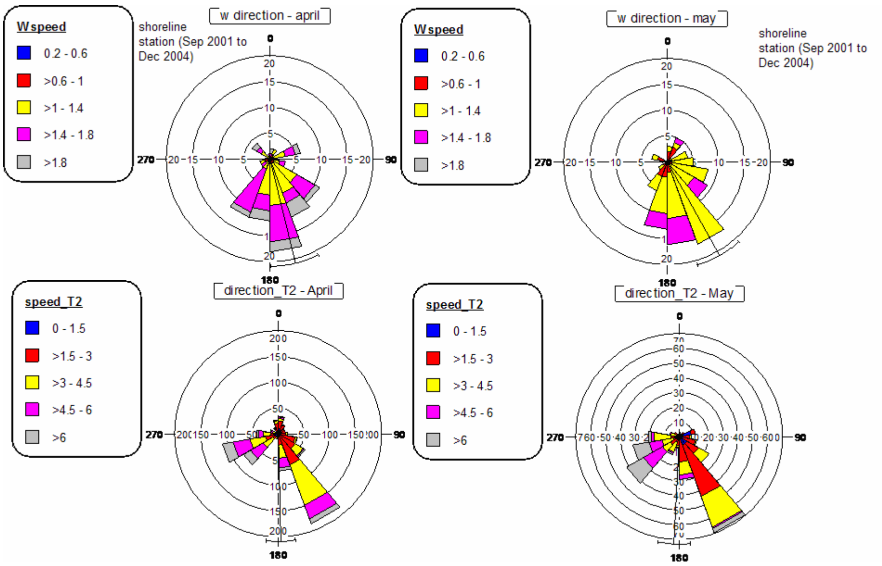


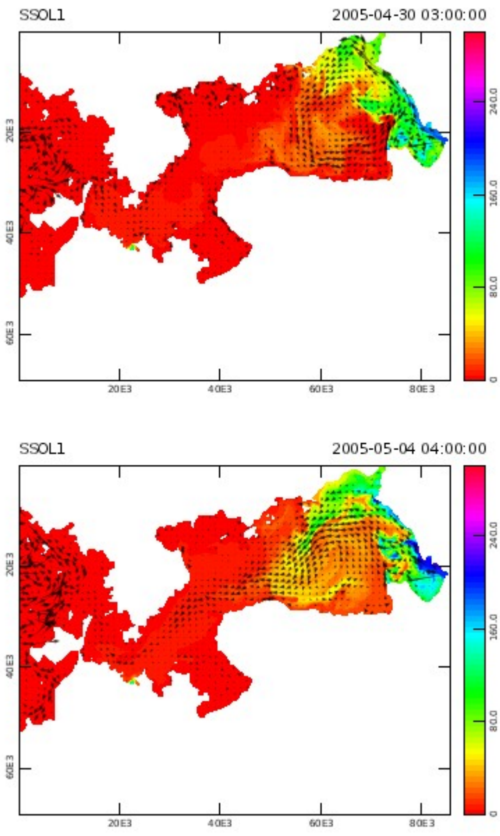
Figure 3. Wind roses at a shoreline station, 100 km northwest of the river mouths (Sep. 2001 to Dec. 2004, upper panels) and at station T2, 55 km east of the river mouths (lower panels).

Table 1. Matrix of scenarios and discharge and total suspended solid concentrations.

Scenario	Discharge	TSS concentration in rivers
1	Low (April-May 2000)	Normal
2	High (April-May 2003)	Normal
3	Low (April-May 2000)	Low (25% of normal levels)
4	High (April-May 2003)	Low (25% of normal levels)

Results

A comparison of the simulated sediment plume with satellite images indicates the model simulates the gross characteristics quite well. Satellite images of Winam Gulf from several dates over the past 20 years consistently illustrate a plume that is transported to the north along the eastern shoreline and then to the west parallel to the northern shoreline, in effect a counterclockwise gyre (Figure 1). These observations were well reproduced by the simulations as shown by the time series of surface sheets from model run with high discharge and normal suspended solids levels (Figure 4) in which time series of the gyre set up within the shallower expanse of eastern Winam Gulf is shown as simulations.



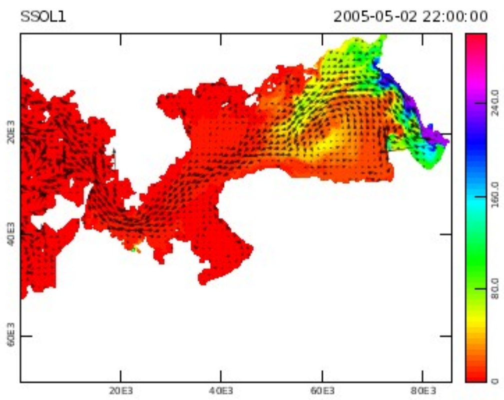


Figure 4. Simulated circulation on April 30, 2005, during a southeasterly (135°) wind event of approximately 3.5 m/s. Arrow length indicates current magnitude and colour bar indicates suspended solids concentration. A two gyre wind response is indicated, where the flow current is downwind in the shallow regions of the lake (mostly around the periphery) and against the wind in the deeper central region of the lake. Although the model is 3D, the depth-averaged currents are shown to indicate overall water movement.

Gyre formation was also found to depend on the wind direction. During the afternoon strong lake breeze, current speed was unidirectional uniformly across the gulf towards the east, while in the night when land breeze are dominant, the gyres are set up with no significant movement of water beyond the area to west of the gyre. The tracer patterns for the two major rivers and two minor rivers provide insight into the large scale transport in the gulf (Fig. 5). Tracers from the two large rivers with confluences in the southeastern corner of Winam Gulf had similar patterns to the suspended solids concentrations. However, tracer levels of the small river that enters the Gulf at Kisumu in the northeastern corner remained elevated in the small embayment, which suggests that the large scale counterclockwise gyre acts as a barrier of transporting these waters from the bay. In other words, this bay is not well-flushed.

Water currents in the surface were seen to correlate well with suspended matter concentration e.g. Fig 7 that shows trend in space of the two quantities made along a north-south transect near the eastern shoreline. ELCOM simulations showed circulations within the far field of the river mouths to be influenced by wind from the south.

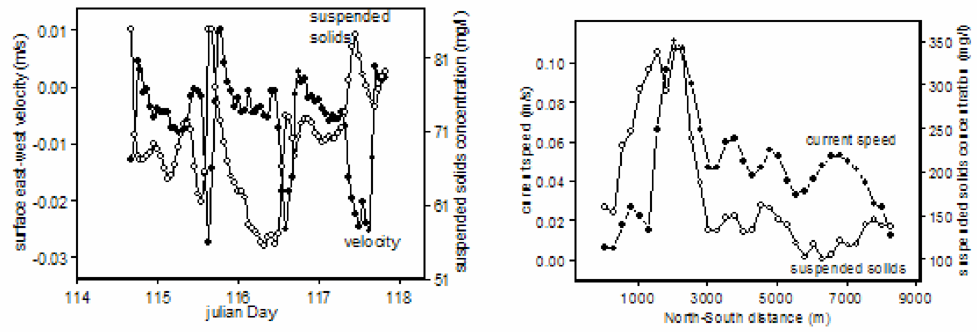


Figure 5. Simulations of the variation in time of suspended solids concentration and along gulf velocity (left panel) and variation in space at the surface layer of current speed and suspended solids concentration along North South transect (right panel).

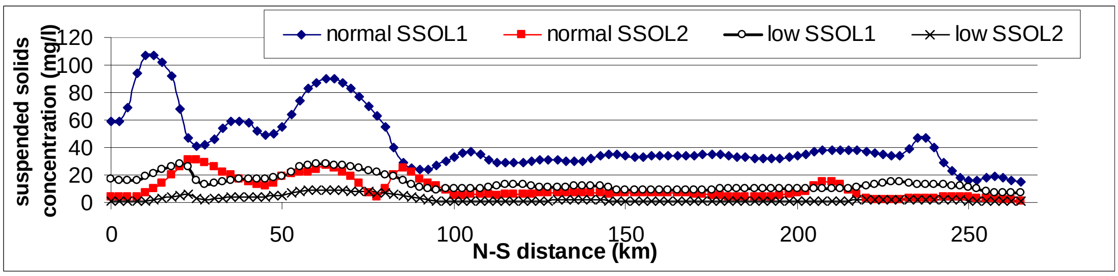


Fig. 6. Variation in space of the suspended solids simulated for the two scenarios.

Another notable observation was the decrease in concentration within the far field of the river mouths by _% when the suspended solids loading from the catchment were reduced to 75% of the normal loading (Figure 6). In the far field, approximately 40 km from the river mouths, a point was picked to examine pattern of spread in time between the two scenarios of loading. It emerged that within the first 9 days of the simulation, SSOL1 maintained a fairly steady concentration (ambient one), but thereafter, the normal conditions concentration rose sharply by 52%, while the concentration for a scenario of 25% reduction in loading remained steady though the discharge increased like in normal loading scenario above. The same picture was reproduced for SSOL2 although the time to concentration decay was higher in SSOL2 than in SSOL1. Those of lighter density, i.e. SSOL1 showed a much less decrease compared to those of higher density (SSOL2). Various phenomena observed are discussed in the following section

Discussion

A number of studies have identified cause of algal blooming in inshore areas of the lake as due nutrient loading by rivers (Bootsma & Hecky, 1993, Hecky, 1993). Recommendations have been made on the need to implement catchment restoration to reduce soil erosion hence ameliorate eutrophication of the lake. Understanding mechanisms involved in riverine sediment spreading in the Winam Gulf has been the main objective of this study. Results from this study suggests that is it largely controlled by wind driven currents, while the intensity of the sediment cloud in time is dependent on riverine discharge and in space, by depth of the location. Wind pattern on a diurnal basis is such that it sets up surface seiches whose frequency of occurrence is 6 – and 12-hour. ADCP and meteorological data showed current to be principally driven by the surface wind stress. Model simulations suggests that the frequent reversals in the wind-driven flow may effectively limit the alongshore extent of the plume in the southern shoreline. Unlike the observation of Churchill et al. (2003) in Ontonagon River mouth (Lake Superior) in which resulting reversals in the alongshore direction of the plume's motion extended in both directions from the river mouth, our simulation showed a plume that followed the direction of the predominant wind direction in Lake Victoria (Podsetchine et al., 1996; Schott and Fernandez-Partagas 1981; Fish 1957). As it is common of many rivers entering large lakes in which emerging river water is effectively impervious to the effects of the Earth's rotation (….), the positioning of the study area a fraction of a degree to the south of the Equator has the implication that the Earth's rotation has insignificant effect on the two rivers. Its movement appears to be principally controlled by Effects of wind forcing, mainly the southeasterlies.

Sediment entering Winam Gulf is expected to be affected by time-varying wind fields, to vertically well-mix given the shallow depth (< 3m) of the area in and around the eastern Winam Gulf, to undergo horizontal dispersion due to turbulence set up by river inflows and wind induced shallow water transport and to be affected by bottom-friction because of the uneven distribution of the bed topography.

Settling velocity of particles and flocs is the complex interaction between particles and turbulence (McCool, and Parsons 2004). Turbulence in the area under consideration was seen to be higher as shown by higher values of dissipation and it is expected to causes perturbations to the straight trajectory of a particle falling under gravity (McCool, and Parsons 2004). Like in Wang and Maxey, (1993); Aliseda *et al.* (2002), turbulence was found to increase the effective settling speed of particles by 10–50% in stratified parts of the lake.

References

Aliseda, A., Cartellier, A., Hainaux, F., Lasheras, J.C., 2002. Effect of preferential concentration on the settling velocity of heavy particles in homogeneous isotropic turbulence. *Journal of Fluid Mechanics* 468, 77–105.

Bootsma, H.A.and Hecky, R.E. 1993. A comparative introduction to the biology and limnology of the African Great Lakes. *J. Great Lakes Res*. 29 (Suppl. 2):3–18.

Churchill J. H, E. A. Ralph, A.M. Cates, J.W. Budd, N. R. Urban. (2003) Observations of a negatively buoyant river plume in a large lake. *Limnol. Oceanogr.,* 48(2), 2003, 884–894

Fish, G.R. 1957. A seiche movement and its effect on the hydrology of Lake Victoria. *Fish. Publ. Lond*. 10:1-68.

McCool W. W and J.D. Parsons, 2004. Sedimentation from buoyant fine-grained suspensions *Continental Shelf Research* 24 1129–1142

Podsetchine, V., and T. Huttula, 1996. Hydrological modelling: activity report for the periods of 01.04-31.05.1995 and01.09. -30.09.1995. FAO/FINNIDA *Research for the Management of the Fisheries on Lake Tanganyika*. GCP/RAF/271/FIN-TD/45: 109p.

Schott, F. and J. Feernandez-Partagas. 1981. The onset of the summer monsoon during the FGGE 1979 experiment off the East African coast: a comparison of wind data collected by different means. *Journal of Geophysical Research* 86:4173-4180.

Wang, L.P., Maxey, M.R., 1993. Settling velocity and concentration distribution of heavy-particles in homogeneous isotropic turbulence. *Journal of Fluid Mechanics* 256, 27–68.

Water management problems in the Lakes District of the Ethiopian Rift: Call of the time

Tenalem Ayenew

Addis Ababa University, Department of Earth Sciences, P.O.Box 1176. Telfax. 251-011-239462, Addis Ababa, Ethiopia

E-mail: Tenalema@geol.aau.edu.et

Abstract

The Ethiopian rift is characterized by many perennial rivers and a chain of lakes varying in size, hydrological and hydrogeological setting. The water resources of the rift are one of the focal points for large-scale development in the last few decades. Some of the lakes and feeder-rivers are used for irrigation, soda abstraction, fishing, recreation and support a wide variety of endemic birds and wild animals. Ethiopia's major mechanized irrigation farms and commercial fishery are confined within the rift. Few of the lakes are shrunk due to excessive abstraction of water; others expanded due to increase in surface runoff and groundwater flux from percolated irrigation water. Human factors, in combination with the natural conditions of climate and geology influenced the water quality. The chemistry of some of the lakes has been changed dramatically. This paper tries to addressee the challenges of surface water resources development with particular reference to environmental problems caused in the last few decades on three rift valley lakes. The methods employed include field hydrological and hyrogeological investigations supported by aerial photograph and satellite imagery interpretations, hydrometeorlogical data interpretation and hydrochemical analysis. The result revealed that the major changes in the rift valley are related to recent improper water use and mismanagement, which appears to have grave environmental consequences that needs urgent intervention.

Key words**:** Ethiopian rift, lake level, water management

Introduction

The Ethiopian rift valley is endowed with a chain of more than 15 permanent lakes, which are highly productive, containing indigenous populations of edible fish and supporting a variety of aquatic and terrestrial wildlife. These lakes are being used for various purposes without integrated water management plan. Water is abstracted excessively for irrigation and soda ash without the basic understanding of the complex hydrological system and the fragile nature of the rift ecosystem. The fastly growing water use and land degradation resulted in noticeable negative environmental changes for the last three decades (Tessema, 1998; MWR, 1999; Ayenew, 1998, 2003; Dagnachew Legesse et al., 2003; OEPO, 2005). The most notable problems are reduction of lake levels; increase in lake level as a result of excessive inflow of infiltrated water from over-irrigated farms through open rift faults; water quality changes, soil salinization and danger on the rich biodiversity.

In this study three lakes and their environs are selected to illustrate the recent anthropogenic influences on the lakes. I have also attempted to relate my assessment of lake level changes with water quality including potential negative effects on biodiversity and salinization.

Methodology

The lake level records were used to reconstruct the recent changes and to correlate with catchment hydrometeorological factors. Information on abstraction of water for irrigation and soda production was gathered from relevant institutions. The conventional water balance estimation allowed an understanding of the relative importance of the various components of the hydrological cycle on the lakes. Independent estimate of the groundwater flux to some of the lakes was made (Ayenew, 2001). To evaluate the spatial variation of lake levels and to reconstruct the positions of the different shorelines, multi-temporal satellite images: Landsat Multispectral Scanner (1978,1979), Thematic Mapper (1987, 1989), NOAA (1994, 1995) and SPOT (1986, 1993), Landsat ETM+ (1999), MODIS (2003,2005) as well as panchromatic aerial photographs at the scale of 1:50 000 (1965, 1967) were used. The synthesis of the results is presented and contains judgments to resolve contrasting evidence derived from analysis of components of the hydrologic cycle and anthropogenic factors. Much of the evidence converges on lake levels. The hydrochemical analysis result provided another independent check.

Description of the Region

The three-studied lakes are located within the Lakes District or what is known us the Main Ethiopian Rift (MER). The selected lakes are Ziawy and Abijata within the Ziway-Shala basin and Beseka far to the north in the Awash river basin (Table 1, Figure 1). Many of the lakes are located within a closed basin fed by perennial rivers. The major rivers are Awash, Meki-Katar and Dijo feeding lakes Abhe, Ziway and Shala respectively. The terminal lake Abijata is feed by the Bulbula and Horakelo rivers, which are outflows of lakes Ziway and Langano respectively.

The closed lake Beseka is situated at the junction of the MER and the Afar Depression close to the southern end of the Awash National Park, which is one of Ethiopia's main parks having many interesting animal species (Figure 1B). It is used only for local recreation, fishing and animal drinking. The lake level was raised by more than 4 m over the last 40 years (Ayenew, 2004). The average annual lake level rise has been estimated to be 0.12 m (MWR, 1999). Abadir irrigated farm (the source of large groundwater input to the lake) and the Awash river are located close to it. Lake Abijata is a closed shallow highly productive alkaline lake whose muddy shore supports a wealth of bird life, almost unequalled perhaps in the whole of Africa, as such it is of great biological importance. Recent development schemes, such as pumping of water from the lake for soda extraction, and the utilization of water from feeder rivers and lake Ziway for irrigation has resulted in rapid reduction in size and level (Legesse *et al.,* 2003; Ayenew, 2004). Lake Ziway is an open lake connected with the terminal lake Abijata by the Bulbula river. It is the largest lake in the region. Large rivers originated from the highlands feed this lake. For the last few decades the level of Ziway has declined owing to diversion of water for irrigation from two main feeder rivers (Katar and Meki) and direct pumping of lake water. The lake is home for many endemic birds and a wide variety of wild animals. Currently, Ziway is one of the main sources of commercial fish farming in Ethiopia. There is also excessive expansion of commercial irrigation activity for floriculture around the lake. Due to the excessive pumping, the once big out flowing perennial Bulbula river (with annual discharge of 48 million cubic meter) is now completely dry

Table 1.Basic hydrological data of the lakes.

Lake	Altitude (m.a.s.l)	Lake area (km^2)	Catchment area (km^2)	Maximum depth (m)	Mean depth (m)	Volume (mcm)	Salinity (g/l)
Beseka	952.4	40	401.5	-	6	-	-
Ziway	1636	440	7380	89	25	1466	0.349
Abijata	1580	180	10740	14.2	7.6	957	16.2
Langano	1585	230	2000	47.9	17	3800	1.88
Shala	1550	370	2300	266	8.6	37000	21.5

(Source: Wood and Telling, 1988; Ayenew, 1998; Tessema, 1998).

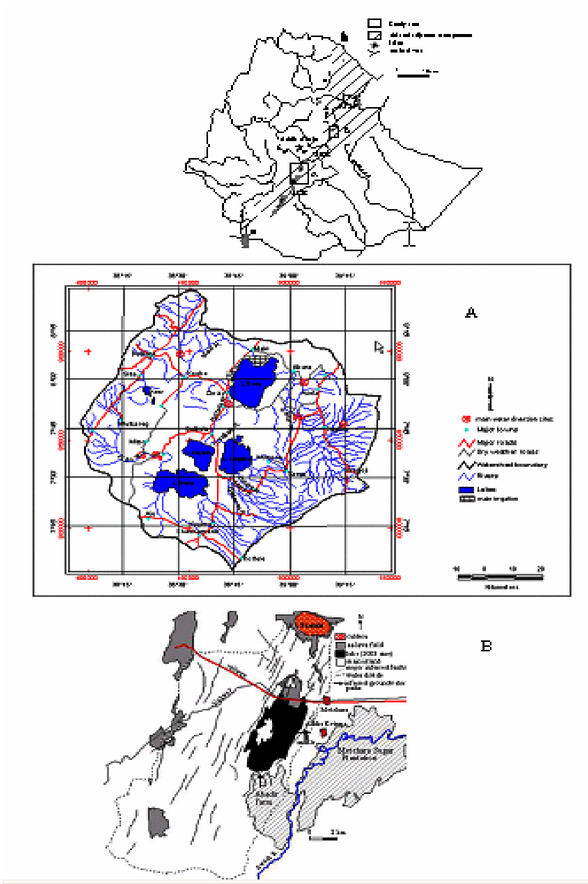


Figure 1. Location map: (A) Ziway-Shala basin and, and (B) Lake Beseka area.

Results

1. Lake Level Changes - Figures 2 and Figure 3 show the temporal variation of the levels of the lakes established based on monthly lake level records. For comparison, the level of lakes is shown with the climatic elements recorded in the region for the same period. The trend of lake levels in the Ethiopian rift is highly variable (Ayenew, 2004). The most drastic changes have been observed in lakes Abijata and Beseka.

The reduction of the level of Abijata is clearly visible from reconstructed shorelines (Figure 2A). The maximum reduction in the level of lake Abijata coincides with the time of large-scale water abstraction for soda production and for irrigation from lake Ziway since the 1980s. The shallower depth and the terminal position of lake Abijata, makes it more susceptible to changes in climate and input from precipitation and river discharge. As a closed lake, the only significant water loss is through evaporation. Groundwater flow model simulations indicate negligible groundwater outflow from lake Abijata (Ayenew, 2001). Generally changes in lake level and volume reflect and amplify the changes in inputs from rainfall and rivers. However, recent lake water pumping for soda extraction, and the utilization of water from feeder rivers and lake Ziway for irrigation has resulted in rapid shrinkage. In 1998, annual artificial water evaporation for soda extraction from Abijata was estimated at 13 million cubic meter (mcm) (Ayenew, 1998). In April 2005 (field observation) the size of the lake has gone down to 176 km^2.

The fluctuation of Abijata follows the same trend as Ziway, with an average time lag of about 20 days. Any abstraction of water in the Ziway catchment results in a greater reduction in the level of Abijata than in that of Ziway. Over the past three decades, the depth reached a maximum of 13 m in 1970–1972 and 7 m in 1989. These extreme drops in levels correspond to water volumes of 1575 and 541 mcm, and lake surface areas of 213 and 132 km^2 respectively. Before 1968, lake level variations, reconstructed from different sources (Benvenuti *et al*., 1995; Ayenew, 1998), showed inter-annual fluctuations of the same order of magnitude, with, for example, a high level in 1940 and 1972, a low level in 1965 (inferred from aerial photographs) comparable to that of 1989, and a level further reduced in 1967 (photo) and 1994 (field checks). The range of lake level fluctuations in Ziway is lower than Abijata, since wide and shallow lakes with an outlet do not usually show a large range of seasonal lake level changes. Referring to Figure 2 the lowest level of Ziway was recorded in June 1975 (0.13 m) and the maximum in September and October 1983 (2.17 m). However, for the last three years of the late 1970s and early 1980s the level was slightly lower due to the dry years of the 1970s.

Air photos taken at different times have shown that the area covered by lake Beseka was about 2.5 km^2 in the late 1950s; currently the total area is a little above 41 km^2. The level of the lake has risen by 4 m over two decades (1976-1997). The starting time of expansion is the early 1960, which is beginning of large-scale irrigation in the area. The main changes in the water balance of the lake come from groundwater inputs, which is related to the increment of recharge from the irrigation fields and due to the rise of the Awash river water level after the construction of the Koka dam located some 152 km upstream (MWR, 1999). Hence, the regulated flow has become a source of continuous recharge to groundwater ultimately feeding the lake.

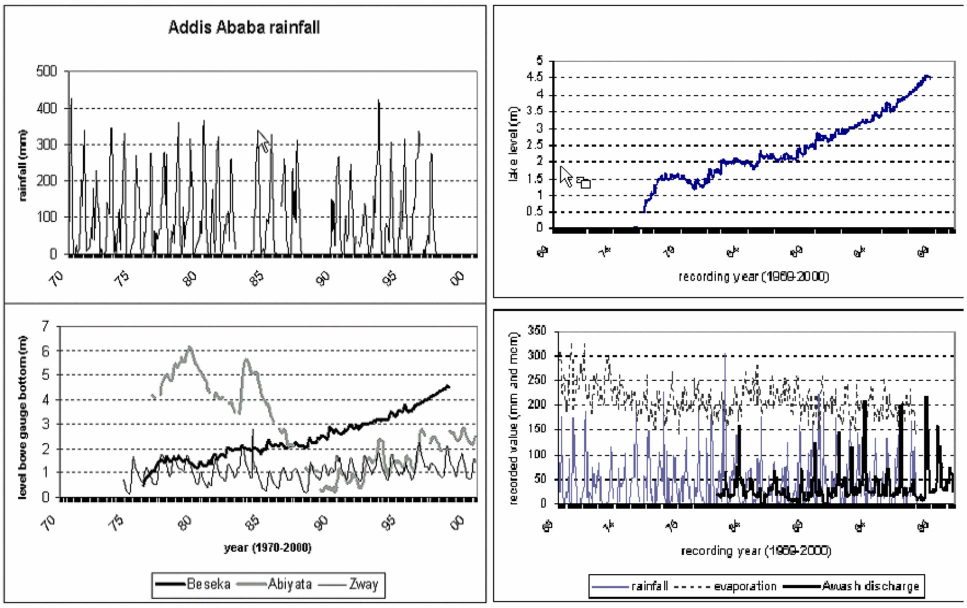


Figure 2. Lake level changes and variability of climatic components.

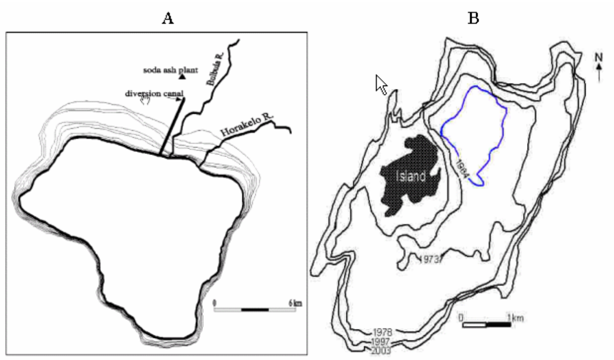


Figure 3. Temporal variation of lake size. (A) Shoreline position of Lake Abijata at different times. Numbers in parentheses indicate elevation above mean sea level. The outer boundary represents the 1971 shoreline (1580 m), and then in order of decreasing elevation: 1983 (1578.8 m), 1984 (1578.5 m), 1976 (1578 m), 1985 (1576.9 m), 1996 (1577 m), 1997 (1576.9 m), 1995 (1576 m) and 1967 (1575 m). The inner thick shoreline is the present day (2003) average lake level (1575.2 m) which is modified from Ayenew (1998). (B) The expansion of lake Beseka at different times (numbers with contours indicate the extent of the lake at that year).

2. Water quality changes - The change of lake levels is well reflected in the temporal variations of the ionic concentration of the lake waters. Water input–output relationships are the dominant feature of the status in the salinity series of the rift lakes (Wood and Talling, 1988). If the total outflow exceeds the inflow, with time, a lake becomes more saline. The extent of ionic enrichment depends on the lapse of time since the system became closed and on the changing rate of abstraction and evaporation over time. Compilation of the sparse chemical data available since 1926 and chemical analysis since 1995 (Ayenew, 1998) has revealed a considerable increase in the total dissolved solids of lake Abijata (Table 2). Between 1926 and 1998, the salinity fluctuated more than 2.6 times (from 8.1 to 26 mg/l), the alkalinity changed from 80 to 326 mg/l, and pH varied between 9.5 and 10.1 in lake Abijata. The conservative anion chloride showed a two-fold increase over 42 years (Omer-Cooper, 1930). The dominant cation, sodium, increased more than three-fold. Between 1984 and 1991 the sodium chloride levels of the lake water increased from 0.25 to 0.7 mg/l, sodium carbonate increased from 0.44 to 1.24 mg/l and sodium fluoride from 0.02 to 0.05 mg/l. The salt concentration in the lake has also increased drastically (Ayenew, 2004). Lake Beseka presents a completely different hydrochemical picture; from an extremely alkaline water body it has changed to a nearly fresh lake over the last 40 years. The electrical conductivity has reduced from 74170 μS/cm to 7440 μS/cm between 1961 and 1991 (Kebede *et al*., 1996) corresponding to a change in size from 3 to 35 km^2 (Table 3).

Source	**Tame of sampling**	**Salinity (g/l)**	**Alkalinity**	**Ca**	**Mg**	**Na**	**K**	**Cl**	**SO_4**	**Total cations**
Omer-Cooper (1930)	Nov, 1926	8.1	80	0.5	0.8	125	-	42	-	
Loffredo and Maldura (1941)	Apr. 1938	8.4	-	0.4	0.5	130	1.9	42	1.4	133
De Filippis (1940)	1939	-	81	0.2	0.1	140	10.3	40		150
Talling and Talling (1965)	May-61	19.4	210	<0.15	<0.6	277	8.5	91	15	285
Wood and Talling (1988)	Jan-76	16.2	166	<0.1	<0.1	222	6.5	51	22.5	228
Von Damm and Edmond (1984)	Nov. 1980	12.9	138	0.1		194	4.9	54	0.3	199
	Nov. 1980	-	180	<0.01	<0.01	231	6.9	82	4	238
	Oct. 1981	21	297	-	-	378	9.9	121	5.7	388
	Mar. 1991	26	326	0.1	-	416	9.7	88	24	425

Table 2. Temporal changes of the chemistry of lake Abijata (ions in mg/l).

Source	Year of sampling	EC	Na	K	Ca	Mg	HCO_3+CO_3	Cl	SO4	pH	Sampling season
Taling and Talling (1965)	1961	74170	774	10	<0.15	<0.6	580	154.8	98	10	-
Kebede et al. (1996)	1991	7440	79	2	0.1	-	46	13	12	9	dry
Tessema (1998)	1997	7384	1850	55	2.2	0.4	1989	589	498	9.5	dry
MWR (1999)	1998	5230	1500	56	1.6	0.49	1756	505	440	9.44	wet
This study	2002	6170	1960	67.6	1.3	17	2838	525	525.8	9.08	dry

Table 3. Temporal changes in the ionic concentration of lake Beseka (ions in mg/l).

3. Effect on biodiversity - The land area around the lakes has very rich biodiversity resource, which is composed of plant, invertebrate and vertebrate species but is mainly known for its bird species diversity (Zinabu and Elias, 1989). Lake Langano serves as a stop over for globally threatened species such as Lesser kestrel, Pallid herrier, and Lesser flamingos and other birds like pelicans, plovers, terns, eagles. The land around lakes Abijata and Shala has been designated as Abijata-Shala lakes National Park (ASNP) in 1970 mainly for the protection of its diverse aquatic bird fauna. According to some estimates, the park provides temporary or permanent home to over 400 bird species, which amounts to almost half the number recorded for the whole country (OEPO, 2005). It is because of its geographical position that the ASNP provides wintering and maintenance station for such a large number of terrestrial and aquatic birds. On account of the high aquatic bird population that it harbours, Lake Abijata has been proposed as an international Ramsar site. In addition to its bird fauna, the savanna habitat provides homage to about 31 species of mammals of which the most commonly spotted are Grant's gazelle, Oribi warthog and the Golden Jackal. Currently the park is not any more park; it is converted to farm land.

In Lake Ziway nutrients such as phosphate, nitrate and silicate have increased in recent years. Zooplankton and fish have remained stable, while the benthos show changes in composition and abundance. Lake Ziway has a total of 122 phytoplankton species of which 50 species are blue-green, 41 green algae and the rest 31 diatoms (Seyoum Mengistou *et al.,* 1991). The same study revealed that as a result of water level changes the aquatic biota has shown considerable changes over the years in lake Abijata as compared to Ziway. The livelihood of the fish and flamingos may be at stake as the lake is a feeding and breeding ground for one of the richest lesser Flamingo birds in Africa. The alkaline lake water of Abijata is conducive for mass culture of the blue-green algae which is essential for fishes which in turn sustains many fish-eating birds.

The high shoreline development of Lake Ziway also implies that the littoral should harbor rich macrophyte growth, which offers refuge to juvenile fish, which are observed in large numbers in the reed belts. The submerged macrophyte is present in some shore areas, especially where human activities are intensive. Lake Abijata has poor littoral development and macrophytes are lacking around most of its shore. The riparian flora of Lake Abijata has shown progressive deterioration because of human intrusions such as farming, cattle ranching and wood felling for charcoal.

4. Salinization **-** Salinization is one of the most critical problems in the Awash valley irrigation fields around and in the vicinity of Lake Beseka; and recently there are manifestations in the irrigation farms around lake Ziway. The most affected field is the Melka Sedi-Amibara irrigation project in the Middle Awash valley north of Lake Beseka and the Metahara farm. The high soil salinity levels in these farms are related to groundwater level rise due to over irrigation; which led to capillary rise. The rise is more pronounced in the banana fields, which use basin irrigation. In the shallow piezometric systems over-irrigation brings about capillary rise and contributes significantly to the salinization process as observed from 71 piezometer readings in Amibara irrigation field since 1984. Groundwater modelling was made to study and delineate the most affected areas (Hailu *et al*., 1996). The result indicates the presence of wide cone of depressions and domes representing local abstraction from wells and over-irrigation in some of the banana fields respectively. There is substantial area with high-rise in groundwater level, which led to capillary rise and salinization; particularly in plots with basin irrigation with no proper drainage system.

Conclusion

The studied lakes and their tributary and spill rivers are unique and vital resources within the broader Rift Valley Lakes system. They provide invaluable benefits in areas of agriculture, recreation, drinking water, industrial development, fish and wildlife habitat and biodiversity. They have influenced the development of the region and continue to be the defining natural features for the region. Protecting these lakes and their environs is the quest of the time that needs urgent intervention. Perhaps the greatest challenge for effectively managing the future of the lakes and their catchments will be to manage the complex institutional environment and to facilitate an efficient, credible and focused program for balancing our continued ability to benefit from the resources while preserving their chemical, physical and biological integrity.

The sustainable use of the water and land resources utilization demands a comprehensive water management and planning strategy requiring the process of protecting and developing the resources in a broad, integrated, and foresighted manner. In practice, this is a complicated endeavour, since comprehensive water management involves a number of functions that are closely related but which are carried out by different agencies and organizations. The purpose of these functions is to identify alternative courses of action to protect and develop the water resources and develop comprehensive water management plan.

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References

Ayenew, T., 1998. *The hydrogeological system of the lake district basin, Central Main Ethiopian Rift*. Published PhD thesis, Free University of Amsterdam, The Netherlands. 259p.

Ayenew, T., 2001. Numerical groundwater flow modeling of the Central Main

Ethiopian Rift lakes basin, SINET: *Ethiopian Journal of Science* 24(2):167-184.

Ayenew, T., 2004. Environmental implications of changes in the levels of lakes in the Ethiopian rift since 1970. *Reg.Environ. Change*.4:192-204.

Baxter, R.M.; Prosser, M.V., Talling, J.F. and Wood, R.B., 1965. *Stratification in tropical African lakes at moderate altitudes* (510 - 520). Limnol. Oceano.10:511-520.

Benvenuti, M., Dainelli, N., Iasio, I., Sagri, M. and Ventra, D., 1995. Report on EEC funded project " Land resources inventory, environmental change analysis and their applications to agriculture in the Abaya lakes region" *Report no.4*, Univ. of Florence, Italy. pp. 6-27.

Dagnachew Legesse, Christine Vallet-Coulomb, Francoise Gasse, 2003. Hydrological response of a catchment to climate and land use changes in Tropical Africa: case study South Central Ethiopia. *Journal of Hydrology* 275: 67–85.

De Filippis, N., 1940. *Condizioni chimiche del lago Hora Abiata*. Boll. Idrobiol. Africa Orientale Italiana 1: 77-79.

Hailu, D., Hess, M., Ayenew, T., 1996. *The problem of high-rise groundwater in Amibara irrigation project, Middle Awash basin. Ethiopian Science and Technology Commission*. Unpublished report, Addis Ababa.

Kebede, E., Gebremariam, Z. and Ahlgren, A. 1996. The Ethiopian rift valley lakes. Chemical characteristics along a salinity-alkalinity series. *Hydrobiologia*, 288: 1-12

Loffredo, S. and Maldura, C.M.,1941. Resulatati delle ricerche de chimica limnologica sulle acque dei laghi dell'Africa orienntale Italiana esplorati della Missione ittiologica. *In.* Piccioli, A., ed. *Esplorazione dei laghi della Fossa Galla.* Collezione scientifica e documenatari dell'Africa Italiana III, Vol.I, 181-200.

MWR, 1999. *Study of lake Beseka. Main Report (volume 1)*. Ministry of Water Resources. Addis Ababa, Ethiopia. 203 pp.

OEPO, 2005. *A review of the current status and an outline of a future management plan for lakes Abijata and Ziway. Oromia Environmental Protection Office*. Unpublished Technical Report. Addis Ababa, Ethioipia. 80 pp.

Omer-Cooper, J., 1930. *Dr. Hugh Scott's expedition to Abyssinia. A preliminary investigation of the fresh water fauna of Abyssinia.* Proc. Zool. Soc. Lond. (1930), 195-591.

Seyoum Mengistou, Green, J. and Fernando, C.H.,1991. Species composition Distribution and seasonal dynamics of Rotifera in a rift-valley lake, Awassa, Ethiopia. *Hydrobiologia*. 209: 203-214.

Talling, J.F. and Talling, I.B.,1965. The chemical composition of African lake waters. Int.Rev.Ges. *Hydrobiolgia.* 50:421-463.

Tessema, Z., 1998. *Hydrochemical and water balance approach in the study of high water level rise of lake Beseka*. M.Sc thesis. The University of Birmingham. 90pp.

Von Damm, K.L. and Edmond, J.M., 1984. Reverse weathering in the closed basin lakes f the Ethiopian Rift. *Amer. J. Sci.* 284: 835-862.

Wood R.B. and Talling, J.F.,1988. Chemical and Algal relationship in a salinity series of Ethiopian inland waters. *Hydrobiologia* 158:29-67.

Zinabu, G.M., and Elias, D., 1989. Water resources and fisheries management in the Ethiopian rift-valley lakes. *SINET: Ethiop. J. Sci.* 12:95-109.

Molecular markers of anthropogenic influence on Plitvice Lakes, Croatia

Ahel Marijan[1] and Senka Terzic

Center for Marine and Environmental Research, Rudjer Boskovic Institute, Zagreb, Croatia
[1]Rudjer Boskovic Institute, Bijenicka c. 54, 10000 Zagreb, Croatia,
Email: ahel@irb.hr

Abstract

Dated recent sediments (0-30 cm) from the pristine karst lakes situated in the Plitvice National Park, Croatia, were analysed for different organic compounds in order to determine the possible impact of anthropogenic activities on that protected ecosystem. The Plitvice Lakes are situated in a sparsely populated area of the Northwestern Dinarides, but during the last several decades, they have been exposed to intensive tourist traffic, reaching 1 million visitors per year in the late 1980s. The selected molecular markers encompassed various types of hydrocarbons and one major class of surfactants. For hydrocarbon analyses, sediment samples were extracted by dichloromethane, separated into aliphatic and aromatic fractions and subsequently analysed by computer-assisted high-resolution gas chromatography mass spectrometry (HRGC/MS). Analysis of surfactant-residues was performed by liquid chromatography-tandem mass spectrometry (LC/MS/MS) after an extraction of sediment samples with methanol. Based on characteristic molecular markers, four major sources of biogenic and/or xenobiotic organic compounds into the lakes were recognised: a) phytoplankton, b) higher terrestrial plants c) oil pollution and/or pyrolytic processes and d) detergents. The profile of phytoplankton marker, n-heptadecane, indicated an increasing eutrophication in the past 100 years, while the maxima of the specific biomarkers of higher plants coincide with the periods of increased timber cutting and sawmill activities. The profiles of pyrolytic polycyclic aromatic hydrocarbons suggested a decreasing trend since the area was proclaimed a National Park. In contrast, the concentration of some xenobiotic chemicals, in particular residues of linear alkylbenzene sulphonates, reached rather high values in the most recent sediments, reflecting an increasing input of detergent-derived chemicals via untreated wastewaters from the hotels situated near the shoreline of the Plitvice Lakes.

Key words: molecular markers, hydrocarbons, surfactants, lake sediments

Introduction

Molecular characterisation of sedimentary organic matter is widely accepted as a valuable method of studying sources and transformations of organic carbon in the aquatic environment. Among various specific organic compounds, hydrocarbons have proved to be excellent marker-compounds capable of indicating contributions from biogenic as well as from anthropogenic sources. Moreover, it has been shown that the composition of sedimentary biogenic hydrocarbons can be used as a clue to assess the inputs from different autochthonous and/or allochthonous sources. Alkanes and alkenes synthesized by planktonic and benthic organisms (Blumer *et al.,* 1971) have markedly different composition to those deriving from higher terrestrial plants (Eglington & Hamilton, 1967). In addition, a significant percentage of aliphatic hydrocarbons, found in recent sediments, are derived from petroleum hydrocarbon pollution (Blumer & Sass, 1972). Polycyclic aromatic hydrocarbons (PAH) show also a widespread occurrence in aquatic sediments (Laflamme & Hites. 1978). The unsubstituted PAH are released into the environment by various pyrolytic processes, including fossil fuel combustion as a major source (Laflamme & Hites, 1978; Wakeham *et al.,* 1980a). As a consequence, their abundance in the older (deeper) sediment layers significantly decreases. In contrast, the dominant PAH in the deeper sediments are those formed by early diagenesis of organic matter buried in the sediments (Wakeham *et al.,* 1980b; Tan & Heit, 1981).

As opposed to hydrocarbons, synthetic surfactants are xenobiotic compounds and therefore directly reflect the anthropogenic pressure on a given ecosystem. Linear alkylbenzene sulphonates (LAS) are the most popular surfactant class with the total annual world consumption exceeding 2000000 t. The distribution of LAS in lake sediments proved to be an excellent marker of the input of detergent chemicals via municipal wastewaters and was successfully applied to document usage patterns and wastewater treatment practices in a given area (Reiser *et al.,* 1997).

The aim of this work was to investigate the influence of different anthropogenic activities and processes on the two pristine karst lakes, situated in the area of the Plitvice National Park, Croatia, using molecular marker approach.

Materials and methods

Study area and sampling

The Plitvice National Park, located in a sparsely populated area of the northwestern Dinarides, central Croatia, consists of 16 karst lakes interconnected by numerous cascades and waterfalls, formed by travertine barriers. The details on the chemistry and biology of the lakes have been presented elsewhere (Srdoc *et al.,* 1985). Sediment cores were taken in 1989, 1990 and 2004 from the two largest lakes, Lake Prosce (sampling depth 32 m) and Lake Kozjak (sampling depth 21.5 m). There is a difference between the two lakes with respect to their exposure to modern anthropogenic sources. The upper Lake Prosce is situated in an area away from the major roads and tourist routes, whilst the lower Lake Kozjak is exposed to a more intensive tourist traffic (the number of visitors in late 1980s was estimated at 1 million per year) with 3 hotels located near its shore. Only few electric-powered and rowing boats are allowed on the lakes. Sedimentation rates in the recent sediment layers of the Lake Prosce and Lake Kozjak were estimated at 1.6 and 0.8 mm/year, respectively (Srdoc *et al.,* 1992).

Sediment cores (30 cm) of lake marl, covering a period of approximately 200 years, were retrieved by scuba-diver using a 100 mm i.d. plastic coring device. For sectioning, the cores were partially thawed, extruded with a piston and cut into 1-5 cm layers with a metal knife.

Analyses

The analytical methodology for the determination of hydrocarbons, used in this study, has been described in detail by Giger and Schaffner (1978). Briefly, air-dried sediment samples (3.5-9 g) were Soxhlet extracted with methylene chloride, fractionated into an aliphatic and aromatic fraction and analysed by high-resolution gas chromatography/ mass spectrometry.

Determination of linear alkylbenzene sulphonates was performed using high-performance liquid chromatography with electrospray ionization tandem mass spectrometry (LC/MS/MS). Briefly, air-dried sediments were extracted with methanol using an ultrasonic bath. After centrifugation, an aliquot of the clear extract was directly injected onto a LC/MS/MS system equipped with C_{18} reversed-phase column. The detection of LAS was performed in the negative ionization mode using specific transitions for each LAS-homologue (Petrovic *et al.,* 2002).

Results and discussion

Four major sources of biogenic and/or xenobiotic organic compounds into the Plitvice Lakes were recognised using hydrocarbons and synthetic surfactants as specific molecular markers: a) phytoplankton, b) higher terrestrial plants c) oil pollution and/or pyrolytic processes and d) detergents.

The analysis of sediments included three groups of molecular marker-hydrocarbons: n-alkanes, diagenetic aromatic hydrocarbons and unsubstituted polycyclic aromatic hydrocarbons. The concentrations of the most hydrocarbon classes are significantly higher in the Lake Prosce than in the Lake Kozjak. The concentrations of n-alkanes in both lakes were approximately one order of magnitude higher than the concentrations of PAH. The most prominent individual alkanes were *n*-heptadecane and odd-numbered higher alkanes (nC_{27}-nC_{31}), indicating two major sources of hydrocarbons into the lakes: phytoplankton and higher plants. The carbon preference indices (CPI) for the higher *n*-alkanes (>nC_{25}) varied in relatively narrow ranges of 5.97-9.49 and 6.92-8.40 for Lakes Kozjak and Prosce, respectively. Such high values of CPI clearly indicated that the major portion of the determined hydrocarbons was of terrestrial origin (Giger *et al.,* 1980).

Sediment depth profiles for the two prominent n-alkanes, *n*-heptadecane (n-C_{17}) and *n*-nonacosane (n-C_{29}) are presented in Figure 1. It is known that n-C_{17} represents a very good indication of autochthonous planktonic input, particularly by diatoms (Giger *et al.,* 1980). Consequently, the increasing levels of this compound towards the youngest sediments were attributed to the enhanced primary productivity of the lakes, i.e. to eutrophication. The depth profile of n-C_{17} in the Lake Prosce shows the main maximum in the layer at 1-3 cm, which is in a good agreement with number of diatomophyte residues in that layer (Srdoc *et al.,* 1992). This would probably suggest that the measures undertaken in the past 30 years to protect the area, after it had been proclaimed a National Park, were efficient in reducing accelerated eutrophication in the Lake Prosce. However, when concentration of n-C_{17} was normalised against the concentration of *n*-octadecane (n-C_{18}), a compound with a very similar chemical structure but of rather different origin, to compensate for possible biodegradation losses (Giger *et al.,* 1980), the depth profiles of the n-C_{17}/ n-C_{18} ratio in both lakes clearly revealed that an enhanced eutrophication started around 1900 and that it significantly accelerated towards the youngest sediments (Figure 2).

In spite of the increased eutrophication, the concentration of n-C_{29} in the Lake Kozjak and Lake Prosce is significantly higher than that of n-C_{17} (Figure 1), indicating the importance of terrestrial plants as contributors to the organic matter input into the Plitvice Lakes. As can be seen in Figure 1, this predominance was reasonably constant in last 200 years. The lakes are surrounded by forest and the largest part of this terrestrial plant input has purely natural origin. However, the input from this particular source was influenced by man's activities such as timber cutting and sawing, which in some periods significantly increased the input of higher plant debris, particularly sawdust, into the lakes. Some features of the concentration profiles of n-C_{29} in Figure 1 can be related to timber cutting and sawmill activities in the area, which started around 1810 were particularly intensified around 1900.

In addition to higher homologues of *n*-alkanes, there is another group of hydrocarbons, which indicated the terrestrial input by higher plants. Diagenetic PAH deriving from pentacyclic triterpenes of the amyrin type were found to be the dominant PAH in deeper sediment layers.

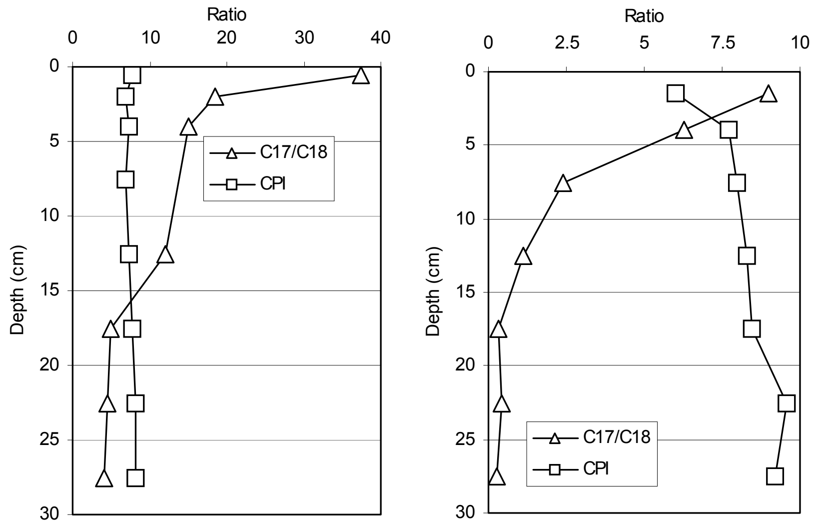


Figure 1 Depth profiles of *n*-heptadecane (n-C_{17}) and *n*-nonacosane (n-C_{29}) in sediment cores from the Lake Prosce (left) and Lake Kozjak (right).

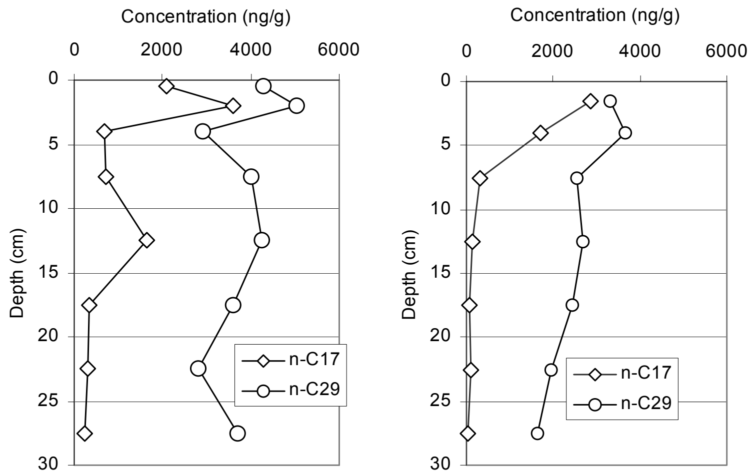


Figure 2 Depth profiles of the *n*-heptadecane/*n*-octadecane ratio (n-C_{17}/n-C_{18}) and carbon preference indices (CPI) in sediment cores from the Lake Prosce (left) and Lake Kozjak (right). CPI = 2 x (n-C_{27} + n-C_{29})/(n-C_{26} + 2 x n-C_{28} + n-C_{30}).

The concentration profile of the compound, which was identified as *3, 4, 7*-trimethyl-*1, 2, 3, 4*-tetrahydrochrysene (TTHC) (Wakeham *et al.,* 1980b), is presented in Figure 3, together with the most common diagenetic PAH, perylene (Waheham *et al.,* 1980b). As can be seen the concentration of perylene increases gradually with the depth down to the layer, which is approximately 150-200 years old. Relatively constant concentration levels in the older sediments from the Lake Kozjak indicate either a lower input of the perylene precursors in that time period or that perylene has been consumed by further transformations. Contrary to that, TTHC shows rather different concentration profile, with a significant decrease in sediment layers older than 150 years.

Very useful information about the man's activities around the Plitvice Lakes was obtained by determining those PAH which have predominately anthropogenic origin (Wakeham *et al.,* 1980a). The concentration profiles of 4 selected PAH, representing compounds with 3-5 rings are presented in Figure 4. Apparently, the concentration profiles for most of the individual PAH are rather similar, suggesting that they have had common origin. The highest concentrations were observed for high-molecular weight PAH, such as indeno(cd)pyrene, reaching up to 200 ng/g. The concentrations are, generally, higher in the Lake Prosce, suggesting more intensive activities in that part of the area and/or faster sedimentation rate. Namely, due to the closest proximity of the two lakes, the background contamination governed by atmospheric transport should be assumed equal. Consequently, the difference must be attributed to direct inputs into the particular lake. Lake Prosce receives direct inputs from several small settlements via two major creeks Bijela Rijeka and Crna Rijeka. The Lake Kozjak, which is situated further downstream in the cascade hydrological system of the Plitvice Lakes, receives waters from which part of waterborne PAH must have been removed by association with particles and their subsequent sedimentation in the upper lakes. As expected, the lowest concentrations of anthropogenic PAH were determined in the deepest sections of the sediment cores. The concentrations found in the deepest layer of the sediment core from the Lake Kozjak (deposited before the year 1700) can be considered indicative of pre-industrial background levels. It is interesting to note that maximum PAH concentrations were not observed at the top of the sediment cores, which indicates that the strict protection measures in the area of the National Park, including the new regulation of the road traffic, were efficient in reducing the input of this type of the pollution. The maximum of the PAH contamination in the sediment sections of 10-15 cm in the Lake Prosce and 5-10 cm in the Lake Kozjak, coincides with the period of intensive timber cutting and sawmill activities around the year 1900. The second peak dated around 1960 is apparent only in the Lake Prosce, since time resolution of core sampling for the Lake Kozjak was not sufficient. After 1960s most of the industrial activities around the Plitvice Lakes have ceased, and the road traffic diverted further away from the lakes. Consequently, the concentrations of the anthropogenic PAH sharply decreased to values similar as those found in sediment sections as old as 150 years.

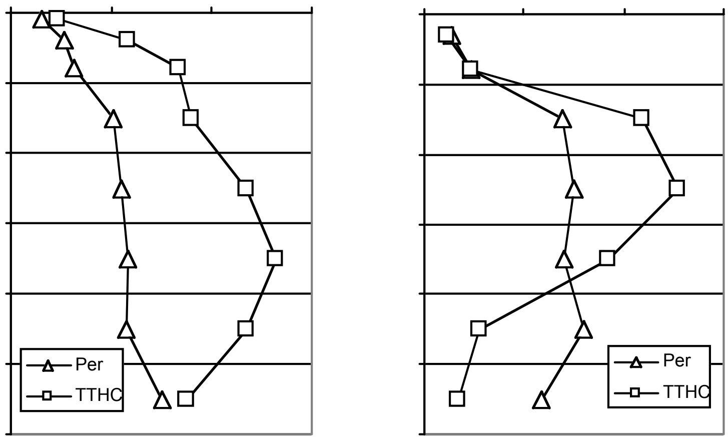


Figure 3 Depth profiles of diagenetic polycyclic aromatic hydrocarbons (PAH) in sediment cores from the Lake Prosce (left) and Lake Kozjak (right): perylene (Per); *3,4,7*-trimethyl-*1,2,3,4*-tetrahydrochrysene (TTHC).

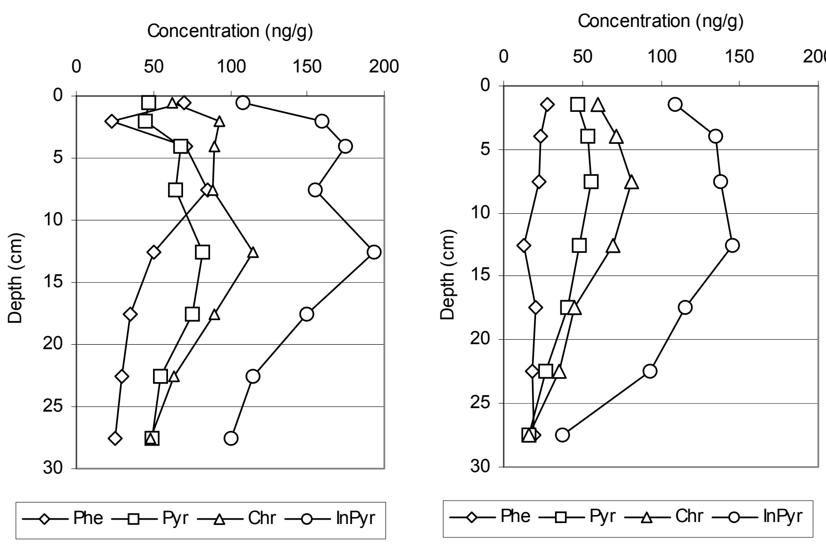
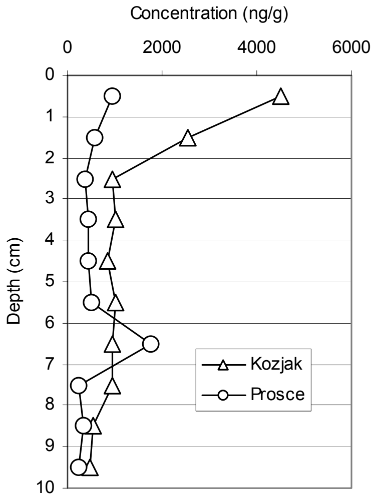


Figure 4 Depth profiles of pyrolytic polycyclic aromatic hydrocarbons (PAH) in sediment cores from the Lake Prosce (left) and Lake Kozjak (right). Phe: phenanthrene; Pyr: pyrene; Chr: chrysene; InPyr: indeno(c,d)pyrene.

The concentration profiles of LAS in Lake Kozjak and Lake Prosce (Figure 5) show some interesting features, which provide evidence for the impact of some specific sources on the vulnerable system of the Plitvice Lakes. As expected the maximum concentration was observed in the uppermost layer, indicating increasing usage of synthetic detergents in the last decades. It is interesting to note that the concentration of the total LAS in the Lake Kozjak in the upper sediment sections largely exceeds the concentration in the Lake Prosce. The explanation for this difference is input of untreated wastewaters from the hotels situated on the shore of the Lake Kozjak. The concentration profiles of individual LAS homologues are very similar and show predominance of C_{12} and C_{11} LAS, reflecting typical composition of biologically unaltered commercial LAS mixtures. While distribution of PAH indicated that rigorous measures undertaken in last 30 years were efficient in reducing hydrocarbon pollution, the occurrence of LAS in relatively high concentrations for a pristine area suggest that wastewaters significantly contributed to euthrophication and input of detergent-derived xenobiotic chemicals.



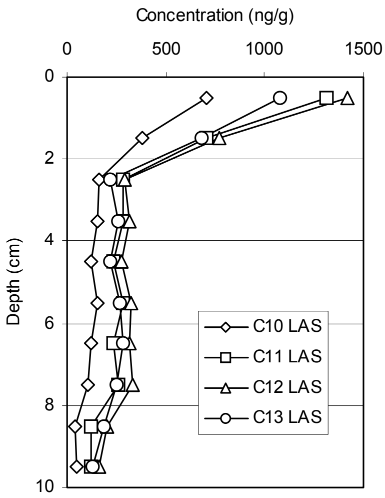


Figure 5 Depth profiles of linear alkylbenzene sulphonates (LAS) in sediment cores from the Lake Kozjak and Lake Prosce. left: total LAS concentration; right: concentration of individual oligomers in Lake Kozjak.

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References

Blumer M., Guillard R. R. & Chase T., 1971. Hydrocarbons of marine phytoplankton. *Mar. Biol.* 8, 183-189.

Blumer M. & Sass J., 1972. Indigenous and petroleum-derived hydrocarbons in an oil polluted sediment. *Mar. Pollut. Bull*. 3, 92-94.

Eglington G. & Hamilton J. R., 1967. Leaf epicuticular waxes. *Science* 156, 1322-1335.

Giger W. & Schaffner C., 1978. Determination of polycyclic aromatic hydrocarbons in the environment by glass capillary gas chromatography. *Anal. Chem*. 50, 243-249.

Giger W., Schaffner C. & Wakeham S. G., 1980. Aliphatic and olefinic hydrocarbons in recent sediments of Greifensee, Switzerland. *Geochim. Cosmochim. Acta* 44, 119-129.

Laflamme R. E. & Hites R., 1978. The global distribution of polycyclic aromatic hydrocarbons in recent sediments. *Geochim Cosmochim. Acta* 42, 289-303.

Petrovic M., Fernandez-Alba A.R., Borrull F., Marce R.M., Mazo E.G. & Barcelo D., 2002. Occurrence and distribution of nonionic surfactants, their degradation products, and linear alkylbenzene sulfonates in coastal waters and sediments in Spain. *Environ. Toxicol. Chem*. 21,: 37-46.

Reiser R., Toljander H., Albrecht A. & Giger W., 1997. Alkylbenzenesulfonates in recent lake sediments as molecular markers for the environmental behavior of detergent-derived chemicals. In: *Molecular markers in environmental geochemistry*. *ACS Symposium Series* 671, 196-212.

Srdoc D., Horvatincic N., Ahel M., Giger W., Schaffner C., Krajcar Bronic I., Petricioli D., Pezdic J., Marcenko E. & Plenkovic-Moraj A., 1992. Anthropogenic influence on the ^{14}C activity and other constituents of recent lake sediments: a case study. *Radiocarbon* 34, 585-592.

Srdoc D., Horvatincic N., Obelic B., Krajcar Bronic I. & Sliepcevic A., 1985. Calcite deposition processes in karstwaters with special emphasis on the Plitvice Lakes. *Carsus Iugoslaviae* 11, 101-204 (in Croatian, with extended abstract in English)

Tan Y. L. & Heit M., 1981. Biogenic and abiogenic polynuclear aromatic hydrocarbons in sediments from two remote Adirondack lakes. *Geochim. Cosmochim. Acta* 45, 2267-2279.

Wakeham S. G., Schaffner C. & Giger W., 1980a. Polycyclic aromatic hydrocarbons in recent lake sediments. I. Compounds having anthropogenic origins. *Geochim. Cosmochim. Acta* 44, 403-413.
Wakeham S. G., Schaffner C. & Giger W., 1980b. Polycyclic aromatic hydrocarbons in recent lake sediments. II. Compounds derived from biogenic precursors during early diagenesis. *Geochim. Cosmochim. Acta* 44, 415-429.
Youngblood W. W. & Blumer M., 1975. Polycyclic aromatic hydrocarbons in the environment: homologous series in soils and recent marine sediments. *Geochim. Cosmochim. Acta* 39, 1303-1314.

The status of persistent organic pollutants in Lake Victoria catchment

Madadi O.V, S. O Wandiga, and I. O. Jumba

Department of Chemistry, College of Biological and Physical Sciences, University of Nairobi, Box 30197, 00100, Nairobi Kenya.
Email:madadivin2002@yahoo.com

Abstract

The use of most organochlorine pesticides has been banned or restricted in the republic of Kenya under the Rotterdam and Stockholm convention due to high levels of persistence in the environment and toxicity to nontarget organisms. Studies conducted in some parts of the country have revealed that residue levels of these compounds are still in the environment. However, the residues of these compounds have not been exhaustively studied in the Lake Victoria catchment area. This study was set to investigate the residues levels of p,p-DDT, o,p'-DDE, p,p-DDD, g-HCH, D-HCH, a-HCH, Aldrin, and Dieldrin, in water samples from Lake Victoria catchment. Samples were collected during the short rain, dry and wet seasons and analysed using gas chromatography equipped with electron capture detector. Residue levels ranging from below detection limit (BDL)-0.44 µg/l in river Nzoia water, between BDL-0.34 µg/l in river Sio water, BDL- 0.26 µg/l in water from Sio Port, and between BDL-0.31 µg/l in water from lake Victoria at Marenga Beach were detected.

Key words: Organochlorine, residues, Lake Victoria

Introduction

The use of chemical pesticides is still indispensable in Kenya, due to the hot and humid tropical environmental conditions that are conducive to the development of a myriad of pests, weeds and disease vectors. The public health sector in Kenya also heavily depends on pesticides to control vector-borne diseases such as malaria, sleeping sickness, biliharziasis and filariasis through pesticide spray programs aimed at controlling disease vectors such as mosquitoes, tsetse-flies and water snails. WHO programs to eradicate these pests in Mwea Tabere settlement scheme, Kano plain and Lambwe Valley succeeded to make them habitable using p,p'-DDT, dieldrin and endosulfan, as the major pesticides used in the control of mosquitoes and tsetse-flies. However, several chemical contaminants from the agricultural fields, comprising of pesticides and other agrochemicals have been reported in the drainage systems and are likely to jeopardise the quality of the water bodies that support the fishery industry and are used for domestic human consumption. The use of the pesticides poses a great challenge to the country to develop satisfactory techniques that can combine optimal agricultural productivity and environmental protection.

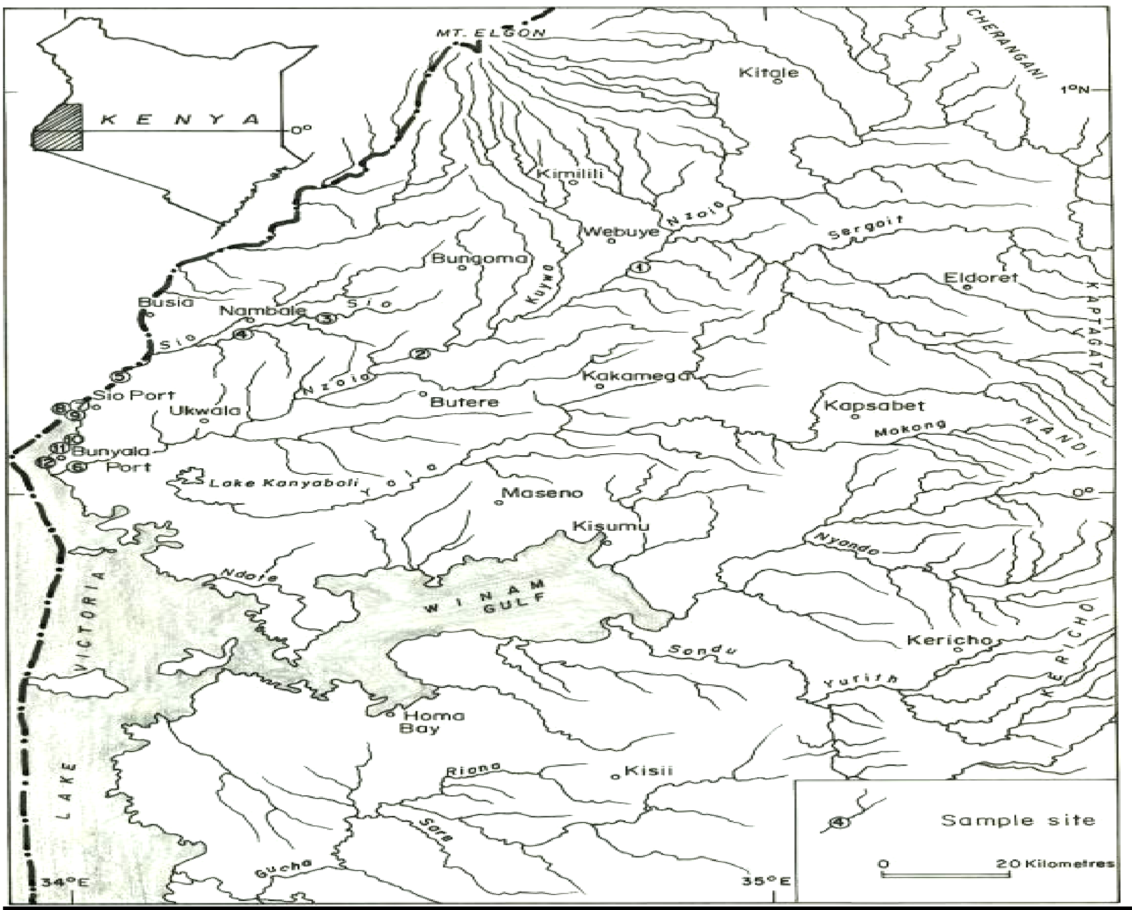
Earlier studies conducted by Mitema and Gitau (1990) detected low levels of -BHC, -BHC, aldrin, dieldrin, lindane, and p,p'-DDT in Nile perch from Lake Victoria. The p,p'-DDT and its metabolites formed the largest proportion of the organochlorine pesticide residues in the fish samples. The presence of these residues was attributed to the previous use of the pesticides in agriculture and aerial control of mosquitoes in the Lake Victoria region. Mugachia *et al.,* (1992b), showed presence of organochlorine pesticide residues in six species of fish from the Athi River estuarine. They reported presence of p,p'-DDE, p,p'-DDT, p,p'-DDD, -HCH, -HCH, heptachlor to o,p'-DDD in samples. Currently information on pesticide residues in the Lake Victoria catchment is fragmentary and inadequate. There is need for data on persistent organic pesticides in the drainage system of Lake Victoria for proper management of the lake water quality, and sustainability of the lake ecosystem. This study was set up to survey the levels of organochlorine pesticides in the Lake Victoria catchment comprising of Rivers Sio and Nzoia.

Materials and methods

High quality pesticide standards of aldrin, dieldrin, p,p'-DDT, o,p-DDE, p,p'-DDD, α-HCH, -HCH, and -HCH of purity over 99 % were used for identification and quantification of residues in the field samples. The pesticide standards were obtained from Dr EHRENSTORFER GmbH, (Ausburg, Germany). Other consumable chemicals were of analytical grade purchased from local suppliers. The field samples were collected from 12 points located on River Nzoia, River Sio and Lake Victoria. The sampling points along River Nzoia included Webuye (4), Mumias (5) and Port Victoria (6), whereas those along River Sio included Alatsi (1), Nambale (2) and Luanda (3). The points selected along Lake Victoria were distributed at Sio Port (7, 8 and 9) and Marenga Beach (10, 11, and 12) (Figure 1). Field sampling was done thrice, covering the short rain, dry, and wet seasons. The water was sampled by grab method into 2.5 L amber bottles, which had been pre-washed with distilled water and dried. Each water sample was treated with 1 g mercuric chloride, and mixed for 5 minutes to kill microorganisms that could degrade the pesticides. The sterilised water samples were kept in icebox containing wet ice during the sampling trip and later stored in a refrigerator at 4^0 C after sampling trip prior to extraction. Solvent-solvent extraction method was used in extraction of all the samples. 2.0 litres of water was transferred into a separatory funnel and pH measured. 50ml of 0.2 M disodium hydrogen phosphate buffer was added to the sample, and pH adjusted to 7 by adding drops of 0.1 N sodium hydroxide and HCL solutions. The neutralised sample was treated with 100 g sodium chloride to salt out the pesticides from the aqueous phase. 60 ml triple distilled dichloromethane was added and shaken for two minutes while releasing pressure. The sample was allowed to settle for 30 minutes to enhance separation of the phases. The organic layer was collected in 250 ml Erlenmeyer flask and stored at 4^0 C in a refrigerator. The extractions were repeated twice using 60 ml portions of dichloromethane, the extracts combined and cleaned by passing through florisil column. The clean extracts were concentrated on a rotar evaporator to near dryness and reconstituted in HPLC hexane to 5 ml. The final samples were analysed by a Varian Chromapack CP-3800 gas chromatograph equipped with electron capture detector.

Quality control and quality assurance procedures included replicate sampling, extraction and analysis for all samples. Extraction of the water samples also incorporated studies of spiked samples to determine the recovery rate of the method used. This was accomplished by spiking 1L distilled water with respective standards of pesticides under investigation to obtain 0.1 µg/l final concentration, and following the same extraction and analytical procedures as for the samples. Pure distilled water samples were also incorporated as blanks, and these together with external standards were used to determine the detection limit of each pesticide investigated.

Figure 1 Map of Lake Victoria catchment showing sapling points.



Results

High recovery rates were obtained using solvent-solvent extraction method. The average recovery rates for the analysed pesticides were α-HCH 95.62 %, β-HCH 93.22 %, γ-HCH 96.52 %, p,p'-DDT 97.53 %, o,p'-DDE 97.11 %, p,p-DDD 98.25 %, aldrin 88.59 %, and dieldrin 96.24 %. These were good recoveries in relation to the recommended rate that ranges between 70 to 120 %. The detection limits for analysed pesticides ranged from 0.001 to 0.004 µg/l.

Organochlorine pesticide residues detected in water collected during short rain season were higher in river samples compared to the levels detected in the lake water samples. The residue levels ranged between 0.01-0.34 µg/l in water from river Sio, 0.01-0.44 µg/l in river Nzoia water samples, 0.01-0.26 µg/l in water from Lake Victoria at Sio Port and between 0.01-0.31 µg/l in water samples from Lake Victoria at Marenga Beach. p,p'-DDT, o,p'-DDE, p,p'-DDD and dieldrin constituted the highest residues detected during the short rain season, whereas -HCH was the least (Figure 2). The residues levels of DDT and HCH were both below the WHO recommended guidelines except for aldrin and dieldrin (Figure 2).

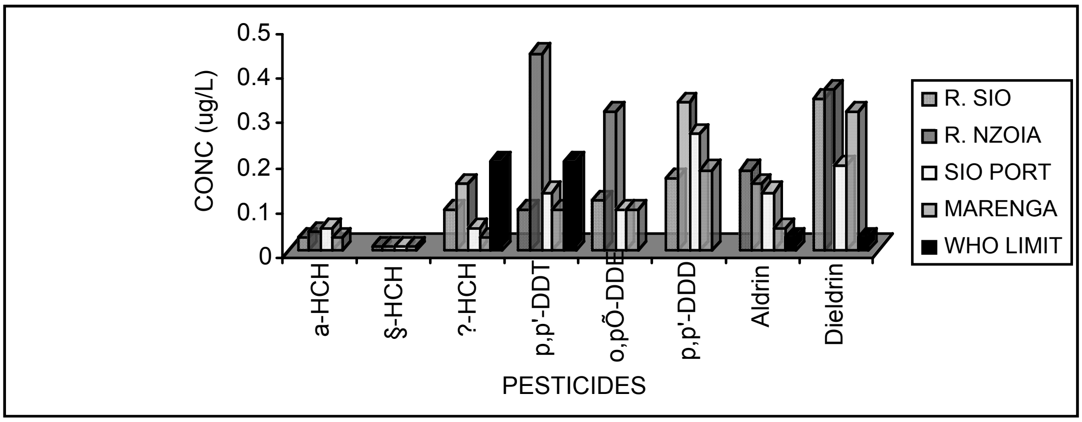


Figure 2. Levels of organochlorine pesticide residues in water during short rain season (µg/l) (WHO values for DDT & γ-HCH x10).

Residues detected in water samples collected during the dry season ranged from 0.01-0.20 µg/l in water samples from river Sio, 0.01-0.21 µg/l for river Nzoia, 0.01-0.09 µg/l in water from Lake Victoria at Sio Port and 0.01-0.31 µg/l in samples from Lake Victoria at Marenga Beach. DDT and HCH were both below the WHO recommended guidelines whereas aldrin and dieldrin were both above the recommended values (Figure 3). The residue levels detected in the water samples collected from the river were higher than those detected in the lake water samples except for p,p'-DDT, o,p-DDE, and p,p-DDD. This could not be explained by either effect of the input from the rivers since the river samples were at lower concentration. No recent input from anthropogenic sources could be attributed to since p,p'-DDD, the metabolite of p,p'-DDT was much higher than the original compound p,p'-DDT. As a consequence the trend was attributed to the contribution from other smaller streams or desorption from sediments bound residues.

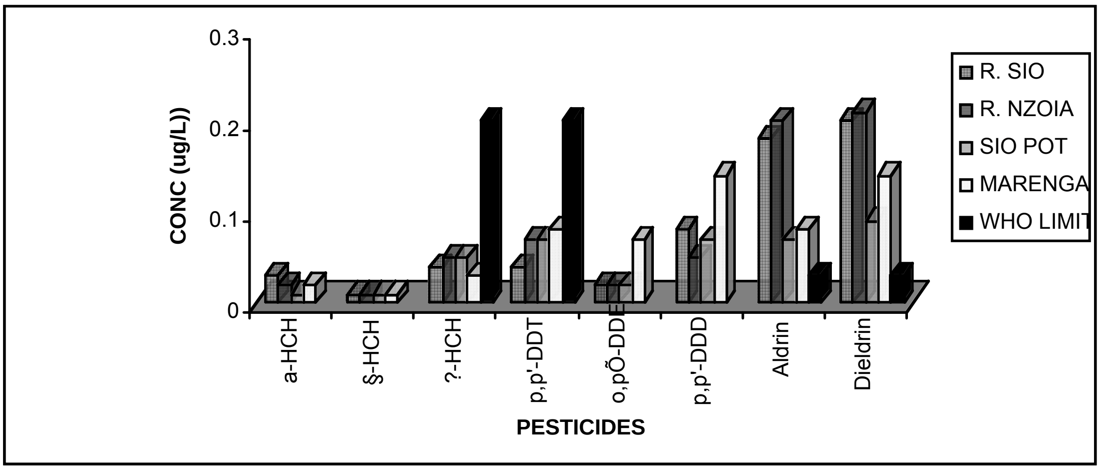


Figure 3. Organochlorine pesticide residues in water during dry season (µg/l) (WHO values for DDT & -HCH x10).

The residues levels detected in water collected in wet season ranged from BDL-0.17 µg/l for river Sio, and from BDL-0.18 µg/l for river Nzoia. Samples collected from the lake showed lower residues levels ranging from BDL-0.12 µg/l in samples from Sio Port, and BDL- 0.31 µg/l in samples collected from Marenga Beach. Aldrin and dieldrin residues were the highest of all detected organochlorines. A similar trend of residue levels as for the short rain seasons was observed with most of residue levels in the river samples higher than those detected in the lake water samples (Figure 4). The residues of DDT and HCH detected during the wet seasons were below the World Health Organization (WHO) recommended guidelines in all the sampling points. However the levels of dieldrin and aldrin were above the recommended WHO limits (Figure 4). Based on the detected residues of dieldrin, which were higher than those for aldrin, the levels detected were attributed to the previous use of the aldrin in the region. The detected levels of -HCH found to be lower that those for -HCH indicating that some farmers might be illegally using lindane. Lindane was initially used for seed dressing to protect crops against termites. However its agricultural use has been banned in the country due to persistence and toxicity to the untargeted organisms.

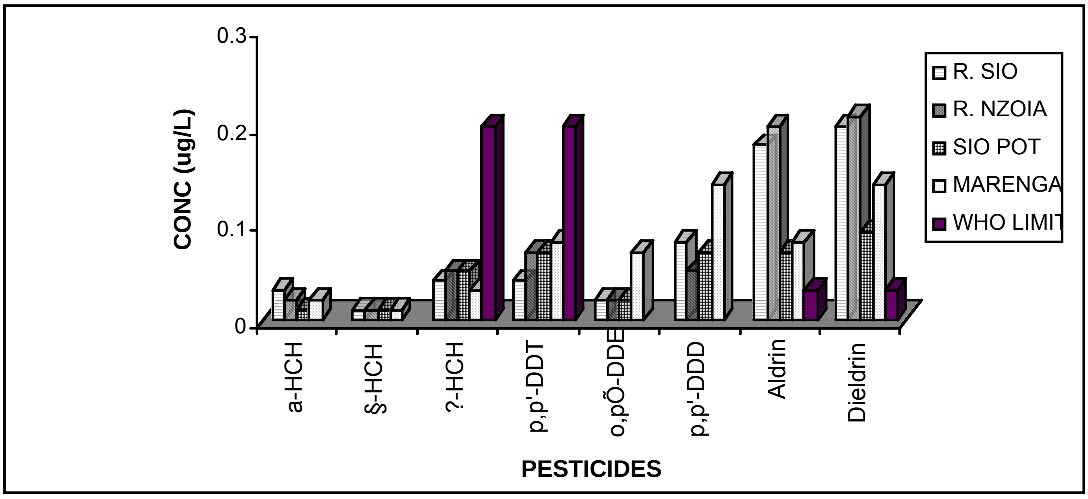


Figure 4. Organochlorine pesticide residues in water during wet season (µg/l) (WHO values for DDT & -HCH x10).

Analysis of seasonal variations of the residue levels across the three seasons indicated that samples collected from river Nzoia during the short rain season contained the highest amount of pesticide residues except for aldrin while the dry and wet seasons had higher aldrin concentrations. For all the three seasons, residue levels of HCH and DDT were below the WHO recommended guidelines for drinking water (Figure 5). The high levels of residues detected during the short rain season compared to the dry season were attributed to the runoff from the fields where those compounds were previously applied. On the other hand the low residue levels detected during the wet season compared to the short rain season could be attributed to dilution effects based on large volumes of rainwater.

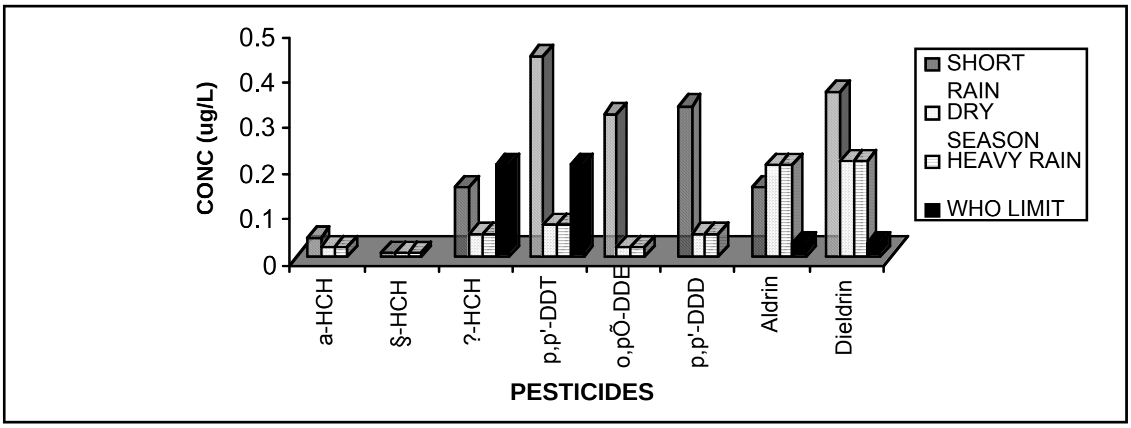


Figure 1.5 Seasonal variations of pesticide residues in river Nzoia (µg/l) (WHO values for DDT & γ-HCH x10).

Comparison of the pesticide residues detected in river Sio against seasonal variations indicated that samples collected during the short rain season contained higher residue levels compared to dry and heavy rain seasons (Figure 6). The trend observed in residue levels in samples from river Sio was similar to that observed for river Nzoia with the highest residue levels detected during the short rain season. Based on individual pesticide residues from river Sio, dieldrin and aldrin constituted the highest residues detected followed by DDT and to its metabolites and lastly the HCHs. This was attributed the fact that aldrin and DDT were the main pesticides previously applied in the region on large scale.

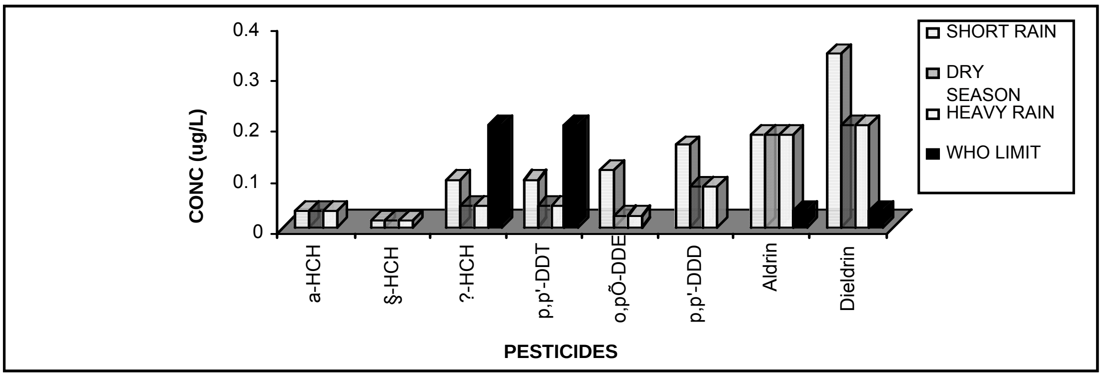


Figure 6 Seasonal variations of pesticide residues in river Sio (µg/l) (WHO values for DDT & -HCH x10).

Analysis of seasonal variations and pesticide residues detected in the water samples collected from Lake Victoria at Sio Port showed that samples collected during the short rain season contained higher pesticide residues than those collected during the other two seasons (Figure 7). DDT and HCH residues were both below the WHO recommended guidelines whereas aldrin and dieldrin were slightly above the recommended levels. Comparison of dieldrin and aldrin ratio (dieldrin/ aldrin) gave value greater than 1 indicating that the detected residues were not likely to be from the recent applications of aldrin in the region. Similar trend was observed in samples collected from Lake Victoria from Marenga Beach where samples collected during short rain season contained the highest residues levels compared to dry and wet seasons.

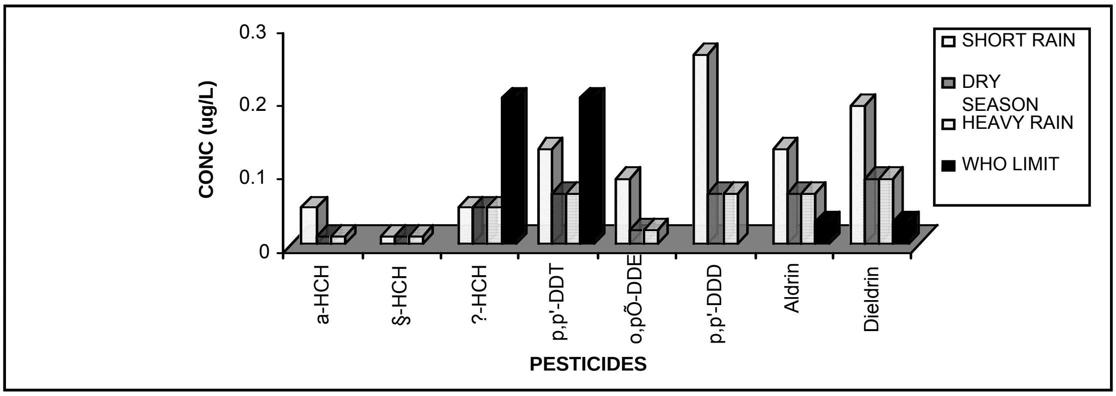


Figure 7 Seasonal vari ations of pesticide residues in Lake Victoria at Sio Port (µg/l) (WHO values for DDT & -HCH x10).

Discussion

In general, the total residues of DDT, HCH, were below the WHO guidelines limit for drinking water, whereas aldrin and dieldrin were slightly above the recommended values. The residue levels reported in this study were lower than those reported in marine environment at the Kenyan coast by Wandiga *et al.,* (2002), and Getenga *et al.,* (2004) in Lake victoria basin. The differences could be attributed to variations in geographical locations, time differences in terms of the period of study, and the extent of previous use of these pesticides. Other studies aimed at providing baseline information on the current levels of organochlorine pesticides in the aquatic system of Lake Victoria showed ratios of DDT to DDE suggesting previous use of the pesticides, and significant use of lindane and endosulfan within the Lake Victoria region (Kasozi, 2001; Mbabazi, 1998). This may explain the trend observed in this study, especially the residue levels of HCH, which could be still illegally used by some farmers.

In comparison to studies carried out in other countries; Mwevura *et al.,* (2002) reported lower frequencies of organochlorine pesticide residues of p,p'-DDT (25%); p,p'-DDE (37%) during dry season, and higher frequencies during wet season giving frequencies of p,p'-DDT (81%); p,p'-DDE (100%); dieldrin (100%) and -HCH (6%) in samples from the coastal area of Dar es Salaam, Tanzania. The concentrations of dieldrin, and p,p'-DDD were notably higher than aldrin and p,p'-DDT, respectively, in most of the samples. Since the later are their degradation products, this indicated possible transformation process taking place on p,p'-DDT and aldrin previously used in the region. Earlier studies showed that the ratios of DDE/DDT, -BHC/lindane, dieldrin/aldrin, and heptachlor epoxide/heptachlor in soil can be used as indicators to recent use of DDT, lindane (-BHC), aldrin, and heptachlor in the environment, whereby low ratios, particularly<1, indicate recent applications (Gonzalez *et al.,* (2003). The calculated values in this study gave dieldrin/aldrin =1.72; and -HCH/-HCH = 0.43. Based on this hypothesis, there had been no recent applications of aldrin except for -HCH. The values for DDE/DDT were not exhaustive since only o,p'-DDE was analysed in this study. However, the value obtained for p,p'-DDD/DDT was 1.5.

The major sources of organochlorine pesticide residues in the Lake Victoria region are previous agricultural activities and aerial sprays in public health vector control. DDT was extensively applied in aerial sprays against mosquitoes to control malaria (Mitema & Gitau, 1990), whereas aldrin and dieldrin are used in termite control in building industry (Getenga *et al.,* 2004). The public use of these compounds was banned or restricted in Kenya.

Acknowledgement

The authors of this paper would like to acknowledge Kenya Plant Health Inspectorate Services (KEPHIS) for the assistance of pesticide standards that were used for screening the pesticide residues in the field samples. DAAD (German Academic Exchange Service) is acknowledged for the scholarship, which contributed to the success of this work.

References

Getenga, Z. M., Kengara F. O. & Wandiga S. O., 2004. Determination of organochlorine pesticides in soil and water from river Nyando Drainage System Within Lake Victoria Basin, Kenya. *Bull. Environ. Contam. Toxicol* 72 (2): 335-342.

Gonzale, M., Karina S. B. M., Julia E. A., Victor J. M., 2003. Occurrence and of organochlorine pesticides in tomato (*Lycopersicon esculentum*) crops from organic production. *J. Agric Food Chem*. 51: 1353-1359.

Kasozi, G. N., 2001. *Analysis of organochlorine pesticide residues in the aquatic ecosystem of Lake Victoria*. M. Sc. Thesis Makerere University. Kampala, Uganda.

Mbabazi, S. B., 1998. *Levels of lindane in soil, plants, and selected soil fauna in a sprayed in maize agroecosystem*. M. Sc. Thesis, Makerere University. Kampala, Uganda.

Mitema, E. S. & Gitau F. K., 1990. Organochlorine residues in fish from Lake Victoria, Kenya. *Afri J. Ecol*. 28(3): 234-239.

Mugachia, J. C., Kanja L., & Maitho T. E. 1992b, Organochlorine pesticides in Estuarine fish from the Athi River, Kenya. *Bull Environ. Contam Toxicol. 49(2)*: 199-206.

Mwevura, H., Othman O. C. & Mhehe G. L., 2002. Organochlorine pesticide residues in waters from the Coastal area of Dar ES salaam and their effect on aquatic biota. *Tanz. J. Sci*. 28(2): 117-130.

Wandiga, S.O., Yugi P. O., Barasa M. W., Jumba I. O.& Lalah J. O., 2002. The distribution of organochlorine pesticides in marine samples along the Indian Ocean Coast of Kenya. *Environmental Toxicology.* 23: 1235-1246.

The study of waste load dispersion in Songkhla Lake

Sae-Chew W.

Civil Engineering Department, Faculty of Engineering,Prince of Songkla University,

Hatyai, Songkhla, Thailand. 90112

Abstact

The changes of water quality in Songkhla Lake due to the water pollution released from the canals and the surrounded areas of the lake, obviously impact to the fishery ,the aquatic life , aqua farming and the habitation around the lake. A large amount of waste water in Songkhla Lake mainly comes from the municipalities and the industries from Sadao , Hatyai and Songkhla district . The study of waste load dispersion by a simulation model method is an approach of the solution . The purpose is to determine and to locate the boundary and the degree of the pollution impacts as the results of the canals. The study focused on Khlong U-Tapao and Khlong Sam Rong canal rather than the others. As they are the main canal which flow through the most active areas of the basin. The other canals namely ; Khlong Laem Po , Khlong Ku-Kut and Khlong Pavong. are not so significant. Since they flow in the remote areas and there are not much activities of development and thus, they may be ignored. Khlong U-Tapao and Khlong Sam Rong canal recieve the municipal and the industrial waste water from Hatyai ,Sadao and Songkhla Cities with an increasing amount every year , for example, the suspended sediment from Hatyai and Sadao region are more than 20 million tons per year . The pollutants such as BOD5 (untreated) , Coliform Bacteria, DO and Suspended Load are interesting and are collected as input data for testing a 2-D mathematical simulation model. Initial boundary conditions and basic parameters such as tidal current , wind speed , wave magnitude, mean sea depth, sea bottom slope and channel characteristics of the canals, etc. are necessary sources of data input. The Collection of field data was done in dry season so as to obtain concentrated pollutant data and to avoid the affects of the high discharge from the rain. The tidal fluctuation and the tidal current from the open sea have influence on the movement of dispersion and the circulation. Both events cause vortexity in the lake in relation to water flow from the canals . The dispersion of the pollution from Khlong U-Tapao will cover the area from the river mouth at the distance within 2 kilometers during high tide. The residence time of the product in the lake is longer before ir is discharged to the Gulf of Thailand during low tide. Other important factors to the delay of the pollution discharge are the narrow outlet of Songkhla Lake to the Gulf of Thailand , a high density sea transportation , a developing of new ports and accumulative sand-sedimentation at the outlet of the lake.

Introduction

Songkhla Lake Basin has a reputation as a significant natural resource that feeds the population in three different districts Songkhla, Pattalunf and Nakornsrithammarat. Increasing population and the need for new land for agriculture and development in the area, has to the decline of natural resources due to increasing pollution from the habitation and the industries. The natural impact occuring in the lake inclede shallowing of the lake due to annually higher sediment, salinity intrusion of sea water into inner lake, shortage of fresh water for water supply in dry season in some areas and a reduction of some species in the natural aquatic life. Finding solution is an indicator of that Songkhla Lake basin is facing a critical recession of natural resource quality. If environmental impact awareness is not created in time, the negative impact of these activities will be irreversible.

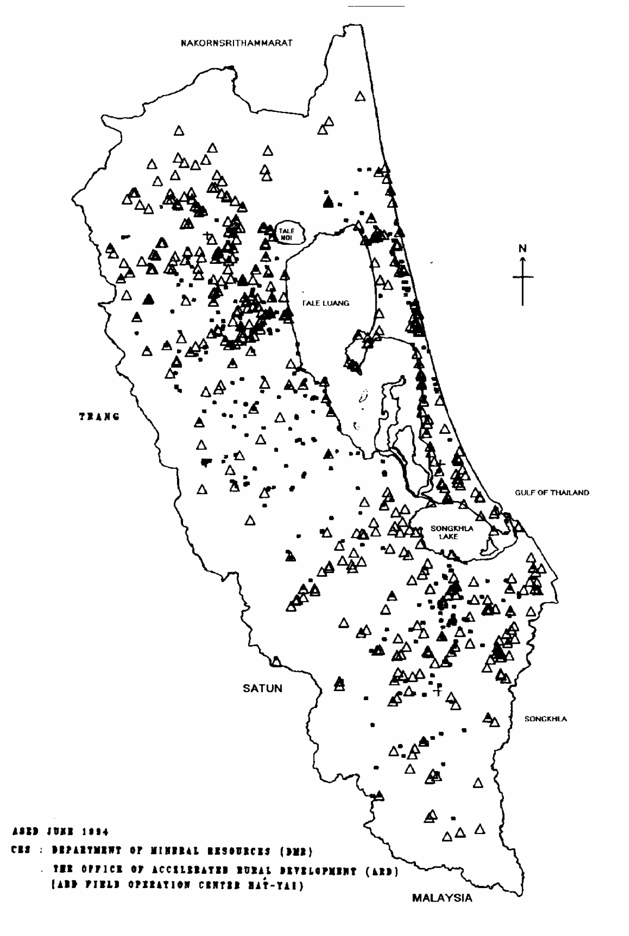


Figure 1. Showed songkhla lake basin.

Objective

This study is an approach to create awareness of the environmental impact to the lake. The Study looked at how the waste water is circulating ,where it reaches and how long it will be detained in the lake. To asses this, a dispersion simulation model was utilized in order to exhibit the occurrence of the impact. Many essential data collections were processed , particularly on Khlong U-Tapao canal which is a major and influent canal and physical characteristics of the lake water such as tidal movement , wave movement ,wind speed and inflow discharge from the boundaries etc. were also studied along with some waste load parameters such as BOD , DO , Salinity , Coliform Bacteria and Suspended Sediment , which are of much interest for input data in simulation program. After testing and running The program, The simulation gave pictures of the dispersion of the waste load in the lake and enabled us to understand the manouver of the impact .

Physical characteristics

Characteristics of Songkhla Lake

Songkhla Lake basin is sited on the east coast of southern Thailand .It has total area of 9,807 sq.km which are divided to water area of 1,046.04 sq. km. and land area of 8,761 sq. km. The basin is connected with 3 districts namely Songkhla , Pattalung and Nokornsrithammarat. The prominent characteristic of Songkhla Lake basin is the biggest natural water resource of fresh and saline water. The topography of the landscape comprise with hill slope from the west ,rolling plain beside the hill extended from the north to the south of the landscape and coastal plain which is accumulated from sea sedimentation. Therefore ; the plain and the coastal area are good for agriculture , fishery and habitation.

Characteristics of Khlong U-Tapao River Basin

Khlong U-tapao river basin has a catchment area of about 2,305 sq. kms. as shown in Figure 1. Physical soil type generally is sandy and covered with rubber plantation. Topography of the landscape varies from mild mountainous plateau on the west side to flat fluvial deposit coastal landscape to the Gulf of Thailand on the east side. With the equatorial monsoon climate , the region has a climate of humid hot rainy season with a mean temperature of about 26.9º Celsius and an average evaporation of 139 mm/d. The average precipitation is about 1800 mm./year. Khlong U-Tapao river basin give a water yield of the catchment about 791.94 mm^3. (1967-1986) and the Khlong U-tapao canal has a discharge of 7.80 cumec/sec in dry season and 88.60 cumec/sec in wet season. The width of water surface was varies between 40-80 meters and the mean depth is approximately between 3.0-8.0 meters. The significant flow distance from the upper region to the mouth of the canal is about 70 kms. With different channel slopes as shown in Table1.

Table 1. Station-distance-bedslope of Khlong U-Tapao Canal.

STATION	DISTANCE (KM.)	BEDSLOPE
1. SADAO-BAN.BANGSALA	70	0.00057
2. BAN.BANGSALA-HATYAI	15	0.00017
3.HATYAI-SONGKHLA LAKE	20	0.00005

Table 2. Sediment station and yearly mean sediment.

RIVER & STATION	LOCATION	DRAINAGE AREA (SQ.KM.)	MEAN ANNUAL SS(TONS)
KHLONG U-TAPAO BAN.BANGSALA	HATYAI, X90	1562 (1978-1985)	5458.0
KHLONG SADAO BAN PHRAI	SADAO,X11	256 (1979-1985)	7218.
KHLONG LAM BAN.THUNGPRAP	SADAO, X113	129 (1979-1985)	5689.0
KHLONG WAT TONNGACHANG	HATYAI ,NEA	14 (1980-1984)	2158.4

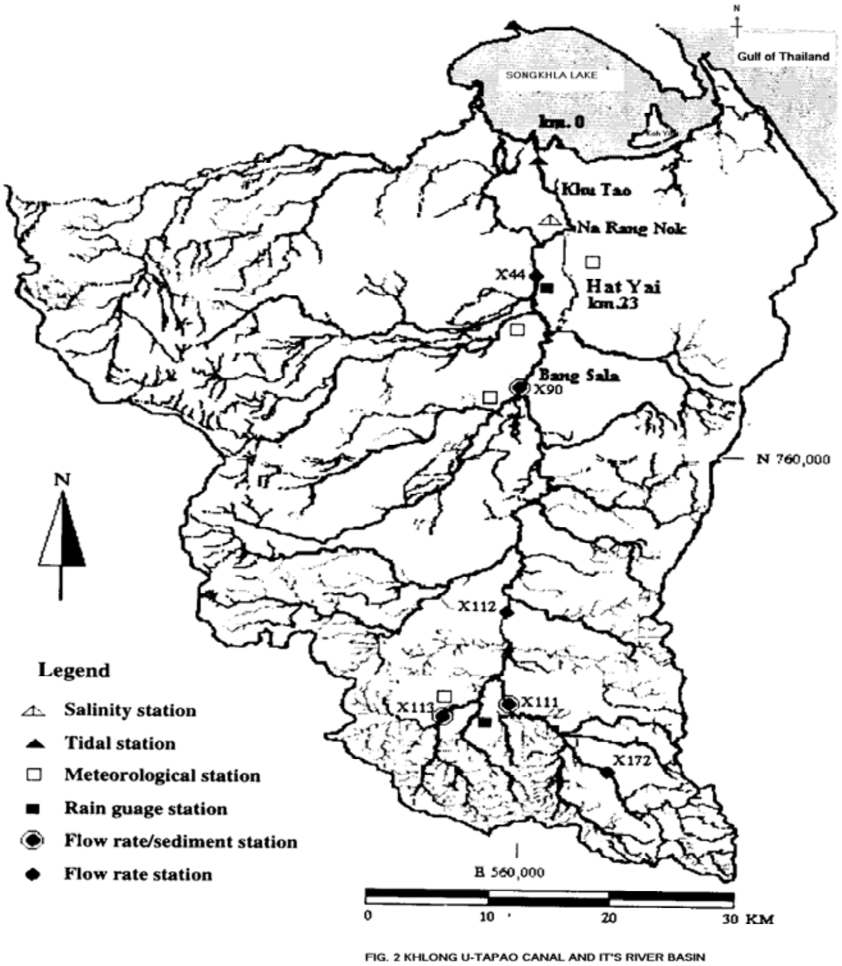


Figure 2. Khlong U-Tapao Canal and the River Basin.

General overview of the canals

Khlong U-TapaoKhlong U-Tapao canal

It is a major streamflow in Songkhla lake river basin . The flow is originated from Sadao catchment area in the south and flows via Hatyai catchment before ending in the Songkhla lake in the north

Khlong Kuut

There is little significant discharge from canal into the lake.

Khlong Pavong

As a result of the highly poor habitation and the fishery industries around the canal there is no sufficient water treatment. So that There is high municipal and industrial waste water released into the lake via Khlong Pavong canal which is flowing at the south of Koh Yo island.

Khlong Samroung

There are 6000 people living nearby the canal (Brans, *et al.,* 1995) who produce polluted water that seriously affect the canal. Presuming such waste water enter the fish farming site in the neighbouring area in the lake in 1989 (R&D 1989).

Process of the waste load dispersion in Songkhla lake

The waste load dispersion process depends on the influences of tidal fluctuation, wind speed and direction , density of the suspended load and the temperature difference of the water and turbulence of the flow etc. Some additional parameters such as streamflow and flocculation are also considered in this study.

Equations used in the Mathematical model

A 2- dimensional hydrodynamics simulation model is utilised by considering the equations of shallow water wave and the approximation method, curvilinear coordinate applications are adapted into the continuity equation and the momentum equation.

The conceptual calculation by the numerical solution method is applied by using staggered grid,semi difference approximation and convective-diffusion of mass.

Continuity equation

$$\frac{\partial \zeta}{\partial \tau}+\frac{1}{r\cos\theta}\frac{\partial UH}{\partial \phi}+\frac{1}{r}\frac{\partial VH}{\partial \theta}-\frac{VH\tan\theta}{r}=0$$

Momentum equation of ϕ

$$\frac{\partial UH}{\partial \tau}+\frac{1}{r\cos}\frac{\partial UUH}{\theta\partial\phi}+\frac{1}{r}\frac{\partial UVH}{\partial \theta}-\frac{UVH\tan\theta}{r}=-\frac{1}{\rho_o r\cos\theta}\varphi\phi+\frac{\tau_b\phi}{\rho_o}$$

$$\varphi_\phi=\int_{-h}^{\zeta}\frac{\partial pdr}{\partial\phi}=\rho_o gH\frac{\partial(h+\zeta)}{\partial\phi}+\frac{gH^2}{2}\frac{\partial\rho'}{\partial\phi}$$

$$\tau_{b\phi}=\rho c_f U\quad(U^2+V^2)$$

Momentum equation of θ

$$\frac{\partial VH}{\partial \tau}+\frac{1}{r\cos\theta}\frac{\partial UVH}{\partial\phi}+\frac{1}{r}\frac{\partial VVH}{\partial \theta}+\frac{U^2-V^2}{r}H\tan\theta=-\frac{1}{\rho_o r}\varphi\theta+\frac{\tau_b\theta}{\rho_o}$$

$$\varphi_\theta=\int_{-h}^{\zeta}\frac{\partial pdr}{\partial\theta}=\rho_o gH\frac{\partial(-h+\zeta)}{\partial\theta}+\frac{gH^2}{2}\frac{\partial\rho'}{\partial\theta}$$

$$\tau_b\theta=\rho c_f\, v\quad(U^2+V^2)$$

WHEN

U,V = depth-averaged velocities in ϕ- and θ -direction, respectively

ζ = water surface elevation

$\tau_{b\phi},\tau_{b\theta}$ =bottom shear stress in ϕ- and θ-direction, respectively

H = depth (MSL)

C_f = bottom friction coefficient

2-D Vertically averaged model of waste load

$$\frac{\partial c}{\partial t}+\frac{1}{r}\frac{U\partial c}{\partial\theta}+\frac{1}{r}\frac{V\partial c}{\partial\phi}=\frac{D_H}{r^2\cos^2\theta}(c_{\theta\theta}-c_{\theta\phi}+c_{\phi\phi})-D_H\tan\theta\frac{\partial c}{\partial\theta}\quad+s$$

$$S=-K_1c$$

WHEN

c = concentration of mass

D_H = diffusion coefficient in ϕ- and θ-direction

S = Source term fate

K_1 = Decay rate = 0.3-04 day^{-1}

Running of hydrological model

1. Designated Songkhla lake grid area for the model showed in Figure 5
2. Selecting a suitable grid width (1 grid width is about 500-1000 m.) and grid number which there are about 18 x 20 numbers and time step 6 hours. for explicit finite difference scheme
3. Input actual mean depth of the lake (maximum mean depth is about 9 m. at the entrance of the lake and the Gulf of Thailand and at the middle of the lake is about 1.5 m.) at the middle of the grid block showed in Figure 3
4. Input tidal fluctuation
5. Input initial condition of the current velocity in the lake which is assumed to be zero
6. Input discharge of the streamflow in a dry season from the canals around the lake
7. Running the model of streamflow discharge interacted with the fluctuation by hour for consecutive fluctuation of 16 days in order to give a liable simulation to compare with actual collected field data and consequently to adjust the coefficient of roughness parameter in the programme
8. Repeating the process 6.4-6.7 until it gives the best fitness of the comparison of model.
9. Input initial condition of the current velocity in the lake which is assumed to be zero
10. Input discharge of the streamflow in a dry season from the canals around the lake
11. Running the model of streamflow discharge interacted with the fluctuation by hour for consecutive fluctuation of 16 days in order to give a liable simulation to compare with actual collected field data and consequently to adjust the coefficient of roughness parameter in the programme
12. Repeating the process 6.4-6.7 until it gives the best fitness of the comparison of model.

Figure 3. The contour of mean sea depth in Songkhla Lake (m.).

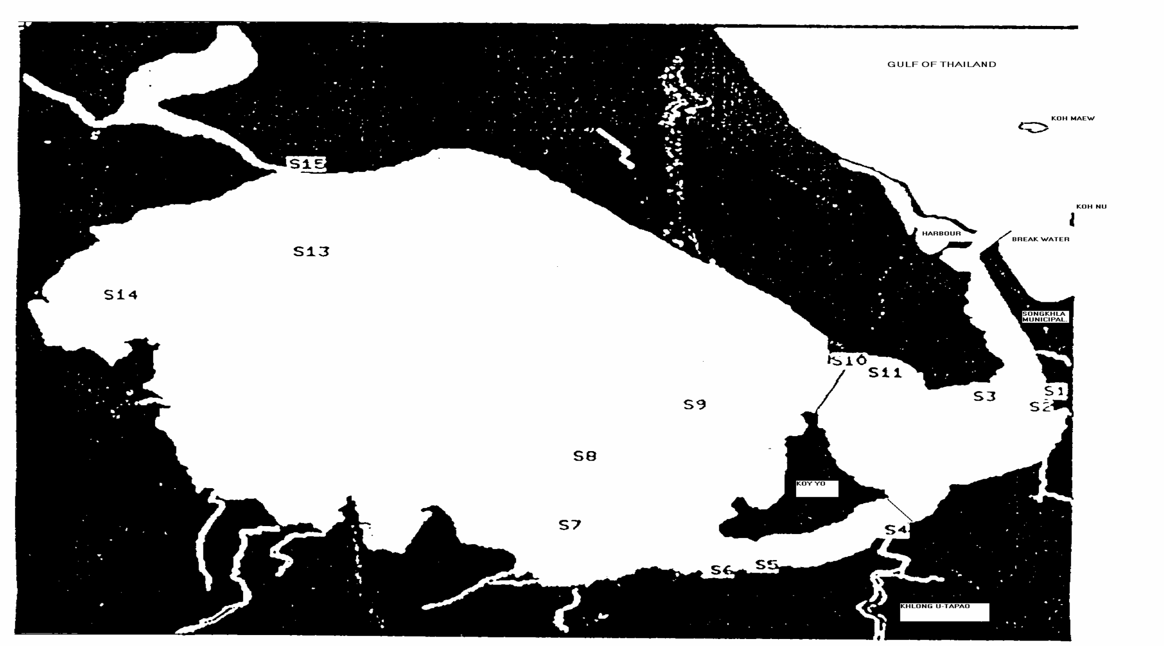


Figure 4. The location of the data collection (23 Feb. 1997).

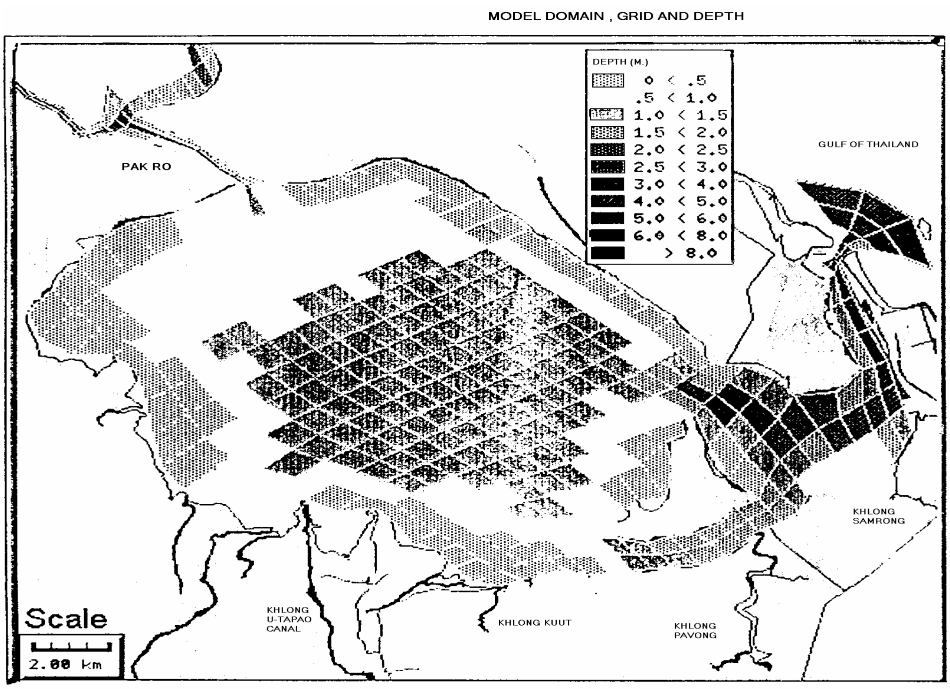


Figure 5. The domain, grid and depth (m.) in the simulation model.

Running of waste load transport model

Waste load transport model will be testing in associate with the hydrological model after obtaining confidence of the model so as to making the forecasting of the waste load dispersion's pattern. The procedures are as follows

1. Input fluctuation data
2. Set initial condition of the current in the lake at stationary condition , the velocity is zero
3. Initial condition for lake water quality is set to be zero
4. Input one by one waste water discharge from the canal which flows into the lake in order to visual the individual impact from each flow
5. Selecting time step of one hour and testing the model to cover the period of 16 days so as to consider whether there is critical event appear.

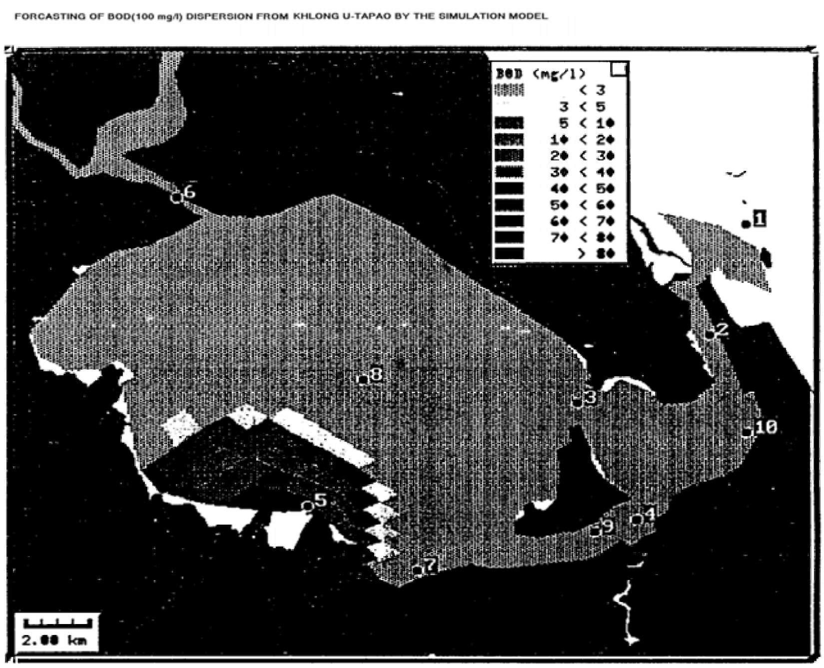


Figure 6. A forcasting of the bod(100 mg/l) dispersion from khlong u-tapao canal by the simulation model.

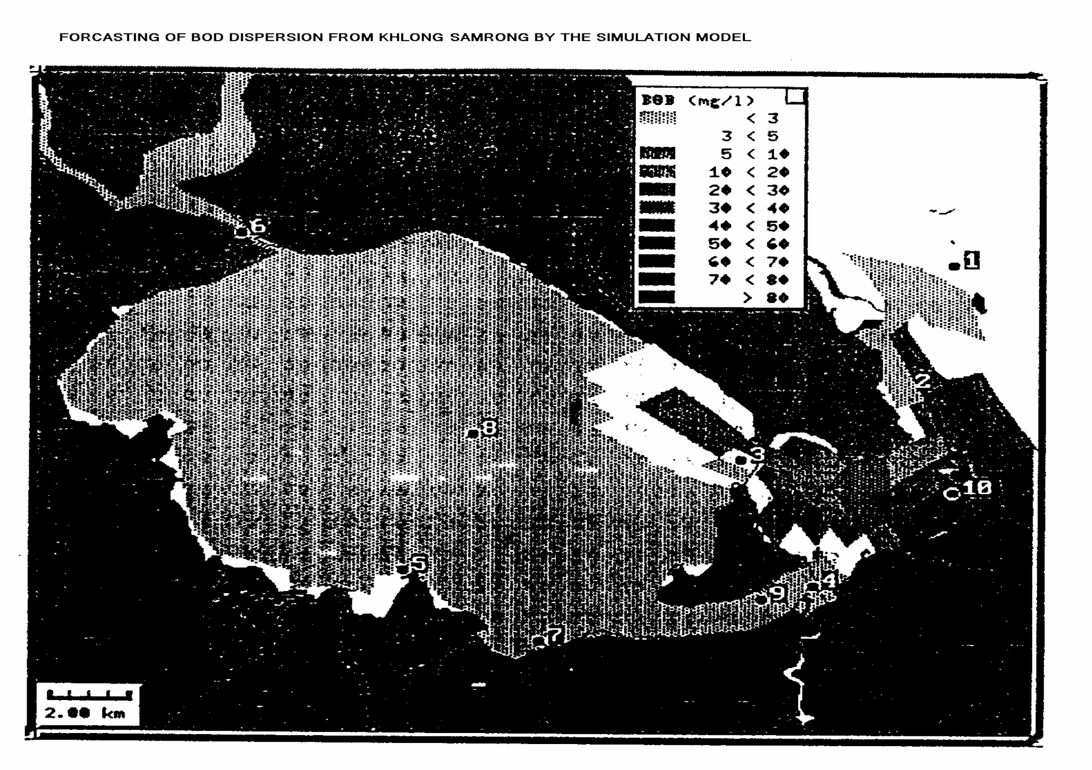
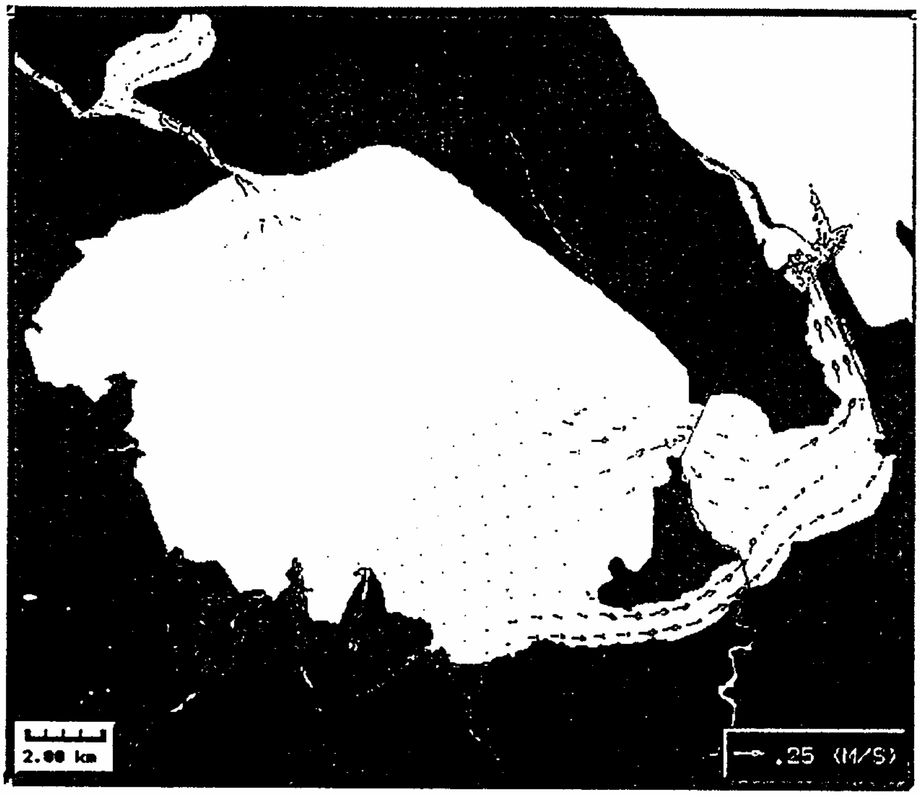


Figure 7. Table 1. Station-distance-bedslope of Khlong U-Tapao Canal.

FORCASTING OF 3 HOUR EBB CURRENT BY THE SIMULATION MODEL



FORCASTING OF 9 HOUR EBB CURRENT BY THE SIMULATION MODEL

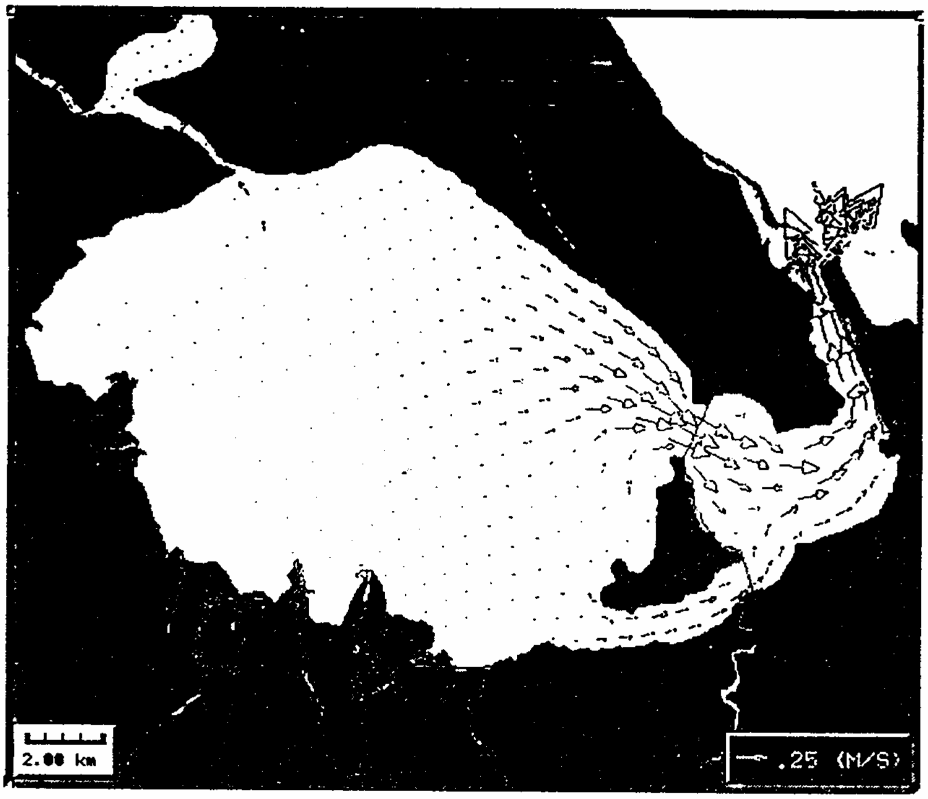
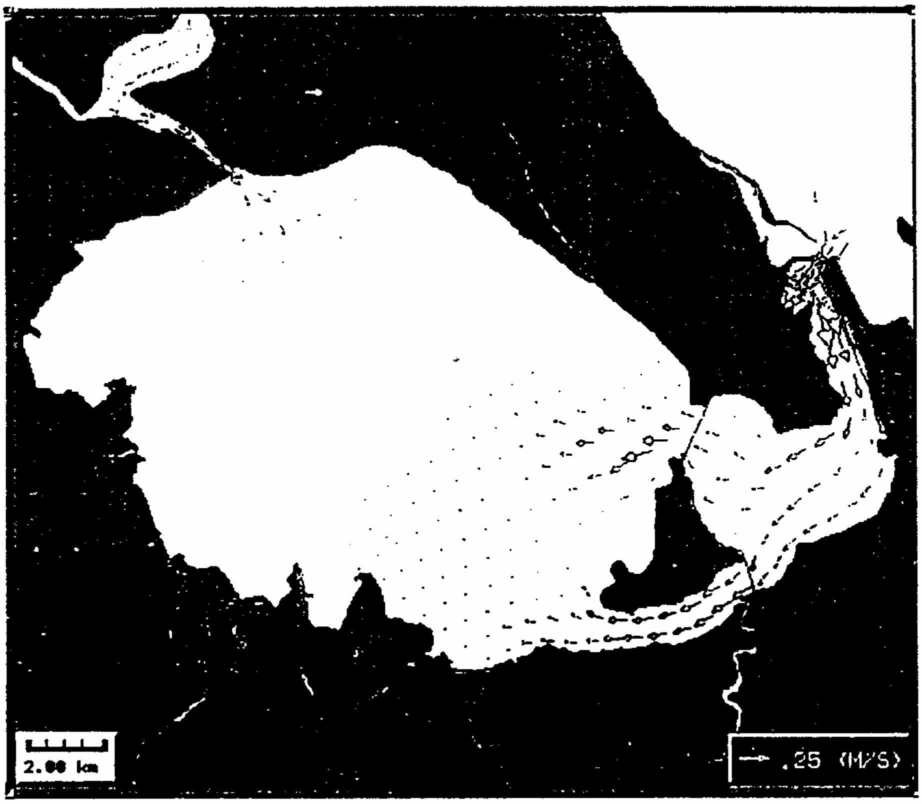


Figure 8. A forcasting of the ebb tidal current by the simulation model.

THE SIMULATION OF THE HIGH TIDAL CURRENT



3 HOUR DURATION OF THE RISING OF HIGH TIDAl CURRENT

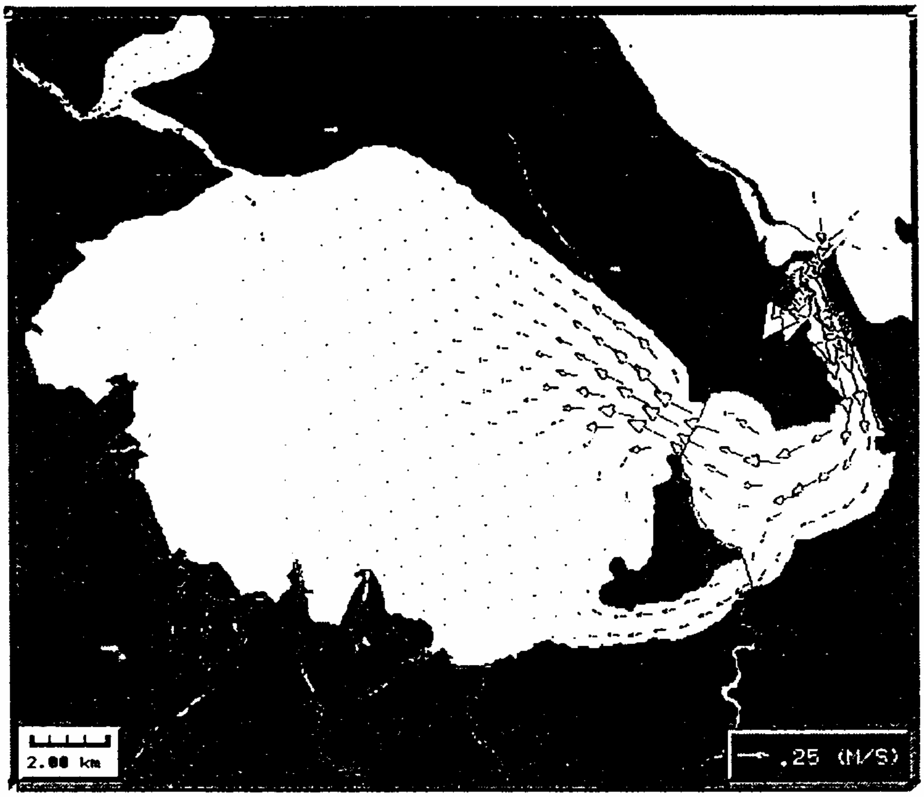


Figure 9. A forcasting of the high tidal current by the simulation model.

Results of the model testing

The waste load dispersion in the model is shown by the raster which is distinguished by the concentration of the colour. The picture showed that the concentration of the waste dispersion is varied in accordance with the current of the flow in the lake .

Conclusion

From the field data collection in dry season , it is found that The pH is varied between 25-31 ppt at Koh Yo and reducing to 12-29 ppt at the neck of the junction of the lake and the Gulf of Thailand. It is found that the pH layer separation appear at a level of more than 3 m. depth and this event causes partial mixture of the water happening. Water temperature is in average of 28 Celsius degrees. The temperature difference of the surface and the bottom of the lake is approximately 1 Celsius degree. The concentration of the suspended load is between 6-25 mg/l. The bed load of the sediment varies from mud in average to become more sand when it is close to the outlets of the lake. BOD5 (untreated) is varied between 2-7 mg/l . Coliform bacteria is between 2-31 MPN/100 ml, while in the canals are much more than 2,400 MPN/100 ml. The tidal difference is about 40 cm. with phase difference of 3.5 hrs. Maximum tidal current is about 0.67 m/sec. In the north and reduce to 0.43 m./sec in the south of the lake close to the junction of Khlong U-Tapao canal . The result of model testing found that roughness coefficient of the lake is about 0.0015 and the current velocity simulation is accurate with the maximum velocity of 0.77 m./sec and minimum velocity of 00.8 m./sec . The collision of water mass in the lake due to the phase and the elevation difference of the water will induce complicated vortexity at the middle and the east side of the lake but meanwhile on the west side of the lake the flow current is rather smooth . The waste load released from Khlong U-Tapao canal tends to disperse radiusly and covers farther distance about 2 kms. from the mouth of the canal with the concentration of 10 mg/l . The model study showed that if there is a consecutive and continuous release more than 2-3 days of waste water, the east side of the lake will be the first place to be impacted seriously by the concentration about 50 mg/l. The impact will not affect much the reservation area and fish farming at Koh Yo . But there is a serious disaster to the fish farming as the result of the influence of the waste dispersion which is released from Khlong Pavong and the impact is covered over 3 kms and 2kms. distance from the mouth of Khlong Pavong. The waste from Khlong SamRong will disperse at the north of Koh Yo rapidly and impounded within the lake as it can not be in time following the low tide. .

References

Winai and Somboon, 1995. Feasibility Study of Sedimentation in Khlong U-Tapao Canal by a Simulation Model.

Brans, R.P., A.C., and Ruyg, B.S.,1995. *Improvement of the Klong Sam Rong." TU Division of Hydraulics*. Faculty of Civil Engineering, TUDelft, Netherland.

Research and Development ,1989. Environmental Impact Assessment of Songkhla Lake Basin " Final Report , Prince of Songkla University.

John Taylor and Sons,1985. Songkhla Lake Basin Planning Study. *Final Repor.* Volume 4: Projects, Water resources.

Swanson, J.C., 1986. *A Three Dimensional Numerical Model System of Coastal Circulation and Water Quality.* Ph.D. Dissertation, Department of Ocean Engineering, Univ. of Rhode Island.

Thomann, R.V and Mueller J.A., 198-, Principles of Surface Water Quality Modeling and Control.

Restoration of polluted lake by ecotechnology - a case study in Taiwan

Lei Yang

Dept. of Marine Environment and Engineering, National Sun Yat-sen University
70 Lien-Hai Road, Kaohsiung 804, Taiwan
Email: leiyang@mail.nsysu.edu.tw

Abstract

Chung-Cheng Lake is located in Mei-Non, a beautiful Hakka agricultural village in south of Taiwan. The lake was initially functioned for irrigation and flood control. However, it has been polluted seriously by sewage, swine wastewater, and non-point source pollution from crop fields. Especially for swine wastewater, there were over several hundreds thousands pigs raised in this village. The point source pollution of the lake comes from two creeks, into which swine wastewater and sewage from the village are discharged. In this study, Chung-Cheng Lake was used as a designing case for lake restoration by ecotechnology. According to Japanese experience on restoration of the Lake Kasumigaura, an artificial inner lagoon and several artificial vegetated floating islands were suggested constructed in the lake area of inflow of creeks to control the contaminants of suspended solids (SS) and nutrients from the creeks. A lakeshore constructed wetland was also suggested to be built in the littoral zones of the lake near the artificial inner lake to treat the organic pollutants in the creeks. It is predicted to cut 10-50% of SS, 40-50% biochemical oxygen demand (BOD), 30-50% of total nitrogen (TN), and 30-90% of total phosphorus (TP). It is concluded that ecotechnology can be an efficient, economical, and ecological (3E) way to restore polluted lakes caused by both eutrophication and organic pollutants.

Key words: ecotechnology, eutrophication, polluted lake

Introduction

Chung-Cheng Lake is located in Mei-Non, Kaohsiung County, a beautiful small Hakka (one of the Chinese tribes immigrated to Taiwan 400 years ago) agricultural village in south of Taiwan. The lake was initially functioned for irrigation and flood control with area about 22 hectares. However, since livestock and aquacultural industries became prosperous in Mai-Non in the early stage of development, and the population of the village people increased fast, different kinds of wastewaters, including sewage, swine wastewater, aquacultural pond wastewater, stormwater and irrigation tailed water from crop fields, have directly or indirectly contaminated the lake to cause it silted up and polluted by such kinds of point and non-point sources pollution. Especially for swine wastewater, it has over several hundred thousands pigs raised in Mei-Non Village. Most of the wastewater was discharged into the Jiang-Zhi-Liao and Tai-Zhi-Ken Creeks, which are jointed as one creak right before discharging into the lake as shown in Figure 1. Although the number of pigs raised has decreased down to several hundreds presently due to the policy of “Depress Livestock Industry in Water Source Areas” in Taiwan, sewage, in stead of swine wastewater, should become the main point source pollution for the lake. Therefore, in order to recover the beautiful scene of this Ha-Ka village, restoration of polluted Chung-Cheng Lake would be an ideal beginner.

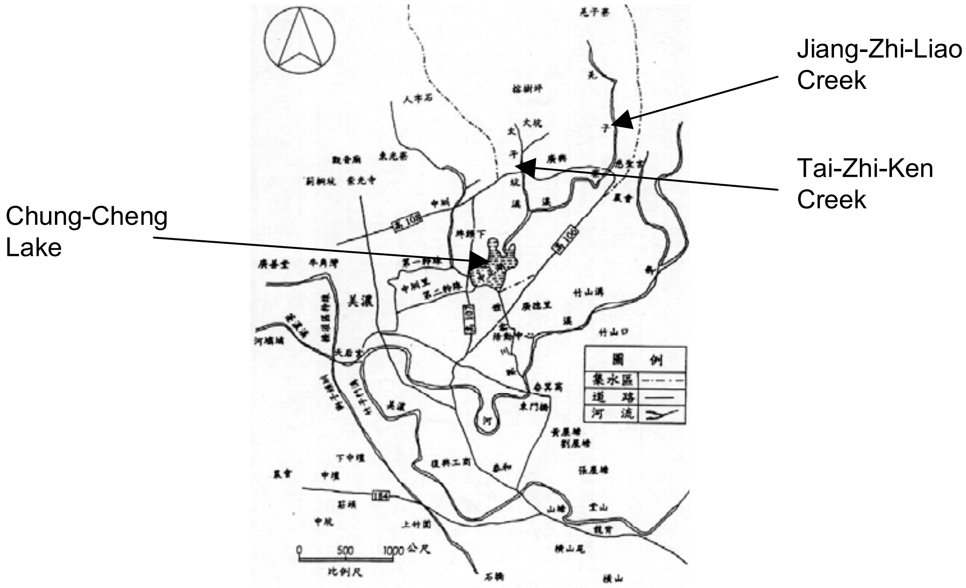


Figure 1 Geographic diagram of Chung-Cheng Lake and main inflow creeks.

Since traditional construction techniques may cause an injury to environmental ecosystem permanently, in this study, we try to use ecotechnology to restore Chung-Cheng Lake as a designing case for restoration of polluted lakes. Ecotechnology, also known as ecological engineering, is a new field of applied ecology developed in 1960's by G. P. Odum and H. T. Odum, which can be viewed as designing or restoring ecosystems according to ecological principles (Mitsch & Jørgensen, 2004). Ecotechnology thus combines basic and applied science to restore, design, and construct aquatic and terrestrial ecosystems (Bergen, 2001). In this study, a case in Japan was reviewed as a reference to help to plan the restoration project of Chung-Cheng Lake.

Design plan of ecotechnological restoration of Chung-Cheng Lake

According to the Japanese experience on restoration of the Lake Kasumigaura, an artificial inner lagoon and several artificial vegetated floating islands were designed and constructed to control the point-source pollution from Kawajiri River as shown in Figure 2. The artificial inner lagoon was built inside the lake along part of the lake shoreline as "a lake inside a lake". The lagoon was confined in a small area inside the lake, which would be manipulated more easily to remove large amounts of SS from inflow river water through the mechanism of sedimentation. Then, the river water flowed into artificial vegetated floating islands area, in which large amounts of organic materials and inorganic nutrients were removed through uptake by aquatic plants and biodegradation by microorganisms attached onto the rhizosphere of the aquatic plants planted on the floating islands. In addition to water purification effect, artificial vegetated floating islands were also functioned for beautiful scene and ecosystem habitat for water fowls and other aquatic organisms. Since the artificial inner lagoon was constructed as a semi-close environment, the inner side of dike built with rock had obviously ability of depressing wave energy, which could protect the floating islands installed along the inner side of the dike inside the artificial lagoon. It was reported that the removal efficiencies of suspended solids (SS) by the inner lake functioned as grit settling tank were measured in the range of 10-50%, while the artificial floating islands exhibited removal efficiencies of 20-50% for total nitrogen (TN), and 30-90% for total phosphorus (TP) in the inflow water of Kawajiri River (Nakamura *et al.*, 1997). Hence, it was demonstrated that such kind of ecotechnology was functioned well to remove point source pollutants of SS, TN and TP into a lake from a polluted river, which prevented the lake from eutrophication and silt up.

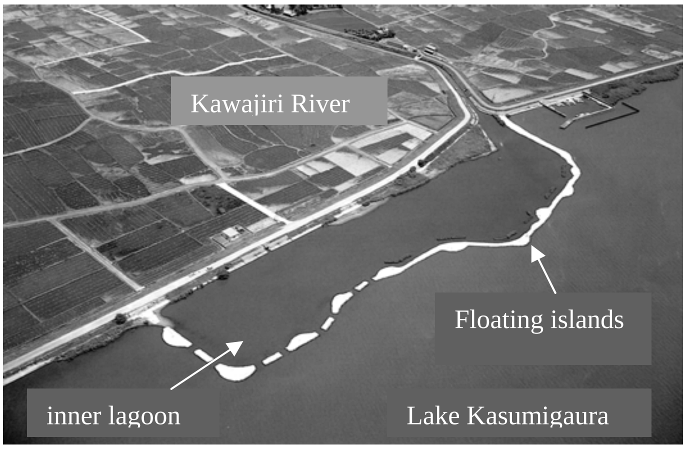


Figure 2 Picture of ecotechnology applied in the Lake Kasumigaura in Japan.

For the case of Chung-Cheng Lake, we found that based on pollution pattern, the lake was similar to the Lake Kasumigaura, with both of the lakes have a point source pollution caused by rivers flowing through agricultural fields and then into the lakes. Therefore, we plan to design a similar artificial inner lagoon inside the Chung-Cheng Lake near the inflow area of Jian-Zhi-Liao creak with area of about 1 hectare and depth of 3 meters, which results in an average hydraulic retention time (HRT) of inflow equal to about 12 hours. The artificial inner lagoon will be designed as semi-open style similar to the one in Lake Kasumigaura shown in Figure 2. Along the inner side of the surrounding dike with several outlets connected with the lake, several artificial vegetated floating islands planted with reeds, cattails, or the aquatic plants with flowers, such as canna for scenic purpose. Wu & Yang (2000) reported that they had successfully used artificial floating islands planted with canna to remove nutrients from Donhu (East Lake) in Wuhan, China. The management and maintenance of floating islands can be divided into two parts: vegetation and island structure. For the management of vegetation, it includes fertilizing, harvesting, and removing weeds. The management of island structure includes checking, repairing and maintaining the structure of island. However, as mentioned previously since the Jiang-Zhi-Liao and Tai-Zhi-Ken Creeks in Taiwan were contaminated by swine wastewater and sewage, which caused more contaminated water quality in these two creeks than in the Kawajiri River, a lakeshore constructed wetland was suggested to be built in the littoral zones of the lake near the river mouth to treat the water from the artificial inner lagoon further, by which organic pollutants and more nutrients and SS can be removed from the river water. The constructed wetland systems can be also used to control partly non-point source pollution of stormwater and irrigation tail water from the nearby agricultural fields surrounding the lake. In addition, this wetland ecosystem can be regarded as a recovered habitat for wild life, which is helpful to keep biodiversity and sustainable development of Chung-Cheng Lake. The schematic diagram of Chung-Cheng Lake manipulated by the ecotechnology mentioned previously is shown in Figure 3.

Construction plan of ecotechnological restoration of Chung-Cheng Lake

Presently, about two thirds of area was silted up in Chung-Cheng Lake. The Bureau of Water Conservancy of Kaohsiung County Government planned to dredge the sediments out of the lake. It was suggested by some scholars and experts on ecological engineering to preserve part of the silted up lake area as the site of building lakeshore constructed wetlands in the future. The county government finally agreed with this suggestion, and preserved about 3 hectares not to dredge. Presently, this part of lake area has been functioned as a natural wetland system, which attracted a large number of water fowls flying here for feeding, resting and nesting. However, a design work is still required to build a constructed wetland system, in which not only the original biodiversity will be kept in the wetland habitat, but also strengthen the water purification function of the wetland. Besides, the county government also agreed to dredge deeply in the lake area near the inflow of Jiang-Zhi-Liao Creak for the purpose of preserving an area as the site of constructing artificial inner lagoon and artificial vegetated floating islands inside the lake. The preserved area in this part was about 1 hectare. Thus, in sum the total lake area of about 4 hectares was preserved for constructing ecotechnological facililits in the future. In addition, landscape will be planned and designed in the same project in order to develop ecotouring in Mei-Non. This project has been approved by the Environmental Protection Administration of central government of Taiwan. It is believed Mei-Non village will become a beautiful scenic site for ecotouring and Ha-Ka culture touring in Taiwan in the near future.

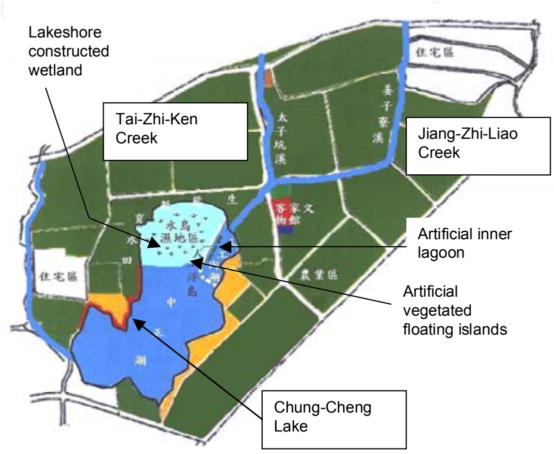


Figure 3 Schematic diagram of Chung-Cheng Lake manipulated by ecotechnology.

Monitoring water quality in Chung-Cheng Lake

As mentioned previously, two creeks are jointed in about 400 meters upstream of the inflow point into the Chung-Cheng Lake (CC Lake). Jiang-Zhi-Liao Creek (JZL C.) flows through some areas with high population density resulting in large amounts of sewage discharged into the creak. Before 2001, Tai-Zhi-Ken Creek (TZK C.) passed most of the feedlot areas in Mei-Non, where about 10000 pigs were rasied, resulting in large amounts of swine wastewater discharged into the creeks. However, due to policy of “Depress Livestock Industry in Water Source Areas” in 2002, the pollution pressure from swine wastewater in Tai-Zi-Ken Creak was less. Figs. 4 and 5 show the variations of NH_3-N and BOD concentrations at different sampling sites before and after the policy had been executed to decrease the pig raising number, respectively. According to these two figures, we found that no matter in the creeks or lake, the water quality was improved after the number of pig raised had decreased, which would be helpful for the ecotechnological facilities to achieve the function of purification of water quality of Chung-Cheng Lake by using artificial inner lagoon, artificial vegetated floating islands and construction wetland systems. This ecotechnological treatment facility is predicted to decrease down 10-50% of SS, 30-50% of total nitrogen (TN), and 30-90% of total phosphorus (TP).

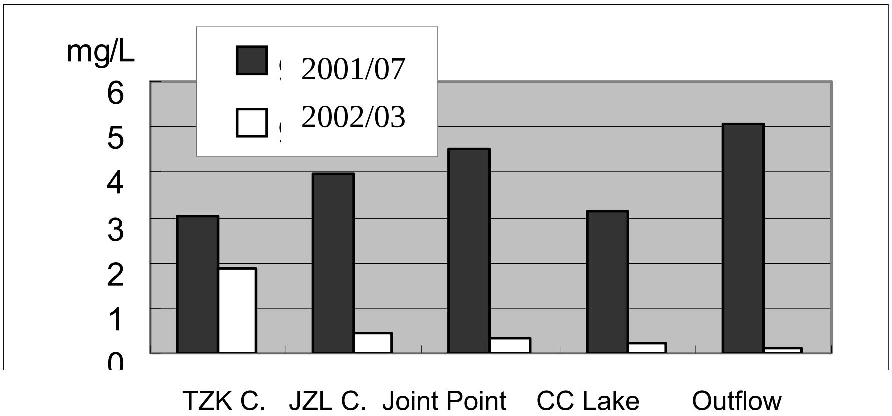


Figure 4 variations of NH_3-N concentrations at different sampling sites before and after the policy had been executed to decrease pig raising number.

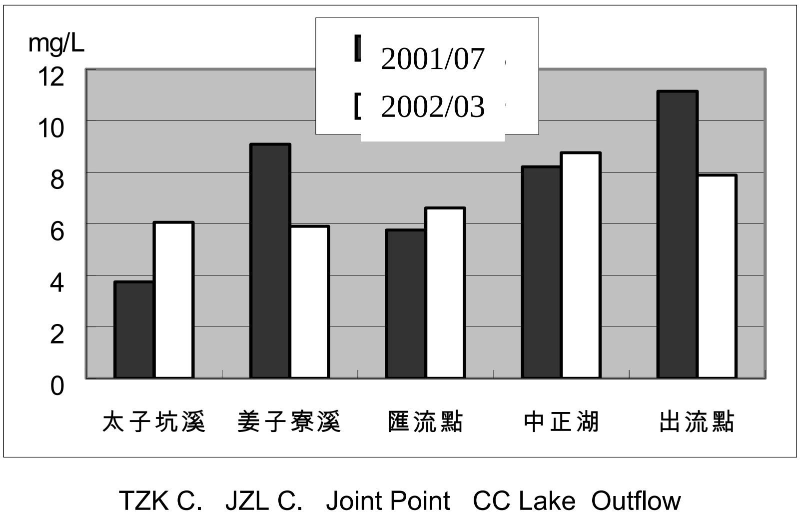


Figure 5 variations of BOD concentrations at different sampling sites before and after the policy had been executed to decrease pig raising number.

Monitoring water fowls in Chung-Cheng Lake

In an ecosystem, the information of number and species of birds exists index meaning. The more abundant of the bird species is, the higher the biological diversity is. According statistic information of surveying, 67 species of birds were recorded in Chung-Cheng Lake, which belonged to 10 orders and 24 families. In addition to some local bird species, a large number of migratory birds fly to here for feeding, resting, and nesting from October to March every year. The species of water fowls observed in Chung-Cheng Lake include Moorhen (*Gallinula chloropus*), Little Egret (*Egretta garzetta*), Great Egret (*Egreeta alba*), Grey Heron (*Ardea cinerea*), Black-crowned Night Heron (*Nycticorax nycitorax*), Little Grebe (*Tachybaptus rufioollis*), Shoveler (*Anas clypeata*), Mallard (*Anas platyrhynchos*) and Pintail (*Anas acuta*). However, the dominant water fowl species is Moorhen in Churn-Cheng Lake, which is a local bird in Taiwan. The variations of bird species and number observed in Chung-Cheng Lake in each month are generally coincident with the time for those migratory birds migrating to and passing through Taiwan.

Conclusion

It is concluded that ecotechnology can be an efficient, economical, and ecological (3E) way to restore polluted lakes caused by both eutrophication and organic pollutants. The improvement of water quality in Chung-Cheng Lake manipulated by ecotechnology including artificial inner lagoon, artificial vegetated floating islands, and constructed wetlands could not only clean the lake water, but also reserve multiple ecosystems, and further achieve biodiversity, which would increase the environmental valuation of Mei-Non. Besides, the special Hakka culture and relics in Mei-Non would also provide abundant touring resources. Ecotouring and Ha-Ka cultural touring is predicted to bring unlimited commercial value to Mei-Non in the near future.

Acknowledgement

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Reference

Bergen, S. D., 2001. Design principles for ecological engineering. *Ecol. Engr.* 18(2), 201-210.

Mitsch W. J. & Jørgensen S. E., 2004. Why ecological engineering and ecosystem restoration? In: *Ecological Engineering and Ecosystem Restoration*. pp. 3-22. John Wiley & Sons, Inc., Hoboken, New Jersey.

Nakamura K., Tsukidate M. & Shimatani Y., 1997. Characteristic of ecosystem of an artificial vegetated floating island. In: *Ecosystems and Sustainable Development*. pp. 171-181. Computational Mechanics Publications, Southampton.

Wu Z. B. & Yang Y., 2000. Water improvement on Donghu Lake by using constructed wetland. *Proceedings of the IWA 7th International Conference of Wetland Systems for Water Pollution Control.* pp. 344-352. Florida, USA.

Consequences of the climate change on the state of lake ecosystem in the sub-boreal zone of Europe (an example Lake Võrtsjärv, Estonia)

Arvo Järvet

Institute of Geography, University of Tartu; Vanemuise 46, 51014 Tartu, Estonia

Abstract

The North Atlantic Oscillation (NAO), which is defined through the variability of air pressure differences between the North (Iceland) and South (Azores), dictates climate variability over a large area of the Atlantic, North America and Europe, especially during winter. Variation in heat and moisture transport between the Atlantic and surrounding continents affect water balance components of lakes, such as precipitations, riverine inflow and evaporation, resulting in changes of the water level.

In this study we compared the impact of local climate variability by regional circulation indices to examine atmospheric circulation influence on the ecological state of shallow nonstratified Lake Võrtsjärv in Estonia. Lake Võrtsjärv is a naturally eutrophic lake in southern Estonia at 34 m above sea level with a surface area of 270 km^2, a maximum depth of 6 m, a mean depth of 2.8 m and a turnover time of 1 year. Shallowness of the lake causes the winter ice cover great importance for its ecosystem. The ice thickness affects the active water volume and the ice period duration affects the annual cycle of primary production.

The ecosystem of L. Võrtsjärv is very sensitive to water level fluctuations, which are following the pattern of NAO index refelecting the changes of climate in northern hemisphere. In Lake Võrtsjärv mild winters associated with high NAO index and secondary bring about higher water level. The years with less precipitation are associated with colder winters, when ice cover is thicker and stays longer.

For the ecosystem of L. Võrtsjärv the warmer and wetter climate could bring about higher water level in winter. The deeper the mixed water column, the lower is the average light intensity causing reduced phytoplankton biomass. In the deeper water both resuspension and denitrification rates are lower, the first reducing the phosphorus release from the bottom sediments and causing lower P concentration while the second raising the nitrogen concentration. Consequently, in warmer world the N/P ratio in Lake Võrtsjärv is higher and N_2 fixing cyanobacteria have less chances to develop. Due to climate warming the highest mean monthly nutrient losses in spring are shifted in 1990es to earlier period (from the beginning of April to mid March) and despite lower intensity of land use the wintertime nutrient losses may be almost as high as during previous years.

Key words: climate change, lake ecosystem, atmospheric circulation.

Introduction

Global climate change due to the increase of greenhouse gas concentration in the atmosphere is a factor expressing the anthropogenic stress on the water environment. The greatest warming is expected to take place in high latitudes. Water resources are very sensitive to climate change, and studies on this topic have been carried out for many regions in Europe. Understanding the sensitivity of water resources is the first and most important step in climate change integrated impact assessment (Figure 1).

There are two main study areas concerning investigations of climate change: 1) it is important to analyse the existing time-series of climate and related hydrological variables and to detect possible changes, 2) there is an obvious necessity to develop climate change scenarios to study future climate for vulnerability assessment. Impact research needs to address these problems, including the development of 1) a more physically based understanding of hydrological processes and their interactions, 2) hydrological parameter measurement and estimation techniques for application over a range of spatial and temporal scales, and 3) modular modeling tools to provide a framework to facilitate water management research (Leavesley, 1994).

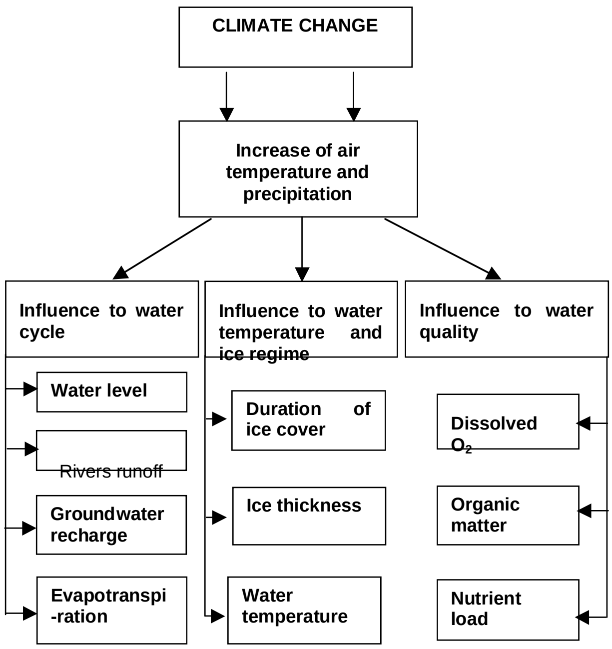


Figure 1. A simplified scheme of climate change impact assessment on the lake and its catchment area in sub-boreal zone.

In Estonia, the western airflow from the Atlantic during positive NAO remarkably increases air temperature and the amount of precipitation in winter. Warmer winters during positive NAO index should favour an earlier onset of the spring bloom and higher production in shallow lakes. Increased westerly winds increase precipitation and wind mixing. Light is reduced due to increased cloud coverage. At high water level in summer and autumn, light limitation is the main negative feed-back mechanism resulting in lower phytoplankton biomass. At lower water level with more light available, nutrient limitation takes over the control of phytoplankton.

Climate change impact on water-bodies

Possible changes in river runoff caused by climate change will certainly have an influence on water management (Becker & Lahmer, 1996). The problems arising for instance with low-flow and flood periods will be solved at the watershed (catchment) level, not only by administrative units (counties). Data on minimum runoff are needed, particularly in water supply and water quality control. Climate warming will be followed by positive as well as negative consequences for water management. Problems of water management associated with changes in river runoff arise mainly during low-flow and flood periods. They are estimated taking into account local conditions in different parts of Estonia (Table 1). This regionalization is made on the basis of hydrological parameters (annual range of monthly mean, maximum and minimum runoff, the character of its seasonal course) following general landscape peculiarities in Estonia. Changes in hydrological conditions in Estonia should also be translated into changes in the ecology of water-bodies, because a number of ecological processes are dependent on hydrological factors (Järvet, 1998).

Table 1. Comparison of climate change impacts on water bodies and water regime.

Changes	Positive impacts	Negative impacts
Increase in minimum winter runoff	Favourable oxygen conditions in water bodies	Unstable ice cover
Decrease in maximum spring runoff	Diminishing of floods in spring	Lengthening of the period with minimum runoff in summer, diminishing of water capacity in soils
Lengthening of the period with minimum runoff in summer	Better conditions in drained forest areas	Increase in productivity of high plants and algae in lakes and water reservoirs, increasing of evaporation from water surface
Increase in maximum runoff in autumn	Favourable ecological conditions in water bodies	More floods in autumn and inadequate drainage of agricultural land
Changes in agricultural discharge	Smaller peak flow in spring, diminishing of pollution load and wash-out of fertilisers	Problems for farmers during the harvest period in autumn
Water level changes	Decrease of flooded areas around shallow lakes during high water periods	Possible drop of water level below the ecologically optimal limit in shallow lakes for the second half of summer
Storage of lakes	Increase of total and active storage of lakes during winter period	

In general, all climate change scenarios for Estonia forecast mild winters, a decrease in snow cover and an increase in the duration of dry periods in mid–summer. The predicted global warming will cause more changes in the water balance elements in the cold period than in the warm season. Modelled annual runoff and its seasonal variability in the study area on the Lake Võrtsjärv catchment area are not very sensitive to climate change. Under the impact of climate change, a long-term series of annual precipitation will continue with a periodicity of 25–35 years. This will cause long-term fluctuations in hydrological regime and also the future alternation of wet and dry periods. No significant changes in the hydrological regime (annual runoff) of rivers in the drainage area of Lake Võrtsjärv caused by global climate changes have been observed.

Consequences of the climate changes on the state of lake ecosystem

It is clear that the scale of climate change predicted in the future will significantly alter the functioning of shallow lakes and seasonal patterns of water quality (Carvalho & Kirika, 2003). The North Atlantic Oscillation (NAO), which is defined through the variability of air pressure differences between the North (Iceland) and South (Azores), dictates climate variability over a large area of the Atlantic, North America and Europe, especially during winter. The variation in heat and moisture transport between the Atlantic and surrounding continents affects the water balance components of lakes, such as precipitation, riverine inflow and evaporation, resulting in changes in the water level. If present trends continue, limnologists believe that weather changes will have a major effect on the dynamics of lakes throughout Europe (Dokulil & Teubner, 2003). Most seriously, ecological changes should occur in the southern part of Europe, where a significant decrease in the mean monthly runoff for months is estimated, and the highest decrease in runoff will be during summer and especially during June (Mimikou *et al.,* 2000).

Typical climate-related problems include increases in lake productivity, increases in water colour and the increased frequency and severity of algal blooms. Many water quality problems that were once assumed, to be driven by the local weather are now known to be influenced by climatic events that operate on a global scale.

The ecosystem of L. Võrtsjärv is sensitive to water level fluctuations, which follow the pattern of the North Atlantic Oscillation (NAO) index, reflecting changes in climate in the northern hemisphere (Figure 2). Phytoplankton biomass was significantly lower in years of high water level and the changes were unrelated to nutrient loading. In low-water years, better water column illumination and increased release of phosphorus from resuspended bottom sediments results in substantially higher phytoplankton biomass than in high-water years (Nõges *et al.,* 2002).

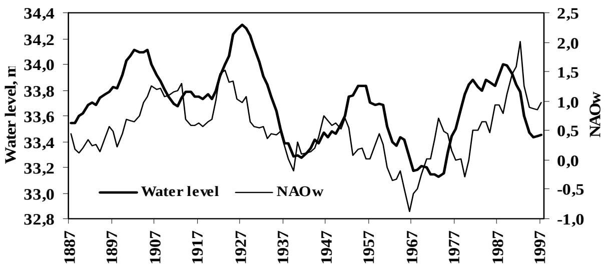


Figure 2. Relationship between NAO index for winter and annual mean water level in Lake Võrtsjärv presented as 7-year moving average values.

For the ecosystem of L. Võrtsjärv, the warmer and wetter climate could bring about higher water levels in winter. The deeper the mixed water column, the lower the average light intensity, causing reduced phytoplankton biomass. In the deeper water both resuspension and denitrification rates are lower, the first reducing the phosphorus release from the bottom sediments and causing lower P concentration while the second causes increased nitrogen concentration. Consequently, in a warmer world the N/P ratio in Lake Võrtsjärv would be higher and N_2-fixing cyanobacteria would have less chance to develop (Figure 3). Due to climate warming, the highest mean monthly nutrient losses in spring are shifted from the 1990s to an earlier period (from early April to mid March) and despite the lower intensity of land use, the winter nutrient losses may be almost as high as during previous years.

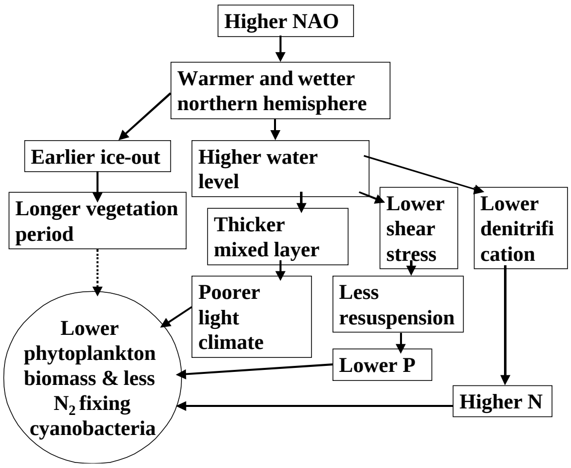


Figure 3. The assumed consequences of global warming on the phytoplankton of Lake Võrtsjärv (Nõges *et al* 2002).

The successions of phytoplankton composition were caused by changes in light and nutrient availability in the fully mixing environment (Nõges *et al.,* 2003). At high water level in summer and autumn, light limitation is the main negative feed-back mechanism resulting in lower phytoplankton biomass. At lower water levels with more light available, nutrient limitation takes control of phytoplankton. The mixed water column is better illuminated while the N:P ratio decreases, due to both increasing P concentration and decreasing N concentration.

It can be concluded that the positive impacts of climate warming on the ecological state of Lake Võrtsjärv are dominant (Table 1), and that all these modelled changes can be considered to be inside the observed natural variation of water balance elements during the baseline period from 1961–1990.

Correlation between circulation indices and ice regime parameters

In northern Europe, winter is the most critical period in the context of weather and climate change, as the greatest change in air temperature and precipitation are expected to occur during this period. The shift towards earlier spring events in lake ice cover is a global phenomenon that can be observed in lakes of the Northern hemisphere. On a global scale, lake ice parameters such as thickness, freeze-up and break-up dates and ice cover duration are good indicators of regional climate change in high-latitude regions. Earlier findings suggested a substantial impact of the North Atlantic Oscillation (NAO) on winter ice cover (Livingstone, 1999). In particular, the interannual variability of late-winter and early-spring temperature in Europe shows an association with the phases of the NAO. The positive phases of the NAO are associated with warmer and rainy late-onset winters and earlier springs in Estonia.

The circulation indices for Estonia calculated by O. Tomingas (2003) were used as a measure for regional-scale circulation. The zonal circulation indices are calculated using daily 5x5 degree gridded sea-level pressure data. It is the difference between standardized average pressure anomalies at three stations located in the south (52.5°N) and north (62.5°N) of Estonia. Positive values of the zonal index represent above-normal westerly circulation, and negative values indicate below-normal westerlies or even easterly airflow. The SW-NE index was calculated as a difference between the SLP at the southeastern and northwestern sides. The SE-NW index was calculated similarly, as a difference between the standardized SLP in the northeastern and southwestern corners of Estonia.

The impact of atmospheric circulation on ice cover duration acts indirectly through air temperature and snow cover conditions in winter. Local meteorological conditions such as type of snow cover and melting-refreezing sub-periods influenced the length of the ice cover period more strongly than the large-scale NAO. In the same conditions, air temperature and warm rainfall significantly affect the timing of snow cover, but do not have too much of an impact on ice break-up in the lake. In general, the ice break-up dates of Lake Võrtsjärv cannot be statistically connected with large-scale atmospheric circulation. Some statistically significant correlations between zonal circulation indices and ice cover parameters have been detected only for the ice thickness parameter (Figure 4).

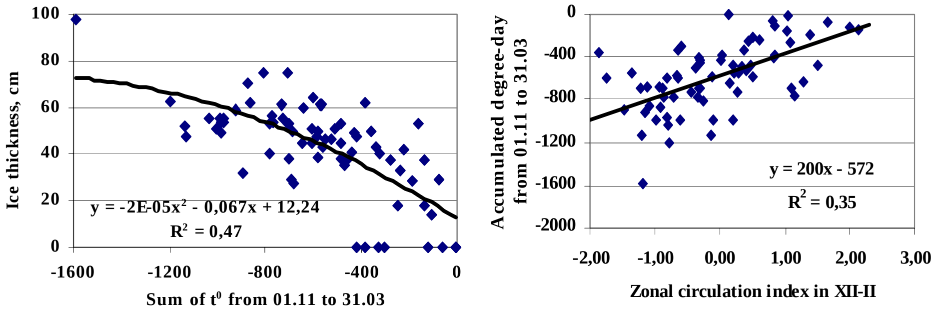


Figure 4. Correlation between zonal circulation index and accumulated degree-day.

The correlation between the zonal circulation index and ice thickness in Lake Võrtsjärv is negative from December to March (the highest correlation –0.44 in December) and for the winter season (D-J-F). The SW-NE index has a significant negative correlation with ice thickness in December, as well as for the winter season (D-J-F). The correlation between the SE-NW index and ice thickness is positive in January, February and during the winter (D-J-F). The meridional index had no significant correlation with ice thickness. These results can be explained by the influence of the meridional circulation index, which in most meterological stations in Estonia is statistically not correlated with winter temperatures (Tomingas, 2003). Our results indicate that the timing of ice break-up dates and the duration of ice cover were not closely associated with zonal circulation indices.

Perhaps in some cases it is possible to obtain a better understanding of the dependence of ice-freeze and ice cover break-up dates and ice seasons by using daily circulation indices. For example, an accumulated degree-day analysis based on daily temperatures offers a relatively good correlation with ice thickness (Figure 36) and zonal monthly circulation indices (Figure 5). The formation and break-up of the ice cover is assumed to occur when in every year the accumulated degree-day reaches a critical value (Yoo and D`Odorico, 2002).

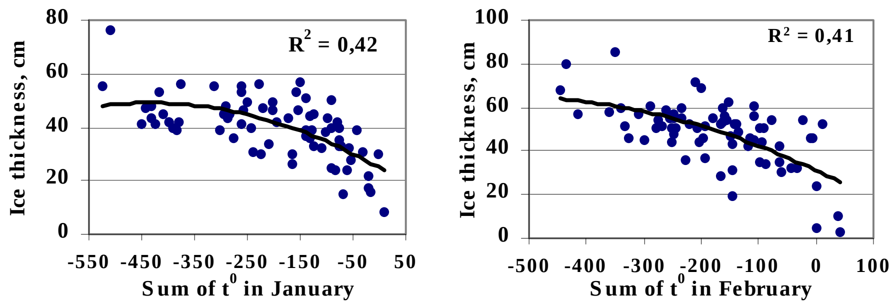


Figure 5. Ice thickness dependence at the ends of January (*left side*) and February (*right side*), from accumulated degree-day.

Nor was the correlation between the snow cover data and the meridional index statistically significant. Since ice formation in winter is closely connected to fluctuations in air temperature, the correlation between wintertime circulation indices and ice thickness in Lake Võrtsjärv follows a similar pattern. Nevertheless, it is obvious that the relationship between ice thickness and regional atmospheric circulation in winter is much more complicated than the direct relationship between atmospheric circulation and air temperature. Maximum lake ice thickness is not a very good direct measure of the severity of a winter, but rather a complicated function of snowfall and temperature patterns. Even with exactly similar frost sums and snowfall totals, the maximum thickness of lake ice may vary considerably (Kuusisto, 1994).

Our results indicate that the timing of ice break-up dates and duration of ice cover were not associated with zonal circulation indecies. The impact of the atmospheric circulation on the ice cover duration acts indirectly through air temperature and snow cover conditions in winter. Local meteorlogical conditions such as type of snow cover and melting-refreezing subperiods influenced the length of the ice cover period more powerfully than the large-scale NAO (Järvet, 1999). In the same conditions, the air temperature and warm rainfall significantly affect the timing of snow cover, but not too much impact on the ice break-up in lake.

References

Becker, A. & Lahmer, A., 1996. Klimawandel und seine Auswirkungen auf den Wasserhaushalt. *Wasser Special,* 137, No 14: 40–50.

Carvalho, L. & Kirika, A., 2003. Changes in shallow lake functioning: response to climate change and nutrient reduction. *Hydrobiologia*, 00: 1–8.

Dokulil, M. & Teubner, K., 2003. Klimaeinflüsse auf Seen in Europa (CLIME). *Österreichs Fischerei,* 56: 176–180.

Järvet, A., 1999. Influence of Ice Cover on the Ecological Conditions of Shallow Lake Võrtsjärv. *Publicationes Instituti Geographici Universitatis Tartuensis*, 84, 92–100.

Järvet, A., 1998. Estimation on possible climate change impact on water management in Estonia. *In*. R. Lemmelä ed. *The Second International Conference on Climate and Water*. Espoo, Finland, 1449–1458.

Kuusisto, E., 1994. *The thickness and volume of lake ice in Finland in 1961–90.* Publications of the Water and Environment Research Institute, 17: 27–36.

Leavesley, G.H. (1994).. Modeling the effects of climate change on water resources – a review. Climate Change, 28: 159–177.

Livingstone D., 1999. Ice break-up on southern Lake Baikal and its relationship to local and regional air temperatures in Sibiria and the North Atlantic Oscillation. *Limnol-Oceanography*, 44: 1486–1497.

Mimikou, M., E. Baltas, E. Varanou & K. Pantazis, 2000. Regional impacts of climate change on water resources quantity and quality indicators. *Journal of Hydrology,* 234: 95–109.

Nõges, T., P. Nõges & R. Laugaste, 2003. Water level as the mediator between climate change and phytoplankton composition in a large shallow temperate lake. *Hydrobiologia,* 00, 1–8.

Nõges T., Järvet, A., Laugaste R., Loigu E., Tõnno I., Nõges P., 2002. *Consequences of Catchment Processes and Climatic Changes on the Ecological Status of Large Shallow Temperate Lakes. Symposium on Conservation, Restoration and Management of Aquatic Ecosystems*. Bangalore: 159–171.

Tomingas, O., 2002. Relationship between atmospheric circulation indices and climate variability in Estonia. *Boreal Environmental Research*, 7, 463–469.

Yoo, J.-C., D`Odorico, P., 2002. Trends and fluctuations in the dates of ice break-up of lakes and rivers in Northern Europe: the effect of the North Atlantic Oscillation. *Journal of Hydrology,* 268: issues 1–4, 100–112.

Implications of climate change on the management of Rift Valley lakes in Kenya. The case of lake Baringo.

Ngaira, Josephine K.

School of Environment and Earth Sciences, Maseno University, P.O. Box 333, Maseno, Kenya
Email: ngaira@eudoramail.com.

Abstract

Climate patterns of the World became very variable during the last half of the twentieth century. Causes of this variability/change have been identified as: Sun Spot activity, ozone depletion, decline in the solar beam, with the more pronounced especially in Africa being Increased atmospheric carbondioxide, and albedo change due to anthropogenic factors.

The most affected weather elements by the above mentioned factors particularly in the tropics are Rainfall and Temperature, and the climatic environments most affected by the named weather elements are the arid and semi-arid lands which are already moisture constrained. The rift valley in Kenya, where most of the lakes are located experience Arid and Semi-Arid climate. The lakes located in this region are tectonic in origin, they are long, narrow, deep and salty except Baringo and Naivasha which are fresh water lakes.

Climate change/variability has caused both direct and indirect impacts on lake Baringo and its ecosystem. The direct impacts of extreme climate events (floods and droughts) include fluctuation of lake levels, salinity and aquatic life disturbance.

The indirect impacts which are anthropogenic in nature include; silting, soil compaction, illegal abstraction of feeder waters to the lake, change in aquatic species composition and famine.

In order to sustainably manage and utilize the waters of lake Baringo, there is need for a serious government policy on illegal water abstractions and massive afforestation of indigenous trees to enhance rainfall regularity.

Introduction

Climate change has been a major focus of attention for atmospheric and other scientists heading into the 21st century. Global climate started changing noticeably in the 2nd half of the 20th century (1960s) and has continued into the 21st century. The change is marked by high frequencies of extremes of rainfall and temperature compared with the 1900 – 1940 period when both temperature and rainfall were more equable in continental interiors and winds more consistent.

There was more erratic weather in the second half of the 20th century with the most vulnerable areas being the sub-humid and semi arid zones on the margins of the great deserts in Africa (Sahara) and Asia.

Possible causes of climate change

Multi – year weather anomalies which affect the whole globe or bigger portions such as the coldest century termed the “Little Ice Age” (1645 – 1715 years) and the “Little Tropical Age” (1870 - 1930) are likely to arise from a number of external causes such as sunspot activity and astronomical or Milakovitch theory. El nino Southern Oscillation (ENSO, increased atmospheric Carbon dioxide, depletion of ozone layer, deforestation and dust arise from anthropogenic causes.

The climatic factor of greatest economic significance in Africa, Kenya included is rainfall. The year-to-year and month-to-month variations are large and frequent in the region - (Table 1)

Table 1. Notable Regional Climate Anomalies In Kenya

YEAR	EVENT
1961	Extremely high equatorial rainfall in East Africa which led to East African lakes rising in levels to above all twentieth century record.
1968 - 73	Severe phase of drought (Sahelian drought) in Ethiopia and the Sahel region which caused severe famines, animal loss, desertification and reduced the size of Lake Chad --
1977	Highest rainfall ever recorded in Kenya which caused severe gulley erosion, Lake level rise in semi - arid regions
1984	Severe droughts in Kenya led to the introduction of “food queues” in supermarkets
1991 - 92	Severe droughts in most arid areas in Kenya led to the introduction of “food for work” programmes, and reduced surface areas of the lakes in the region.
1997 - 98	Kenya experienced the worst floods in 36 years caused by El-nino which increased surface area of Lakes in the rift valley.

The current debate on climate change has raised questions about possible effects of this change on the hydrologic systems. The Intergovernmental panel on climate change predicted that the warming of the Earth will affect precipitation patterns, evaporation rates, the timing and magnitude of run-off, the frequency and intensity of storms and therefore interfere with the quality and quantity of both surface and groundwater resources. The temperature and precipitation changes will affect the water demands for agriculture, industrial and domestic purposes.

The most vulnerable areas to climate change are the sub-humid (tropics) and the semi-arid zones on the margins of the great deserts of Africa (Sahara) and Asia.

Formation of Rift Valley Lakes

Two major theories attempt to explain the formation of the Rift Valley. One theory relies on the forces of tension while the other on the forces of compression. Both theories depend on up-warped swells along the sides of which faults develop. (Bunnet, 2003)

The most impressive rift valley system in the world is in the Eastern part of Africa where it is known as the Great Rift Valley of Africa. It extends for over 5600km.

The Western part of this rift valley passes through lakes Tanganyika, Mobutu (Albert) and Amin (Edward) while the Eastern part passes through lakes Magadi, Baringo, Bogoria, Nakuru, Naivasha and Turkana (Bunnet, 2003) Figure 1

The lakes found in Kenya's Rift Valley are therefore tectonic in origin, they are also known as fault lakes, where water is tapped in the down faulted valleys. These Lake were formed during the Miocene period, 25 million years ago.

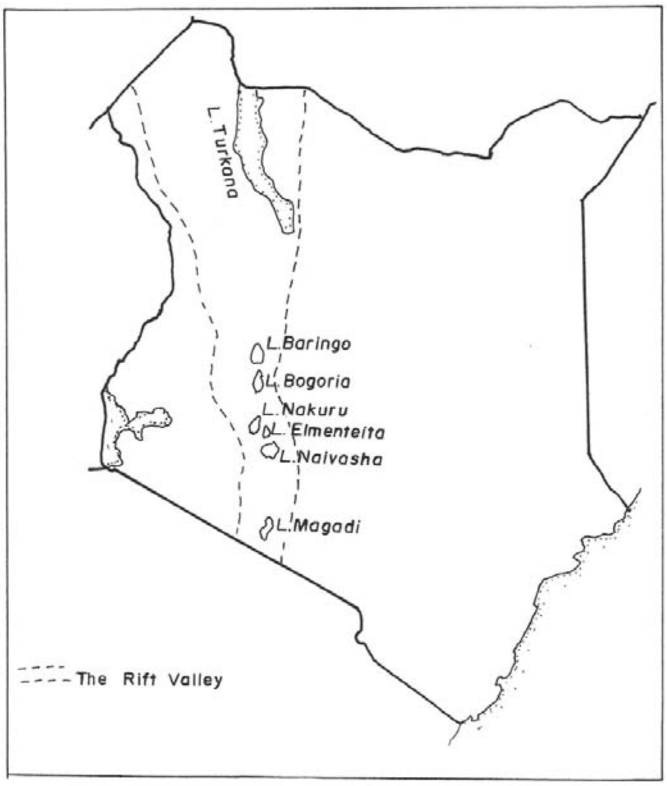


Fig I Location of Rift Valley Lakes in Kenya

Figure 1 Location of Rift Valley Lakes in Kenya.

Characteristics of Rift Valley Lakes

i. They are found on the floor of the rift valley at low altitude of between 900m to 1800m above sea level.

ii. They are Techtonic in origin and are also referred to as fault lakes.

iii. They are long, narrow, deep and with irregular shapes

iv. The climate varies from arid (L. Turkana and Magadi) to semi – arid as in Lake Baringo, Nakuru and Naivasha.

v. Annual rainfall ranges between 250mmin the North rift to 750mm towards the South.

vi. Annual Evaporation rates are very high ranging between 1800mm to 3500mm rendering most of the lakes salty except Lakes Baringo and Naivasha which are fresh due to the under ground sepage.

vii. Very high Temperatures reaching 41ºC and large annual diurnal temperature range of about 22°C are frequent as found in Lake Magadi where salt is mined.

viii. Because of their origin (faulty location on the floor of the rift valley) they have Tourists attraction sites. Hot springs in the Lake Bogoria, Flamingoes on Lake Nakuru.

The volumes of water in the lakes depend on the run off from seasonal streams. Management of lake levels is therefore dependent on rainfall, which is normally little and erratic.

The size and depth of these lakes is controlled by the precipitation received in the catchment area.

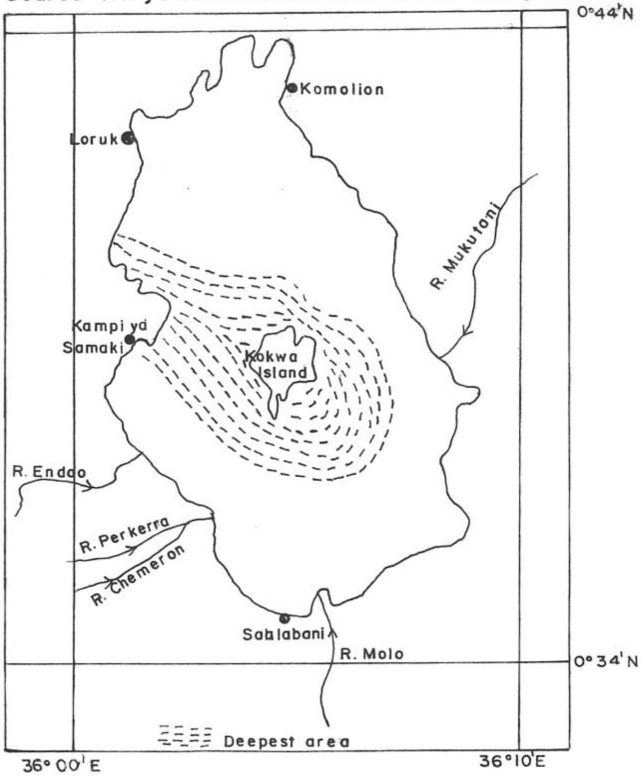
Lake Baringo (a case study)

Lake Baringo is a fresh water lake. It is located in a sparsely populated Marigat division of semi-arid Baringo district, in the Rift Valley of Kenya. It lies at an altitude of 915m above sea level, latitude 0 º44 ´ and 0º 34 N and longitudes 36º 00´ E and 36 º 10´E.

The main rivers are Perrenial rivers Perkerra and Molo, the seasonal rivers include Endao, Mukutani, Or Arabel and Wesegese Figure 2.

Fig.2 Location of Lake Baringo

Source - Kenya Marine Research Institute - Marigat 1996



Climate

The area experiences low and erratic annual rainfall which varies between 500mm and 750 mm in most parts of the region, characteristic of semi – arid regions

Annual evaporation rates vary between 2000mm and 2800 mm, which are much higher than the annual rainfall

The dominant controls of weather and climate in the region and East Africa as a whole are: (i) The Inter-tropical convergence zone (ITCZ) (ii) Altitude (iii) Latitude.

The low latitude, coupled with Acacia Scattered vegetation (Semi – Arid) means that most of solar radiation escaping into space is radiated back to the surface by high carbon-oxide concentration in the atmosphere, leading to consistent high temperatures of between 35^0C to 39^0C for most of the year.

Methodology

The research was funded by the Institute of Research and Postgraduate Studies (IRPS) Maseno University Kenya in 1997.

The study sought to investigate the impact of climate change on the following.

(A) (i) Lake levels (size and depth)

(ii) Salinity levels

(iii) Aquatic life disturbance

(B) Implications of Climate Change on the Management of the Lake Under the following anthropogenic factors.

(i) Siltation

(ii) Grazing activities and soil compaction

(iii) Water abstraction

Data for the study was collected from the semi-arid Baring district.

Documented rainfall data was collected from the KARI Perkerra meteorological station, Nyinyang health center and Lake Baringo weather stations.

Data on Lake levels was collected from: Baringo District hydrology Office, District Water engineer's Office and Kenya Marine Research Institute (KMRI) in Kampi Samaki.

Results and discussion

(i) Lake depth

Table 1 show that lake Baringo is shallow. It attained its maximum depth of only 5m in 1978. The lake depth remained below 3m for most of the period between 1973 and 1976. 1973 was the end of the Sahelian drought (1968-1973), which affected most lakes in Africa. The lake depth has been steadily declining due to the prolonged drought, which commenced in the 1990's.

The lake levels i.e. size and depth keep fluctuating depending on the amount of rainfall received in the region.

Table 2. summarizes the implications of rainfall variation on lake Baringo size and depth between 1970 and 1995.

Table 2. Annual rainfall totals and fluctuating lake level 1970-1995.

Year	Rainfall (mm)	Area (km^2)	Depth (m)
1970	650	153	3.7
1971	740	153	3.8
1972	530	151	3.0
1973	410	142	1.6
1974	700	134	2.2
1975	730	134	2.3
1976	450	132	2.2
1977	1030	129	4.0
1978	690	129	5.1
1979	570	143	4.7
1980	390	143	3.6
1981	650	144	3.7
1982	640	147	3.8
1983	650	147	3.9
1984	250	142	1.9
1985	510	144	2.4
1986	580	132	2.2
1987	890	138	2.2
1988	880	138	3.1
1989	590	141	3.0
1990	520	130	2.9
1991	340	130	2.2
1992	280	128	2.0
1993	330	118	1.9
1994	330	112	1.8
1995	320	112	1.7

(ii) Evaporation and lake water quantity and quality management.

Evaporation is extremely high ranging between 2100mm and 2800mm annually.

Table 3 summarises the relationship between mean annual rainfall and Evaporation 1970-1990.

Table 3. The relationship between mean annual rainfall and Evaporation 1970-1990.

Year	Rainfall mm X	Evaporation (mm) X
1970	650	2500
1971	740	2400
1972	530	2350
1973	410	2600
1974	700	2300
1975	730	2100
1976	450	2400
1977	1030	2050
1978	690	2060
1979	570	2500
1980	390	2800
1981	650	2200
1982	640	2150
1983	650	2400
1984	250	2600
1985	510	2200
1986	580	2600
1987	890	2600
1988	880	2000
1989	590	2400
1990	520	2800

Rainfall-evaporation balance= $\frac{\text{Mean annual rainfall} \times 100}{\text{Mean annual evaporation}}$

Calculated Rainfall evaporation balance=24%

Implications

i. There is high water deficiency in the Lake Baringo Catchment area.

ii. Most of the rivers which drain into the lake are seasonal-leading to fluctuating levels of the lake.

iii. Depth of the lake drops drastically during dry seasons because seasonal rivers dry up (Endao, Wesegese, Or Arabel) giving a shortage.

iv. Lake Baringo's water are risky, unreliable source for irrigation agriculture.

v. High evaporation rates make the water alkaline (mild salty) during droughts.

vi. High evaporation rates in relation to rainfall is one of the causes of fluctuating lake levels.

vii. The Lake water can only be used for irrigation during the rainy season.

viii. The lake Baringo is under recession as rainfall and stream flow fall by about 40 milliom m^3 per year short of evaporation (Messny, 1977). The short fall lead to annual drawback of lake Baringo level by about 270mm

ix. A rainfall – evaporation balance less than 50% indicates a water deficiency condition for continous stream flow and balanced lake levels.

Table 4. Drought year in Baringo basins

Mean annual 607m

Year	**Annual rainfall (mm)**
1970	680
1971	775
1972	558
1973	Drought corresponding to Sahalian 433 drought
1974	730
1975	710
1976	477 Drought
1977	1030
1978	727
1979	650
1980	414 Drought
1981	679
1982	670
1983	680
1984	260 Drought
1985	716
1986	510 set of drought
1987	580
1988	890
1989	880
1990	590 Drought
1991	650
1992	340
1993	280 Prolonged drought
1994	330
1995	330

Source - Marigat meteorological station 1996. Droughts occurred after every 2-3 yrs in the 1970s and 1980s. 1990s received prolonged draughts exceeding 4 consecutive years.

(iii) Siltation

One important factor leading to rapid recession of Lake Baringo in recent years is siltation caused by variability in rainfall. The soil in the Baringo catchment is very loose with no bedrock for depths of between one and two metres. Layers of exposed fine sediments are a common feature in the area. This makes the catchments area highly vulnerable to water erosion.

The many rivers which flow from the southern highlands into the lake such as Molo, Perkerra, Chemeron, Endao, Or-Arabel and Wesegese are very turbulent during the rainy seasons (April-August), and they carry along silts, mud, gravels, rocks and uprooted trees which are all deposited in the Lake. Erosion and siltation in the lake are highest during rainy season.

Approximately 400 tonnes of silt is deposited in the lake during every rainy season.

Siltation has compromised the quality of lake Baringo water which is ever muddy and brown in appearnce.

(iv) River abstraction

Lake Baringo is endangered because too much of its water has in the recent past been abstracted for irrigation purposes. There are many illegal abstraction of surface water flowing into the lake. Many storage reservoirs (dams) have been constructed along both perennial and seasonal rivers flowing into the lake. This has been done to support the many small scale self help irrigation projects started in the areas as a strategy for copying with frequent droughts (District hydrologist, personal communication, 1987).

It is estimated that the annual average surface inflow into the lake from rivers and streams between 1970 and 1985 was about 2200mm, while the annual average inflow between 1990-95 was reduced to 1150mm (District Hydrologist, 1996). This means that about 1050mm of water supposed to reach Lake Baringo as surface inflow every year is withdrawn and diverted to seasonal irrigation farms. Because of the abstractions and diversions, some of the rivers do not reach the lake anymore. Leading to reduced lake depth and surface area. For example, River Molo waters no longer reaches the lake during dry season because of too many abstractions along its course (Table5). River Mukutani has become very small since the establishment of Mukutani irrigation scheme in 1989. Its water rarely goes beyond the Mukutani scheme. Rivers Chemeron and Endao waters reach the lake only when there are heavy floods during the rainy season (Fisheries Department Report, 1996). All the irrigation projects found along seasonal Rivers Chemeron, Endao, Mukutaani and Or-Arabel operate during the rainy seasons only.

Implications on management

The abstracted water for self help irrigation schemes are far more than approved by the water engineer, this is because the decision of issuing permits for water abstraction is made at the central government (Nairobi) without consulting the local experts.

The abstractions on lake Baringo catchments area have been done without proper knowledge of the lakes water balance and without any assessment of the sustained yields of the rivers (District Hydrologist, 1996).

The subsurface outflow to Kapedo hot springs is 32million m^3 per year and this is what keeps the lake slightly saline to fresh.

Table 5. Surface water abstraction in Baringo Basin.

River	Dam Name	Irrigation Project	Location	Year when Started
Chemeron	Chemeron	(i) K.V.D.A ((ii) Losekem (iii) Lamalok	Endao Salabani Salabani	1984 1985 1986
Endao	Endao	(i) Salabani (ii) Endao	Njemps Endao	1989 1989
Wesegese	Sandai	(i) Sandai	Loboi	1988
Or Arabel	Kiserian	(i) Kiserian	Mukutani	1990
Mukutani	Mukutani	(i) Mukutani	Mukutani	1989
Molo	Loboi	(i) Kapkuikui (ii) Kamaskoi (iii) Eldume (iv) Nyoro	Loboi Loboi Eldume Loboi	1985 1988 1988 1989

Source- Ministry of Agriculture and Ministry of Water Development-Baringo District 1996.

Aquatic life changes

Fish Production in Lake Baringo fluctuates between dry and wet years. The severe drought of the 1990s reduced the production to very low levels. The worst decline was during the prolonged drought of 1990 to 1995, which led to a ban on fishing from the lake due to shortage of fish in the waters. The 1967 – 1972 drought reduced fish catch from the lake by 83%.

Conclusion

- The study established that Lake Baringo is under recession due to variability in rainfall caused by climate change.
- The depth of the lake dropped from 5M in 1978 to 1.7M in 1995 due to prolonged drought spells and water abstraction necessitated by the changing climate trends towards drier conditions.
- The lake has become slightly saline due to high evaporation rates caused by high temperatures which reach 39ºC.
- Fish production in the lake dropped by 70% between 1978 and 1995 due to reduced lake depth and poor water quality.

References

Bunnet R.B., 2003. Physical Geography in diagrams for Africa. Longman Group Ltd.
Messny M., 1977. Irigation Potential in Baringo and Kerio – valley water. MOWD Vol.2 Nairobi.
District Fisheries Department Kampi Samaki 1996.
Ministry of water Development Repor. Baringo. 1987.

Limnology of mining lakes: case study of Aguas Claras, Brazil

Eduardo von Sperling[1], César Augusto Paulino Granchamp[2]

[1]Department of Sanitary and Environmental Engineering, Federal University of Minas Gerais, Av. Contorno 842 / 7th floor ; 30110-060 Belo Horizonte Brazil
Email: eduardo@desa.ufmg.br
[2]MBR (Minerações Brasileiras Reunidas): Av. da Ligação 3580 ; 34000-000 Nova Lima, Brazil

Email: cpg@mbr.com.br

Abstract

Mining lakes form a new type of aquatic environment, which has been so far very little explored in the technical literature.These lakes are generally narrow and deep, presenting hence a meromitic behaviour.Most of the technical papers related to the ecology of pit lakes deals with the formation of acidic environments, which are very common in Central Europe.This paper presents the general limnological aspects during the formation of a pit lake from an iron ore mining activity.Lake Aguas Claras is located near the city of Belo Horizonte, Brazil, in the geological region known as Iron Quadrangle.Since the original rock is hematite, which is free from sulphur content, there are no problems related to acidic drainage.The water used for the filling of the lake comes from rain, ground water and the complementary pumpage from a close river.At its final stage, which will be reached around year 2010, Lake Aguas Claras will have a surface area of 0.67 km^2 and the impressive depth of 234 m, which will make it the deepest lake in the country.An extensive monitoring program (monthly samples) has been carried out since 2001.The results show a very good water quality, practically free from contaminants.The most conspicuous aspect in the limnology of Lake Aguas Claras is the frequent shift in algae dominance, with prevalence of Chlorophyta, Chrysophyta and Pyrrophyta.Regarding the composition of the zooplankton community, a clear alternation in the dominance of Rotifera and Crustacea can be observed.One of the most relevant issues in the environmental study of Lake Aguas Claras is the destination of the water body.Due to the prognosis of a good water quality the possible uses of the lake will be directed to recreation (swimming, diving, sailing, fishing), amenity value and water supply.

Key words**:** limnology; mining lakes; water quality

Introduction

Mining lakes form a new type of aquatic environment, which has been so far very little explored in the technical literature. They are generally narrow and deep, presenting hence a meromitic behaviour. This feature restricts hydrobiological growth and biodiversity in these habitats. Most of the technical papers related to the ecology of pit lakes deals with the formation of acidic environments (Klapper & Schultze, 1995; Miller *et al.,* 1996, Levy *et al.,* 1997; Geller *et al.,* 1998; Stevens & Lawrence 1998; Packroff 2000; Lessmann *et al.,* 2000; Kalin *et al.,* 2001; Boland & Padovan, 2002; Hindak & Hindáková, 2003).

This paper gives information about the water quality during the creation of a pit lake generated after the exhaustion of an iron ore mining activity (Lake Aguas Claras).The morphology of the lake points out a very high value of *relative depth* (25 %), according to Håkanson, (1981), indicating the existence of a meromitic behaviour and the probable anaerobic condition that will be obtained at the bottom of the water body. Meromoxis in mining lakes has been discussed in Stevens and Lawrence (1998).However, due to the high maximum depth, phosphorus remobilization should not reach the euphotic zone, preventing hence the onset of an eutrophication process.

Materials and methods

A monthly monitoring programme has been carried out since the first steps of the lake formation (August/2001).The most relevant physical, chemical and biological indicators for the evaluation of the water quality have been continuously analysed. All employed analytical methods are based on the recommendations of the *Standard Methods for the Examination of Water and Wastewater* (APHA, 1998).Due to the small surface of the lake, there is just one sampling point, which is located in the central part of the water body, corresponding to its maximum depth. Samples have been taken at the surface (Secchi depth) and at the bottom of the lake.

Results and discussion

A synthesis of the results of the water quality monitoring in Lake Águas Claras is presented bellow. For the most relevant parameters a graphic representation is inserted.

Water temperature: a clear seasonal distribution of temperature values can be observed (Figure 1).The stratified condition dominates most of the year, while circulation takes place only during winter months (June to August).This *monomitic* behaviour is a typical feature in the majority of tropical lakes.

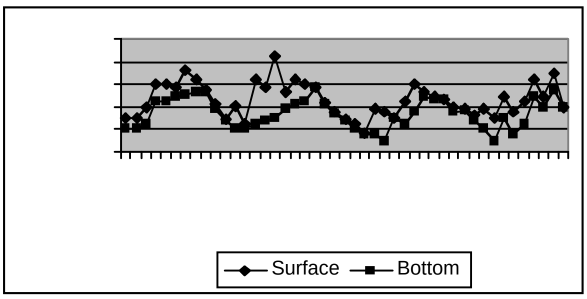


Figure 1. Water temperature.

Dissolved oxygen; there is a marked influence of the temperature in the rate of atmospheric oxygen transfer to the water, with higher values being obtained in colder months (June to August) (Figure 2).Moreover the algae photosynthetic activity leads to the prevalence of higher concentrations in the upper layers, with occasional records of supersaturation.

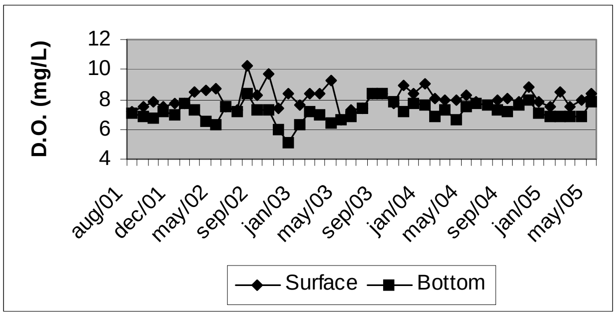


Figure 2. Dissolved oxygen.

Alkalinity: values between 24 mg/L and 39 mg/L;
Hardness: values between 39 mg/L and 55 mg/L;
Turbidity: generally low values, under 24 NTU, with marked seasonal variations (increase in the rainy period);
Colour: very low values, most of them under 1 mg/L;

Secchi depth: values between 0.5 m and 4.5 m; there is an upward trend in the clarity of water as long as the lake is being filled (Figure 3);

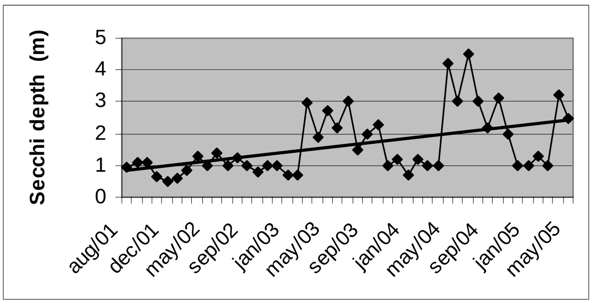


Figure 3. Secchi depth.

pH: ranges from 6.9 to 9.6 (Figure 4), with higher values being registered at the surface of the lake (primary production, CO_2 absorption) in comparison with the bottom (decomposition of organic matter, CO_2 release);

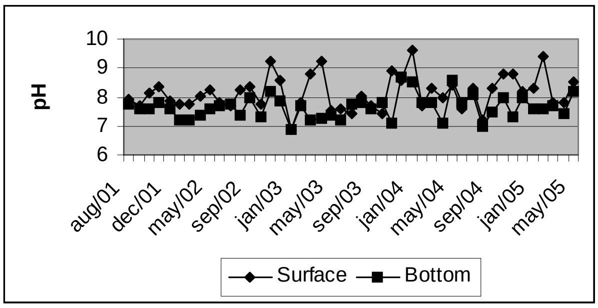


Figure 4. pH.

Electric conductivity: Values oscillate from 68 μS/cm (bottom) to 113 μS/cm (surface); since May/04 they are situated in the narrow range of 70-80 μS/cm, indicating that a certain stability in the amount of dissolved salts has been reached;

Nutrients

Total phosphate concentrations (Figure 5) show a slight trend of higher values (max. 0.18 mg/L) at the bottom layers during stratification periods, according to other experiences in tropical lakes (Tundisi & Saijo, 1997).These concentrations cannot be considered as elevated, since there is a noteworthy background presence of phosphate in the soils of the geological region of the State of Minas Gerais. Almost all values of soluble phosphorus are below 0.01 mg/L, with a maximum concentration of 0.02 mg/L. In a future scenario this soluble fraction will probably predominate at the bottom of the lake as a consequence of internal fertilization processes.

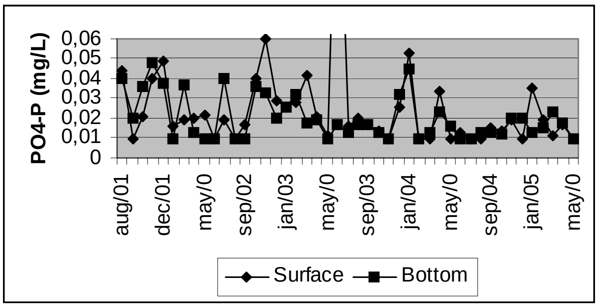


Figure 5. Total phosphate.

Ammonium nitrogen shows values between < 0.05 mg/L and 0.4 mg/L (Figure 6), while for nitrate nitrogen the concentrations range from < 0.01 mg/L to 1.3 mg/L (Figure 7), with a clear dominance of the oxidized fraction (nitrate) over the reduced one (ammonium), that is consistent with the good oxygenation conditions in the lake.

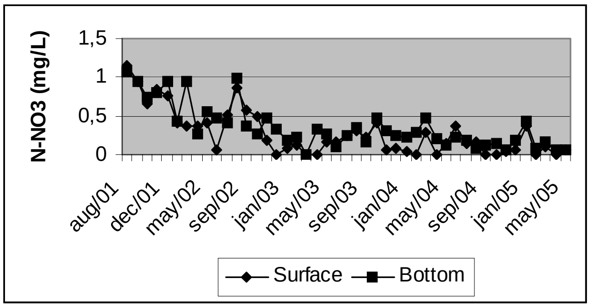


Figure 6. Ammonium Nitrogen.

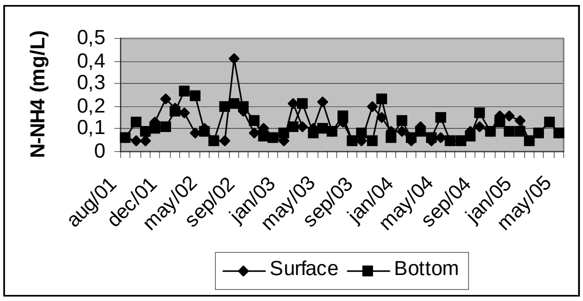


Figure 7. Nitrate Nitrogen.

BOD: values range from < 0.1 mg/L to 4.3 mg/L, with about 80 % of the results under 1 mg/L;

Fe and Mn: Iron concentrations oscillate from < 0.05 mg/L (surface) to 1.73 mg/L (bottom). Such values are typical for drainage basins with high iron contents from geochemical origin, as is the case of Lake Aguas Claras; manganese values range between < 0.05 mg/L (60 % of the results) and 0.17 mg/L (at the bottom);

Chloride: permanently very low values, from < 0.25 mg/L (about 50 % of the results) to 1.7 mg/L;

Heavy metals and other pollutants (phenols, oil and grease, cyanide): virtually absent, only aluminium has been occasionally detected (0.12 to 0.22 mg/L)

Bacteriology: very good bacteriological quality; about 90 % of the results of faecal coliforms, *Escherichia coli* and faecal streptococci are lower than 2 MPN/100mL.

Hydrobiology

A conspicuous shift in the dominance of phytoplanktonic groups can be observed (Figure 8). There is a general prevalence of Chlorophyta (35 % of the population), followed by Chrysophyta (32 %) and Pyrrophyta (23 %).Cyanophyta algae, in spite of being present only in small densities – 5 % - (with the exception of the first monitoring month, August/01, when it was dominant) is always a serious concern in Brazilian lentic waters, since the first registers of human deaths due to ingestion of cyanotoxins happened in 1996 in the city of Caruaru, Brazil (Azevedo *et al.,* 1996).

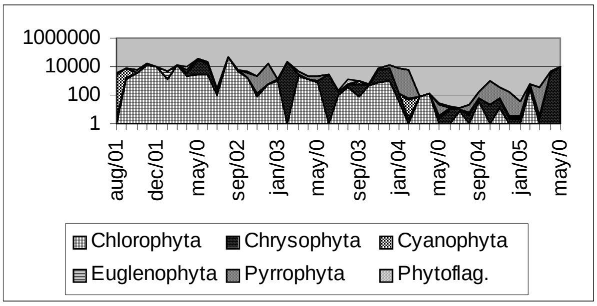


Figure 8: Phytoplanktonic groups.

These frequent alternations in the algae dominance is typical of aquatic systems that are undergoing a process of formation, such as mining lakes. Due to an enhanced nutrient concentration in the dry season there is a trend in obtaining higher algal densities in the winter time (May to August).Phytoplankton densities in Lake Aguas Claras varied in the amplitude of 8 ind/10 mL (July/04) to 42,223 ind/10 mL (August/02) (Figure 9).

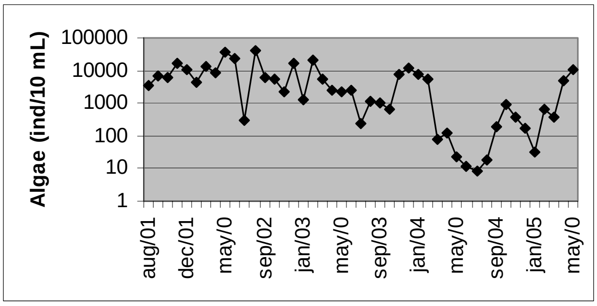


Figure 9: Algal populations.

The occurrence of phytoplankton peaks in the period following the end of the rainy season (March to May) is a typical feature in some Brazilian lentic systems (Esteves 1998, Pinto-Coelho *et al.,* 2003), possibly as a consequence of the onset of favourable limnological conditions (decrease in turbidity, weaker winds) after the end of the wet period.

With respect to the zooplankton, changes in domination are also noteworthy, with a general prevalence of *Crustacea* and *Rotifera* (Figure 10).Protozoa are found only scarcely.

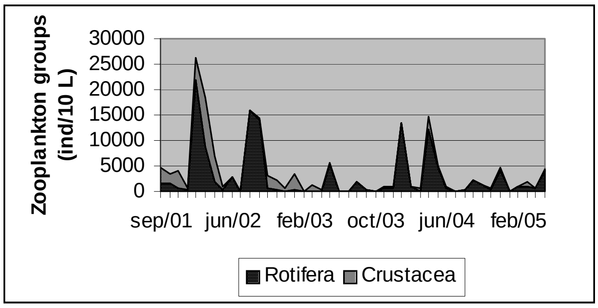


Figure 10: Zooplanktonic groups.

Regarding zooplankton populations (Figure 11) there are marked density variations (from 41 ind./10 L to 26,100 ind./10 L), which can be probably associated to the natural instability of the new aquatic system.Peaks in the zooplankton population have been observed in the dry period (winter time), that could be caused by enhanced salinity due to evaporation. Researches in Brazilian lakes have shown an increase in zooplankton abundance in the rainy season (Sendacz, 1984), while some authors present rain as a lost factor for the zooplankton (Campbell *et al.,* 1998).

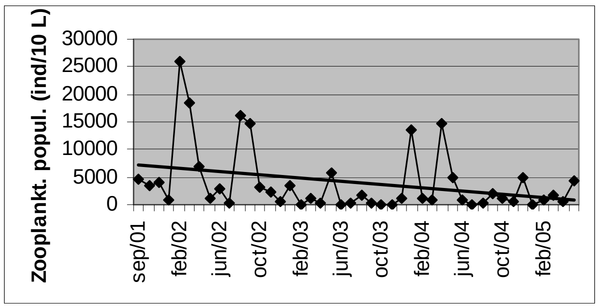


Figure 11: Zooplanktonic populations.

Conclusions

The evaluation of a monthly monitoring programme in Lake Águas Claras (period August/01-May/05) show the prevalence of a good water quality (well oxygenated, low values of colour and turbidity, limited degree of mineralization, pH slightly alkaline, low nutrient concentrations, excellent bacteriological conditions), together with a quite interesting shift in the dominance of phytoplanktonic groups, indicating the high instability of lakes that are undergoing a process of formation. Considering the probable maintenance of these favourable conditions, the possible uses of the lake will be directed to recreation (swimming, diving, sailing, fishing), amenity value and water supply.

References

APHA, 1998. *Standard methods for the examination of water and wastewate.* 20. ed., Washington DC, American Public Health Association

Azevedo, S.M.F.O., Evans, W.R., Carmichael, W.W. & Namikoshi, M., 1996 First report of mycrocystins from a Brazilian isolate of the cyanobacterium *Microcystis aeruginosa. Journal of Applied Phycology* 6, 261-265

Boland, K.T. & Padovan, A. V., 2002. Seasonal stratification and mixing in a recently flooded mining void in tropical Australia. *Lakes and Reservoirs: Research and Management* 7, 125-131

Campbell, C.E., Knoechel, R. & Copeman, D., 1998. Evaluation of factors related to increased zooplankton biomass and altered species composition following impoundment of a Newfoundland reservoir. *Canadian Journal of Fisheries and Aquatic Sciences* 55, 230-238

Esteves, F. 1998. *Fundamentos de Limnologia* (in Portuguese), Ed. Interciência, Rio de Janeiro, Brazil

Geller, W., Klapper, H. & Salomons, W., 1998. *Acidic mining lakes*. Springer, New York

Håkanson, L., 1981. *A Manual of Lake Morphometry.* Springer Verlag, Berlin

Hindak, F. & Hindáková, A., 2003. Diversity of cyanobacteria and algae of urban gravel pit lakes in Bratislava, Slovakia: a survey. *Hydrobiologia* 506, 155-162

Kalin, M., Cao, Y. Smith, M. & Olaveson, M. M., 2001 Development of the phytoplankton community in a pit-lake in relation to water quality changes. *Water Research* 35, 3215-3225

Klapper, H. & Schultze, M., 1995. Geogenically acidified mining lakes – living conditions as possibility of restoration. *Internationale Revue der Gesamten Hydrobiologie* 80, 639-653

Lessmann, D., Fyson, A. & Nixdorf, B., 2000. Phytoplankton of extremely acidic mining lakes of Lusatia (Germany) with pH < 3. *Hydrobiologia* 433, 123-128

Levy, D.B., Custis, K.H., Casey, W.H. & Rock, P. A., 1997. The aqueous geochemistry of the abandoned Spenceville copper pit, Nevada County. *California. Journal of Environmental Quality* 26, 233-243

Miller, G.C., Lyons, W. & Davis, A., 1996. Understanding the water quality of pit lakes. *Environmental Science and Technology News* 30, 118-123

Packroff, G., 2000. Protozooplankton in acidic mining lakes with special respect to ciliates. *Hydrobiologia* 433, 157-166

Pinto-Coelho, R., Bezerra-Neto, J.F., Giani, A., Macedo, C.F., Figueiredo, C.C. & Carvalho, E. A., 2003. The collapse of *Daphnia laevis* (Birge, 1878) population in Pampulha Reservoir, Brazil. *Acta Limnologica Brasiliensia* 15, 53-70

Sendacz, S., 1984. A study of the zooplankton community of Billings reservoir, São Paulo, Brazil. *Hydrobiologia* 113, 121-127

Stevens, C.L. & Lawrence, G. A., 1998. Stability and meromixis in a water-filled mine pit. *Limnology and Oceanography* 43, 946-954

Tundisi, J.G. & Saijo, Y., 1997, *Limnological studies on the Rio Doce Valley Lakes, Brazil.* Brazilian Academy of Sciences, Brazil

Wetzel, R., 1983. *Limnology*. Sounders Company, Philadelphia

Importance of aquatic macrophytes in controlling water quality of shallow lakes

Amano Kunihiko

Public Works Research Institute, 1-6 Minamihara, Tsukuba, Ibaraki, 305-8516 Japan

Email:amano@pwri.go.jp

Abstract

Aquatic macrophytes function in several ways in aquatic environment. I have analyzed the effects of aquatic macrophytes focusing on the reduction of resuspension of bottom sediments. Aquatic macrophytes are known to suppress the development of wind wave in shallow waters. Reduced wave height leads to the reduction of the resuspension of bottom sediments. This function that aquatic macrophytes may have seems important in deciding the water quality of shallow water bodies. A numerical simulation model was developed to estimate the quantitative effects of aquatic macrophytes on the water quality of a shallow lake. Simulation results showed good agreement with observed water quality data. A simulation case that assumed dense vegetation of which area was maximum in recent years resulted in less turbid and lower organic matter concentration in water column. The degree of reduction in suspended solids presumed by the simulation agreed well with the previous records in Inba lake, Japan that experienced the decay of aquatic macrophytes. Restoration of native aquatic macrophytes is important to restore the lake to a previous clear state.

Key words: aquatic macrophyte, resuspension, bottom sediments.

Introduction

Inba lake in Chiba prefecture, Japan became eutrophic shallow lake due to the urbanization in its basin. Its water quality has been recorded as the worst among reservoirs that are used for a source of city water. Annual average concentration of total phosphorus (T-P) and total nitrogen (T-N) in inflowing rivers rose to 0.5 mg/L and 6.0 mg/L, respectively in 1970's. The deterioration of water quality in inflowing rivers has called for the deployment of sewage treatment facilities and this decreased the concentration of T-P and T-N to 0.1 mg/L and 3.0 mg/L, respectively. In contrast to the improvement of inflowing river water quality, water quality indices such as Chemical Oxygen Demand (COD) and Chlorophyll-a (Chl-a) in lake has not been improved. Water quality in terms of COD, Chl-a, and suspended solids (SS) was worsen in particular in northern lake. This trend seems to be related to the environmental changes in northern lake. Inba lake was divided to northern and western lake after the reclamation in 1969.

One of the major environmental changes in northern lake is the decline of aquatic macrophytes. Macrophytes can play an important role in improving the water quality of shallow lakes and marshes by suppressing the resuspension of bottom sediments (Van den Berg, 1998; James, 2004; Scheffer, 1998). 13 species of submersed macrophytes were found in the survey conducted in 1977 in northern lake. However, submersed macrophytes gradually declined and canopy-forming *Trapa natas* became dominant covering almost entire northern lake in 1986. Wide spread *Trapa* was harvested thereafter because they annoyed fishery boat navigation. Only emergent plants along the shore and small patches of *Trapa* can be seen now (Kasai, 1993).

I have developed a numerical simulation model to estimate the suppression effect on resuspension of bottom sediments by macrophytes and analyzed the relation between the water quality deterioration and the decline of macrophytes.

Methods

Site description

Inba lake consists of northern lake, western lake, and drainage channel that connects two lakes (Figure 1). Average depth is 1.7 m, total area is 11.6 km^2, total volume is 27,700,000 m^3, area of the basin is 541 km^2, and total population in the basin is 718,000. Two major rivers that flow into western lake carry most of the nutrient loads to the lake. In particular, Shin river that flows into the lake through Aso bridge is most urbanized in its basin.

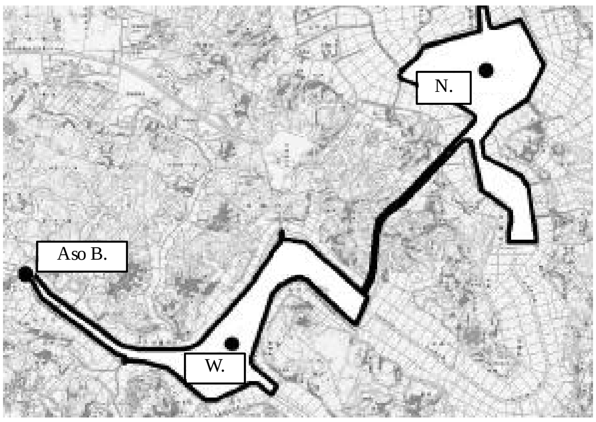


Figure 1. Plan view of Inba Lake Each point shows the location of water quality sampling

Model description

Three dimensional dynamics model was combined to water quality model and macrophyte model. Macrophytes can affect water quality of lakes through nutrient uptake, shading, providing place for microorganism to attach and for zooplankton and small fishes to hide themselves and allelopathy besides suppression of bottom sediments resuspension. However, since water quality change after the harvest of macrophytes in northern lake is characterized by the increased turbidity, only physical effect is considered in the macrophyte model.

Wind wave can be the most influential factor to affect the resuspension of bottom sediments in shallow lakes like Inba lake. The height and frequency of wind wave can be estimated by wind velocity, fetch, and water depth. Equations derived from Ijima and Tang (1962) are used to estimate wave height (*Hs*) and wave cycle (*T*) of wind wave if there is no macrophytes in the lake (Equations (1) to (6)). Patch of macrophytes can attenuate the wind wave and this leads to the suppression of resusupension of bottom sediments. Therefore, this effect should be considered in the wave model. The attenuation of wind wave by macrophytes can be estimated following Kobayashi and Asano (1992) (Equation (7)).

$$\frac{gH_S}{W_{10}{}^2} = 0.283\tanh[a]\tanh\left[\frac{g}{\tanh a}\right] \qquad (1)$$

$$\frac{gT}{W_{10}} = 2.8p\tanh[z]\tanh\left[\frac{d}{\tanh z}\right] \qquad (2)$$

$$a = 0.530\left(gh/W_{10}{}^2\right)^{0.75} \qquad (3)$$

$$z = 0.833\left(gh/W_{10}{}^2\right)^{0.375} \qquad (4)$$

$$g = 0.0125\left(gF/W_{10}{}^2\right)^{0.42} \qquad (5)$$

$$d = 0.077\left(gF/W_{10}{}^2\right)^{0.25} \qquad (6)$$

where, g(=9.81m S^{-2}) is the acceleration of gravity, W_{10}(m S^{-1}) is the wind velocity 10 m above the water surface and *F*(m) is fetch.

$$H_{Sm} = H_{Si}\cdot\exp\left(-k_i\cdot F_2\right) \qquad (7)$$

where, H_{Sm} is the wave height at a point in vegetation, H_{Si} is the wave height at the upwind margin of the vegetation, k_i is the attenuation coefficient (m^{-1}), and F_2 is the length between the point in vegetation and upwind margin.

The rate of resuspension of bottom sediments (*E*) is calculated following Luettich et al. (1990) (Equations (8) and (9)).

$$E = 0 \qquad t < t_C \qquad (8)$$

$$E = a\left(\frac{t - t_C}{t_{ref}}\right)^n \qquad t \geq t_C \qquad (9)$$

where, *a* and *n* are model parameters, t, t_C, t_{ref} are bottom shear stress, critical bottom shear stress, and reference bottom shear stress, respectively (unit; dyn cm^{-2}).

Since bottom shear stress can be approximated by bottom shear stress caused by wind wave, bottom shear stress can be estimated by wave height (Equation (10), (Luettich et al.,1990)).

$$t_{\text{wave}} = H\left[r\frac{\left(uw^3\right)^{1/2}}{2\sinh kh}\right] \qquad (10)$$

where $w \equiv 2p/T, k \equiv 2p/l$, ?($S^{-1}$) is wave frequency, *T*(S) is wave cycle, *k*(m^{-1}) is wave number, ?(m) is wave length, ?(g cm^{-3}) is the density of water, ?(=0.01cm^2 S^{-1}) is kinematic viscosity, *h*(m) is the water depth.

Calculated rate of resuspension is used as a source of SS at the bottom of each calculation grid of water quality model.

Vegetation attenuates not only wave height but also flow. Resistance caused by vegetation is considered by adding following term to the momentum equation in dynamics model (Kobayashi and Asano, 1992).

$$F = \frac{1}{2}rC_D bNu_m{}^2 \qquad (11)$$

where, *F* is resistant force to *x* and *y* direction, C_D is drag coefficient, *b* is projected area of plants in unit height to normal direction of the flow (=0.015m^2), *N* is the number of plants (100m^{-2}), u_m is velocity in *x* and *y* direction.

Drag coefficient can be estimated by Equation (12 and (13)) (Kobayashi and Asano, 1992).

$$C_D = \left(\frac{2200}{R_e}\right)^{2.4} + 0.08 \quad (2200 < R_e < 18000) \qquad (12)$$

$$R_e = \frac{bu_m}{n}, \qquad (n = 0.01) \qquad (13)$$

Velocity that is accompanied by wave is calculated by the model of Ijima and Tang (1962).

Model simulation

Model simulation was conducted using hydrologic and meteorological information between July 1, 2002 to March 31, 2003. Size of the horizontal grid is 50m X 50m and water column is divided to 5 layers vertically. Hourly data is available for required hydrologic and meteorological input of the simulation for above period. Since vegetation was almost disappeared during above period, water quality calculation was done in the condition without vegetation and compared with observed water quality data to calibrate the model.

Assuming the vegetation cover in 1988 (Figure 2), water quality simulation was conducted using same input data as one used in calibration. Because these is not available input data of 1988 for model simulation, I used the data of 2002 to 2003, instead. Although this calculation is like a sensitivity analysis of the effect of vegetation, it can assess the water quality in Inba lake with macrophyte cover.

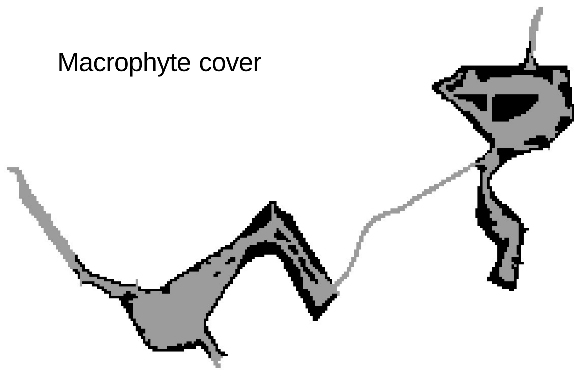


Figure 2 Vegetation cover in 1988

Results

Model parameters were calibrated using observed information, and then water quality change in Inba lake was simulated. Calculated water quality values seem in good agreement with those observed (Figure 3). Calculated SS showed rapid change caused by storms, however, since there is no observation while storm events, observed values look smaller than calculated ones. Observed COD values are high during summer. Simulated results show the similar trend, which seems to be the consequence of the frequent resuspension of bottom sediments. Resuspended particulate organic compounds which are contained in the bottom sediments of Inba lake seem to attribute to the increase of COD value. Also attributable to the high COD values during summer seems phytoplankton bloom. T-N concentration gradually increases from fall to spring, reflecting the high concentration of nitrate in inflow river water during this period. Simulated results show good agreement with observed. T-P change seems to be influenced by resuspension. Model simulation showed the strong influence of resuspension in not only SS but also COD and T-P.

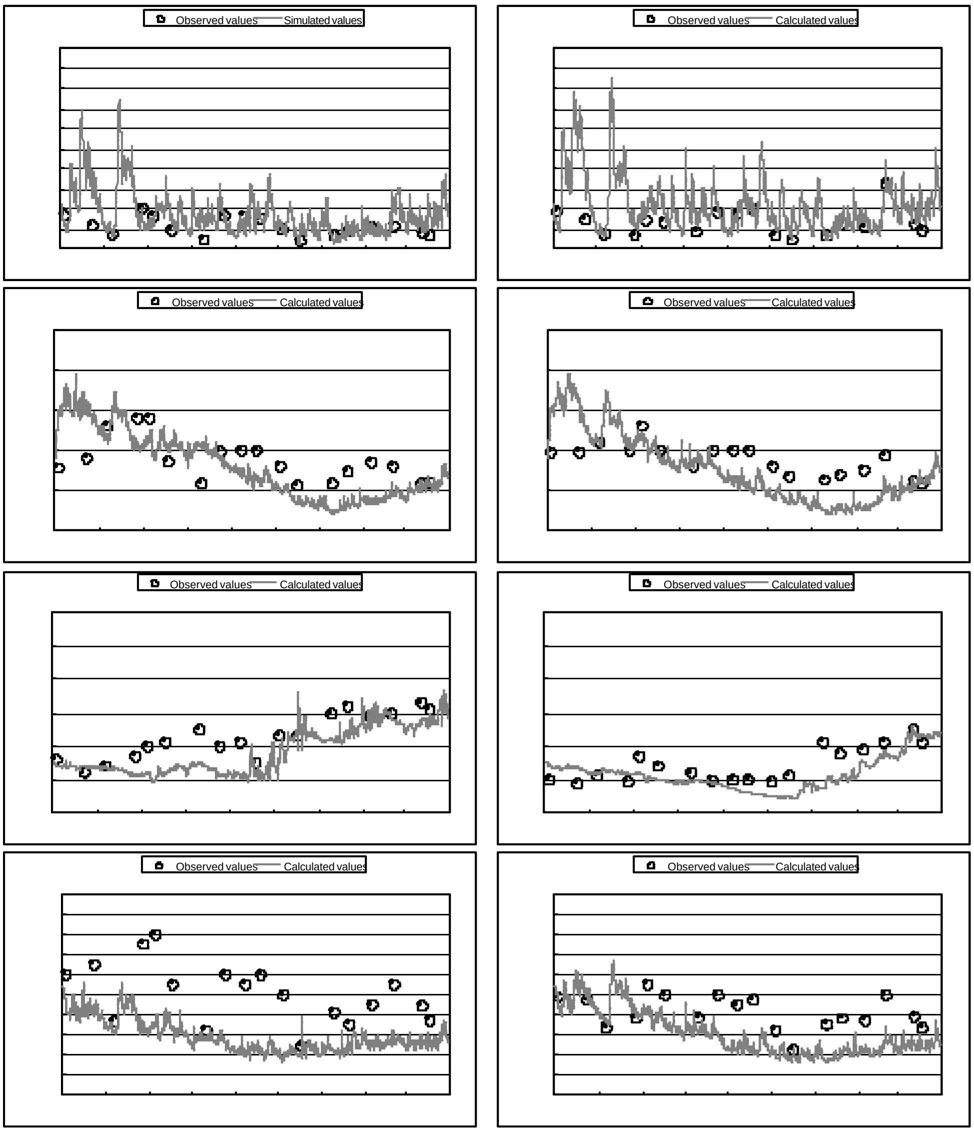


Figure 3. Water quality simulation with calibrated parameters and comparison to observed values (left; western lake, right; northern lake).

There was a storm event on July 14, 2002. Simulation showed that SS concentration increased in entire lake during this event (Figure 4a). However, result from the simulation case that assumes vegetation in 1988 seems different nevertheless it was conducted with same meteorological input (Figure 4b). The increase of SS concentration is significantly suppressed by the presence of aquatic macrophytes.

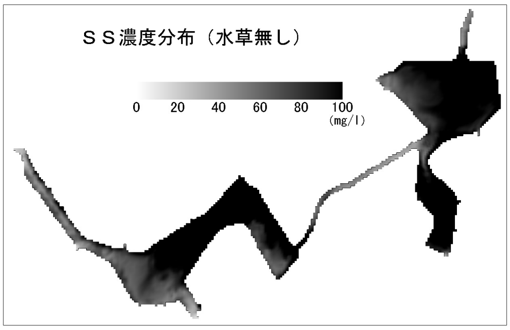


Figure 4a. SS distribution at storm (w/o vegetation).

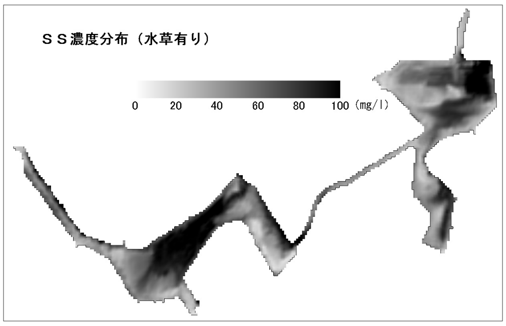


Figure 4b. SS distribution at storm (w/ vegetation).

Kasai (1993) reported that *Trapa* showed maximum coverage in 1986. More than 90 % of the surface of northern lake and 80 % of the surface of western lake was covered by the canopy of *Trapa*. The amount of *Trapa* started to increase conspicuously from 1984. Harvest of Trapa continued from 1987 to 1991. Water quality survey has been conducted twice a month in Inba lake. SS, COD and Chl-a concentration showed increase during after the harvest period (Figure 5). Observed data were averaged at each half month period. The period for analysis was divided to three (i.e. before harvest, after harvest, and 1986).

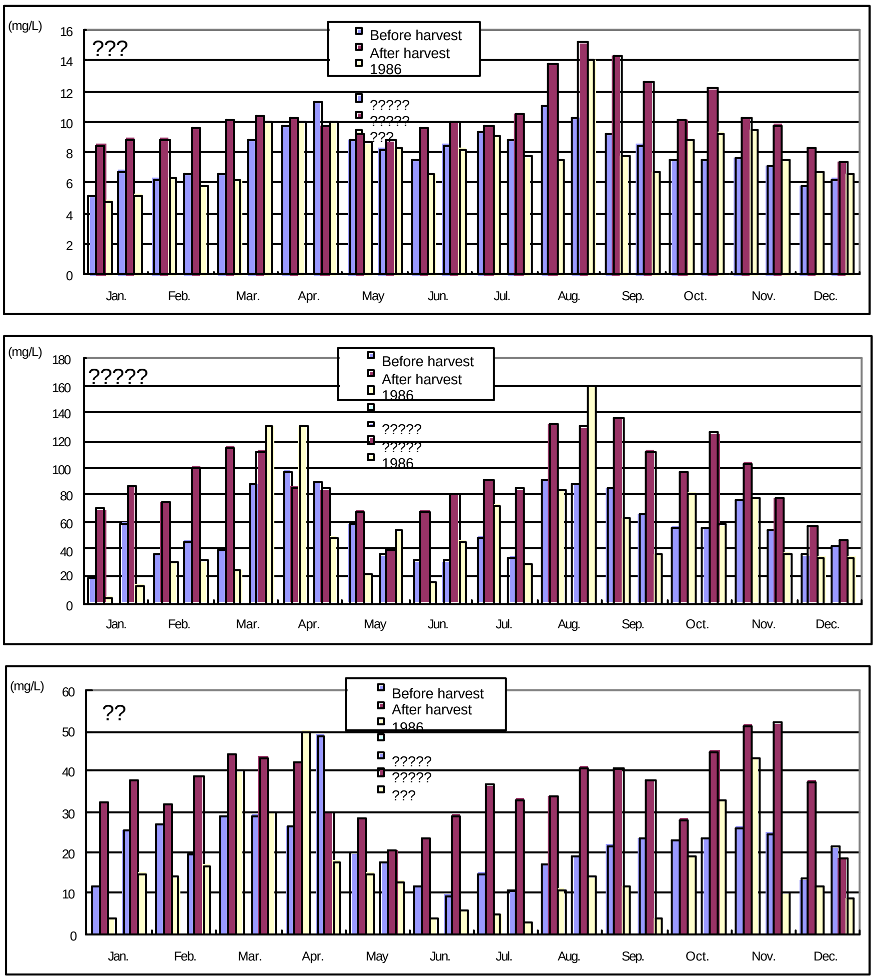


Figure 5. Observed water quality change in northern lake. Averages of half month period are compared.

Data obtained from 1985 to 1991 are used to calculate the average for the period of “before harvest”. From 1992 to 2000 are for “after harvest”. SS concentration was significantly low from June to September in 1986 when *Trapa* covered almost entire lake. Also suppressed was the value of COD except one in latter half of August. This exception can be explained by the bloom of phytoplankton. These results imply that recent water quality deterioration in northern lake is a consequence of the harvest of *Trapa*. Increased resuspension of bottom sediments after the harvest of *Trapa* may have shifted the lake to turbid stage (Scheffer, 1998).

Discussion

Scheffer (1998) categorized two stable stages for shallow lakes, one is clear stage with abundant macrophytes and the other is turbid stage with abundance of phytoplankton. SD transparency was about 80 cm before the harvest of *Trapa* in northern lake. However, it declined continuously after the harvest of *Trapa* and turned out to be about 40 cm in 1999. Since inflow river water quality was almost no change or slightly improved during this period, the decline of transparency cannot be attributed to the progress of pollution in the basin. Rather, it can be explained by the accelerated resuspension of bottom sediments. Resuspended bottom sediments of which particle size are small do not settle immediately and stay long in water column leading to the increase of SS. Resuspended organic particles raise the value of COD and released nutrients from bottom sediments enhance the growth of phytoplankton, which also increases COD. These increase of SS and COD can be a cause of turbidity and decline of transparency. Recent water quality deterioration in northern lake can be expressed by the disappearance of aquatic macrophytes (*Trapa*).

Simulation case in which vegetation cover of 1988 was assumed also indicated the suppression effect on SS and COD by macrophytes. Since enough input information is not available, simulation for the condition of 1988 was not conducted. Instead, simulation in which only vegetation cover condition was different was conducted. Although the results of this simulation cannot be compared to the observed values directly, statistics of them can be compared. Calculated annual average of COD was reduced by 2.3 mg/L in the simulation with vegetation. The annual average of COD of observed data was 10.3 mg/L after the harvest of *Trapa* and 8.1 mg/L before it. The magnitude of the effect of *Trapa* in reducing COD value seems reasonably simulated. The annual averages of observed SS were 35.7 mg/L and 21.5 mg/L after the harvest and before it, respectively. Those of simulated SS were 42.8 mg/L and 17.1 mg/L, respectively. Conformity between observed and calculated values implies the good estimation of the effect of macrophytes on water quality in Inba lake and supports the hypothesis that aquatic macrophytes maintained a clear stage in Inba lake.

Acknowledgements

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References

Ijima, T and F.L.W. Tang (1962), Numerical calculation of wind waves in shallow water, *Proc. 10th Conf. on Coastal Engineering, ASCE*, v.2, pp. 38-45.

James, W.F., J. W. Barko and M. G. Butler (2004), Shear stress and sediment resuspension in relation to submersed macrophyte biomass, *Hydrobiologia*, 515, pp. 181-191.

Kasai, S.(1993), Succession of aquatic plants in Inba lake, *In: Tega lake and Inba lake*, (eds. Y. Yamada, K. Shiratori and H. Tachimoto), pp. 57-64, Kokon Shoin, Tokyo

Kobayashi, N., A. W. Raichle and T. Asano (1992), Wave attenuation by vegetation, *J. Waterway, Port, Coastal, and Ocean Engrng.*,v. 119, ASCE, pp. 30-48.

Luettich, R. A., D. R. Harleman and L Somly?dy (1990), Dynamic behavior of suspended sediment concentrations in a shallow lake perturbed by episodic wind events, *Limnol. Oceanogr.*, vol. 35, n. 5, pp. 1050-1067.

Scheffer, M. (1998), Ecology of shallow lakes, *Kluwer Academic Publishers*, Dordrecht.

Van den Berg, M.S., M. Sheffer and H. Coops (1998), The role of characean algae in the management of eutrophic shallow lakes, *J. Phycol.*, 34, pp. 750-756.

Sustainable management of sediments at reservoirs – a comparative study from Asia, Africa and Europe

Chandran Remi and Libor Jansky

Environment and Sustainable Development Programme, United Nations University, 53-70, Jingumae 5-chome, Shibuya-ku, Tokyo 150-8925, Japan.

Email:chandran@hq.unu.edu, jansky@hq.unu.edu

Abstract

The concept of water management is getting increasingly important as the situation of water scarcity is growing at an alarming rate. The management of natural and artificial reservoirs are important for sustainable development. It is estimated that 0.5 - 1% of the world reservoir volume is lost from sedimentation annually. The World Commission on Dams knowledge base indicates that, while sedimentation potentially undermines the performances of a large dam project, the conditions, and therefore, the frequency of occurrence of this phenomenon are project and site specific. In natural reservoirs, sedimentation problems often arise due to soil erosion processes in the catchment and thus, its management requires a more comprehensive approach.

Often the problems on sedimentation lead to water shortage and non utilization of reservoirs and leading to depletion of the resource base of a country. This, then leads to scarcity of water which thereby affects the livelihood of common man and in certain cases, leading to conflicts.

The objective of this study is to look into factors leading to sedimentation and to bring together suggestions for management of natural and artificial reservoirs from Africa, Asia and Europe thereby integrating a code structure of good practices for sustainable development. The cultural and economic aspect would also be looked into apart from the national and international policies on water applicable to the region.

The study would be focusing on management practices from Danube basin in Europe, the Narmada river basin in India and the Abaya lake basin in Ethiopia.

Key words*:* sedimentation, reservoirs, Lake Abaya, Danube, Narmada river, world commission on dams.

Introduction

Sediments originate in the catchment through erosion processes and are transported in river systems in the direction of the coast, with oceans being the final sink. As such land use, hydrology, geology and topography determine erosion and transport processes. In the river system temporary deposition can take place. Important in this respect are floodplains and lakes. In many regulated rivers, sediments are trapped behind dams and reduce sediment supply downstream.

Though reasons for sedimentation can be anthropogenic, there is also a strong reason to prove that climatic variations can be one of the factors that may cause an increase in sedimentation in rivers and reservoirs. For example, in temperate areas, the sedimentation of a reservoir is usually a slow process. A reviewed onsedimentation rates at 19 reservoirs in Central Europe showed that their storage capacity (which ranged between 120 and 183,000 acre feet) was depleted by sedimentation at an average rate of 0.51 percent per annum. (Glymph, 1973).

Another study on the sedimentation rates of small, medium and large reservoirs in the United States (Dendy *et al.,* 1973), found that:

- The rate of sedimentation in 1,105 reservoirs with a capacity of less than 10 acre feet was approximately 3.5 percent a year.
- In the case of medium-sized reservoirs (with a storage capacity of more than 100 acre feet) the annual storage loss was 2.7 percent per annum and the median rate of sedimentation was 1.5 percent.
- For reservoirs with a storage capacity of more than a million acre feet, the rate of sedimentation was only 0.16 percent per annum, with the mean rate coming out at 0.11 percent a year (Dendy *et al.,* 1973).

Whereas, the sedimentation rates in the tropics are much higher and can be explained by the devastating effect on which deforestation has had on tropical soils. In those areas where forest cover has been depleted, however, the rate of soil erosion increases dramatically: the organically poor soils of the tropics are particularly vulnerable to erosion and although the monsoons only last for a short time, they can quickly wash away the soils from deforested slopes.

Given the present rate of deforestation in the tropics (10 hectares of rainforest are lost throughout the world every minute of the day) it is hardly surprising that rivers in the region carry enormous quantities of silt. Indeed, in many areas, the increased sediment load of rivers is clearly visible to the naked eye. In this study we would be looking at three case studies and the management practices for sediment management in Danube Basin in Europe, the Abaya lake basin in Ethiopia and Narmada river basin in India.

Water reservoirs in the Danube basin

Damming has become a practical necessity and has provided huge benefits to agriculture, industry and urban development. The report of the World Commission on Dams (WCD, 2000) has highlighted the scale of human intervention of ecosystems by the construction of large dams. Dams, inter-basin transfers and water withdrawals for irrigation have fragmented over 60% of the world's rivers and changed the sediment load of rivers to the coastal sea.

In Western and Northern Europe reservoirs can be found in many catchments depending on their main purpose: hydroelectric reservoirs are common in Scandinavia and in the Alpine range from France to Slovenia, as well as in medium-high mountains (Tatra, Carpathians). Reservoirs of various sizes have been constructed in the Vistula, Elbe, Seine and Danube catchments.

Most of the reservoirs in Danube are man-made sediment traps in which more than 90% of the sediment transport of an incoming river can be stored when the residence time of the water exceeds two months. For the impact of damming on the global water and sediment flux, quantitative estimates have recently been made. The large reservoirs in the Danube basin intercept more than 40% of water discharge and approximately 70% of this discharge maintains a sediment trapping efficiency of more than 50% (Sednet, 2004). It is estimated that about 25 to 30 % of the sediment flux to the coastal sea is trapped behind dams (Sednet, 2004). One of the positive environmental effects is the trapping of contaminants associated with sediments and in this way protecting downstream areas. For example, in the territory of Slovakia, Danube basin area is around 48,950 km^2 and is divided into 10 sub-basins (9 belonging to the Black Sea basin and one to the Baltic sea basin). The most serious problem in these reservoirs are the quality of water for most of these reservoirs do not meet the standards required for sustainable quality use of water.

Management of sediments in the Danube - Danube River Protection Convention (DRPC)

Under the Danube River Protection Convention (DRPC) and within its organisation, the MLIM/EG (Monitoring, Laboratory and Information Management Expert Group) is responsible for "operating" the Trans National Monitoring Network (TNMN) for water quality in the Danube River Basin. One of its tasks is to set up programmes to improve laboratory analytical quality assurance. It also facilitates the preparation and exchange of (in-stream) water quality and quantity data among the contracting parties (Transnational Monitoring Network, 2001). The MLIM was set up in December 1992 as a Sub-Group of a former Task Force of the EPDRB with the aim to strengthen national and international capacity to provide reliable information on surface water flows and the quality of waters in the Danube river basin, to improve the comparability of sampling techniques and laboratory analysis, and to develop compatible information management systems for the exchange of information at the international level. Three MLIM's working groups were set up for Monitoring, Laboratory Management, Information Management.

An important task of the Monitoring group has been to develop a Trans National Monitoring Network (TNMN) which also enhances the Bucharest Declaration monitoring network. In order to provide data for TNMN, a National Reference Laboratory Network (NRLN), with equal technical (equipment) and methodological capabilities (practices for sampling and analysing) in each participating country has been created under the working group of Laboratory Management. A National Information Centre Network (NICN) was created to exchange data in a common format between countries and international bodies. (Transnational Monitoring Network, 2001).

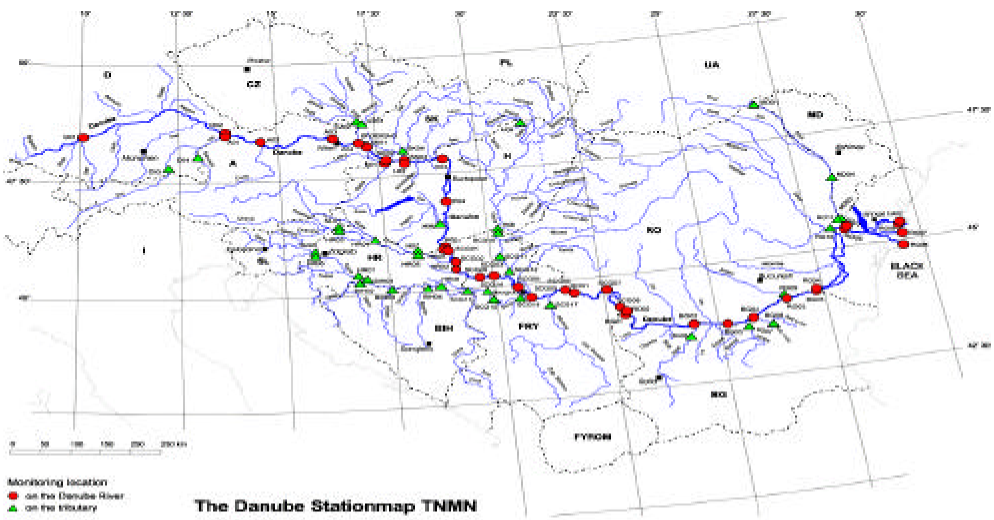

Figure 1. Sediment Monitoring locations in the Danube (Transnational Monitoring Network, 2001).

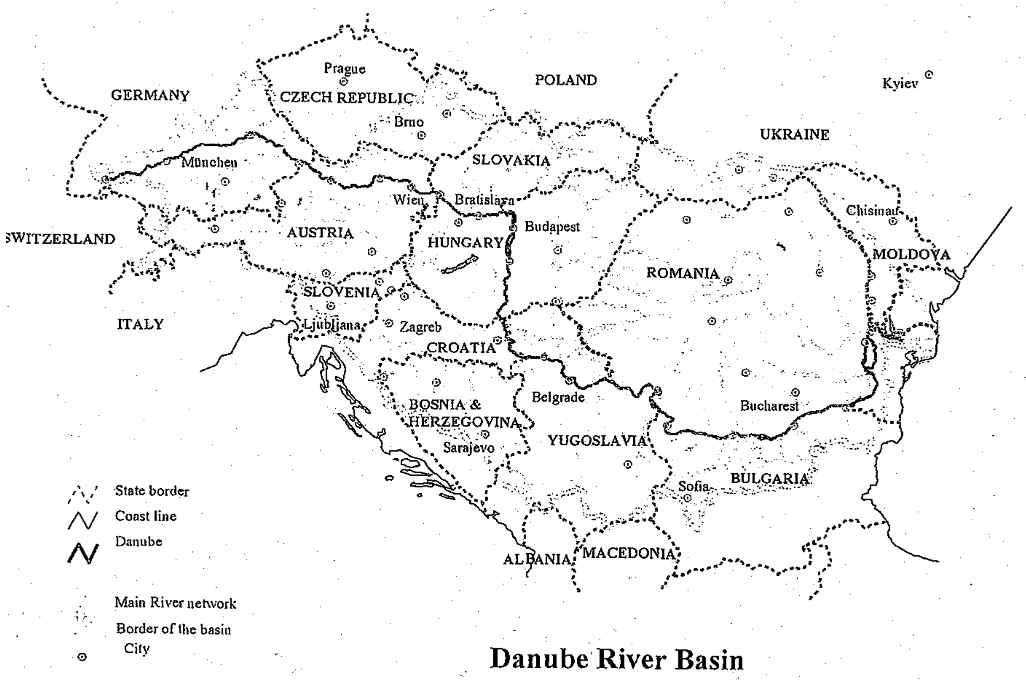

Figure 2. Hydrological catchments of the Danube (Source; International commission for the Protection of Danube River, 2005).

The Danube Water Quality Transnational Monitoring Network (DWQ-TNMN) is officially operating, starting June 9, 1999. The fully participating countries at present are Germany, Austria, Czech Republic, Slovak Republic, Hungary, Slovenia, Croatia, Romania and Bulgaria. Developments are under way to fully include Moldova, Ukraine, and Bosnia and Herzegovina.

The aim of the DWQ-TNMN is to enable the International Commission for the Protection of the Danube River as well as national authorities from the Danubian countries to manage and improve the quality of water resources in the whole region (Fig.2). Such management is based on the data and information obtained from the monitoring activities carried out.

The Transnational Monitoring Network has sampling and measuring locations in all countries starting from the river source in Germany and down to the three mouths in the Danube Delta where the river discharges into the Black Sea (Fig.1). There are 61 sampling stations in the present structure of the TNMN which were selected from the national monitoring networks. The selection was made based on criteria and objectives agreed between the countries. Other monitoring parameters like the list of determinants, sampling frequencies, analytical procedures were also agreed by the countries on basis of the advice given by the Expert Groups and Expert Subgroups.

Sinc 1996 data is provided on an annual basis by the Danube countries to the Central Information Point (CIP) where TNMN data are assembled in a well defined structure using rules of reference integrity.

Sedimentation of reservoirs in Danube Basin – A case study from Slovakia

Water-management operation of reservoirs and dams are related with many issues and consequences, which are to be maintained at the minimum sustainable level. The common problems noticed in Slovakian reservoirs are, sedimentation, wave abrasion, and changed hydrological regime in the water course downstream of the reservoir. the problems caused by sedimentation are rather difficult in case of reservoirs with low regulative capacity, i.e. Krpelany, Hricov, Nosice, Drahovce on the Váh river. In case of dredging these sediments, additional problem arose regarding their classification as “waste material”, requiring special storage. The most extensive sedimentation occurs at the reservoir Velká Domaša with average specific disposal of 990 m of material from the catchment.

Another problem noticed was of wave abrasion occurring at large reservoirs with extensive flooded areas, where certain wind conditions creates intensive wave regime. Wave motion induces wave abrasion especially where the reservoir banks are constructed as sloppy structures of clay detritus, or easily erodable rocks (slates, claystones, etc.). These factors were experienced at 6 largest reservoirs: Orava, Velká Domasa Liptovská Mara, Starina, Nová Bystrica, Vihorlat.

There are also many small reservoirs in Slovakia serving useful functions and improving the total water balance. Official standards define small reservoirs as basins with a capacity of not more than 2 million m^3 of water, a maximum depth of 9 m and a hundred year peak discharge no greater than $60 m^3/s$. Nationally, there are around 350 such small water reservoirs in Slovakia with a total surface area of 1910 ha and a design capacity of 45 million m^3 (Haigh et al 2004).

Sedimentation studies in this area have revealed that the amount of sedimentation ranges from 4.8% to 83.6 % of the total storage capacity which directly decreases the storage volume by 0.32- 9.3% with an anticipated design life of 100 years. Hence, maintenance clearance would be required on an average of every 15 years (Haigh *et al*., 2004)

Sedimentation studies in Lake Abaya

Lake Abaya is the largest lake of the main Ethiopian Rift Valley (Figure 3). The area in which the lake lies is mainly volcanic in origin while the waters of the lake are permanently turbid due to a heavy colloidal suspension of ferric oxide. One interesting factor in the sedimentation of lake Abaya is that, prior to 1970's, the major reason for sedimentation was attributed to climatic factors. However, since 1970's it is noted that changes in cultivation methods, has also caused a dramatic increase in sediment yield of Lake Abaya tributaries, thus, influencing basin bathymetry and volume (Schütt et al 2005) Because of its shallow depth (max. depth of 26 m) the lake level of Lake Abaya reacts sensitively to changes of water and sediment input (Schütt *et al*, 2005).

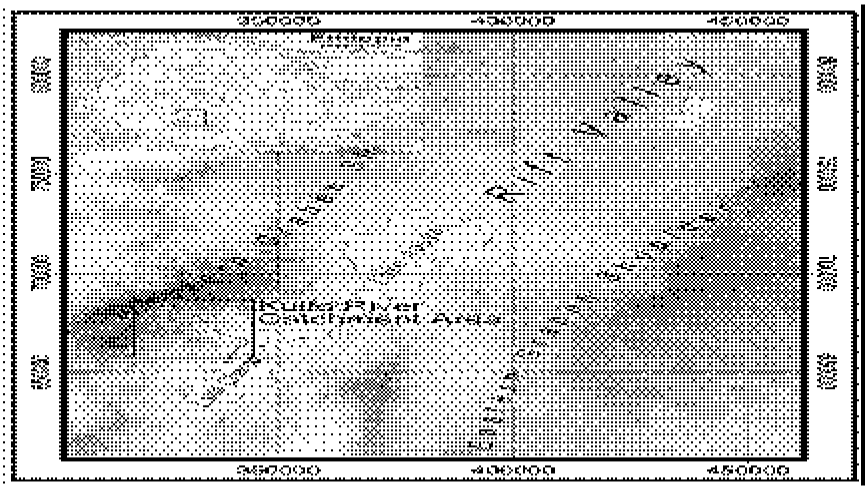

Figure 3. Topographical map of the Lake Abaya-Lake Chamo basin. (Source: Schütt et al, 2005).

Earlier investigations on sediment management studies have proved a distinct differentiation between sediments incoming from 2 different tributaries discharging directly into the bottleneck of central Lake Abaya, one with a non anthropogenic influence and the other tributary with heavy human interaction. This clearly helped in identifying the results of sedimentation due to man made factors and climatic interactions. Furthermore, the characters of the lacustrine sediments indicate an existing interchange between the waterbodies of the northern and the southern Lake Abaya sub basins (Blumberg & Schütt, 2005).

Causes for sedimentation in Lake Abaya

There are two major reasons associated with sedimentation of lake Abaya,

- Due to climatic and paleogenic changes
- Due to human interference

Due to climatic and paleogenic changes

High concentrations of quartz, silicates and iron-oxides point out that lake-floor deposits of lake Abaya bottleneck are predominantly allochthonous (Jones & Bowser, 1978). According to the high erodibility of soils in the catchment area Lake Abaya tributaries bring in high quantities of suspended load (*Krause et al.,* 2004, Schütt & Thiemann 2004).Quartz and feldspar are primary formed in magmatic rocks. In the Lake Abaya drainage basin they occur area wide as well in the Tertiary volcanic rocks as in the underlying Palaeozoic sediments (Mohr, 1961). Correspondingly, both minerals were also recorded area-wide in the lake-floor sediments from Lake Abaya bottleneck. They have a high alteration resistance in the given limnic environment (Hakanson & Jansson, 1983). Distinct differences in the distribution Hematite is a mineral formed by pedogenic processes under tropical and subtropical environmental conditions (Schwertmann & Taylor, 1989). It occurs prevalently as a weathering cover, especially around clay-minerals (Berner, 1971). Thus, due to the ubiquity of clay minerals in the Lake Abaya bottleneck, spatial pattern of hematite is linked to the occurrence of clay minerals.

As for the drainage basin of Lake Abaya calcite deposits do not occur (Mohr, 1961), calcites detected in Lake Abaya's lacustrine sediments have to be of autochthonous origin. Calcium derives from weathering of calcium-bearing feldspars (hydrolysis) while carbon is made available by decomposition of organics (Meyers & Ishiwatar, 1993). High water temperature as well as high pH values cause a decrease of the calcium carbonate's solubility product and afford its precipitation (Sonnenfeld, 1984). Thus, in shallow water areas like in the coves south of the Shope and Gelana River deltas calcite precipitation is improved by the local environmental conditions.

Due to human interference

In Lake Abaya typical topset beds are most recent deposits and correspond to soil-sediments and their mobilization and final deposition is due to soil erosion processes. In the Shope River drainage basin high accessibility causes a high population density – especially along the major traffic routes and, thus, high land use intensity. Correspondingly, expected soil erodibility in the Shope River drainage basin is high and results in high deposition-rates of topset beds. In contrast, accessibility of the Gelana River catchment area is poor as main roads are lacking. In consequence, population density and land use intensity are low, thus, also expected soil erodibility remains relatively low (Lal, 1990) resulting in relatively small input of suspended load into Lake Abaya. Accordingly, it can be confirmed that the volume of topset beds is controlled by the input of suspended load and indicates intensity of soil erosion processes in the hinterland.

Sediment management practices in Lake Abaya

One of the major problem regarding the management of lake basin in developing countries is that, they do not have a strong sediment management system. In the case of Ethiopia, there is a practice of strict limitation of grazing and prohibition of tree felling in eroding areas, which appears to be meeting with some success.

The necessity of a more rational distribution of land use is also evident, but several factors hamper its realization. A physical constraint on changes in land management is the pressure to maintain, or better increase, total production levels, to support growing population and livelihood expectancies. A social and political constraint is the present land tenure regime, in which land belongs to the Kebelé; these institutions have proved a very effective framework for land management in their interior, but have caused a significant rigidity towards any shift involving more Kebelé. As the Kebelé are usually quite small, land reallocation on any larger scale is practically impossible.

Narmada river basin sedimentation concepts

The South Asian region accommodates a population of 1.4 billion people, which is more than one fifth of the global population. The Ganga, Brahmaputra, Indus, Narmada, Krishna, Godavari, Padma, Irrawadi and Salween are the major rivers in the region. Most of these rivers are well known for high sediment concentration during the months of summer monsoon. In particular, the rivers originating from the Himalayas are known as some of the highest sediment laden rivers of the world. These large rivers, along with other medium and small rivers of the region, transport a significant amount of sediment in addition to other riverine materials towards oceans. The region contributes almost 15% to 20% of the global sediment flux towards the oceans (Milliman & Meade, 1983; Milliman & Syvitski, 1992; UNEP, 1995). The sediment received by the oceans at river mouths has created some of the largest deltas of the world such as the Ganga-Brahmaputra delta (50,000 km^2) and the delta formed by the Irrawadi River in Myanmar(30,000 km^2).

Narmada is one the major rivers draining towards Arabian Sea. The region consists of volcanic areas and ancient rocks with less than two percent recent deposit areas. Biksham and Subramanian (1988) and Chakrapani and indicate that the geological formation within the region is the major controlling factor of sediment transport. Archaean rocks occur on either side of the Narmada river and include granites and gneisses and several outcrops of Bijawar rocks.

From Table 1 it is well understood that high soil erosion and tectonic factors have lead to heavy sedimentation in the Narmada river.

Sediment management in Narmada

Information on sediment in the region is far less than adequate and significant uncertainties lie in the proper understanding of its behavior. Educational and research activities must be promoted in the field of soil erosion, soil conservation, sediment transport, and river morphology. Similarly, methods for monitoring sediment transport must be standardized for obtaining reliable and standard data for the assessment of sediment flux. Major sources of sediment data were the publications on reservoir sedimentation. However, there was inadequate information about the trapping efficiency of the reservoirs. Besides, the data on reservoir sedimentation not provide other essential information such as the hydrological and geographical data. Inclusion of such information would provide a lot of scope for the assessment of sediment transport characteristics.

Table 1. Various parameter measured at the Narmada Basin.

Physical parameters	Specific run-off mm/km/yr	Rainfall (mm)	Length (km)	Molar Ratio	
	531.5	1000	1312	0.71	
Chemical parameters	**Fe**	**Mn**	**Cu**	**Ni**	**Zn**
	7.6	1125	128	70	125
Denudation rates	Chemical Denudation rate	Total denudation rate			
	12.11	19.1			

Source: Alagarsamy and Zhang (2005).

Conclusions

This paper describes the various trends in sedimentation in lakes, reservoirs and rivers and also explains reasons for sedimentation. It is also understood that, climatic and human interactions have a big role in building up sediments in basins and catchments.

It would also be ideal to have strong policy measures to control sedimentation due to human factors. Another important feature we notice by comparing the regional studies (Danube, Lake Abaya and Narmada) is that, the developed nations have a more stricter rules and measures for monitoring and control of sediments, whereas, in developing countries, though sedimentations studies are carried out intensively, management practices are quite weak and do not answer critical problems.

The size of the water shed and basins also plays a decicive role in sediment management. We see from examples from the Danube river, the size of the network catchments is quite big and hence the sedimentation load into the river is also high though most of it get deposited into the artificial reservoirs in the catchment. In Narmada river, sediment deposits are carried from the tectonic regions in Himalayas and is brought down to the river thus causing heavy sedimentation down stream. In the case of lake Abaya, the sedimentation is caused mainly by erosion from its tributaries.

Sedimentation is also a natural process and we can reduce the sedimentation rates into the rivers and reservoirs by managing its catchments in a sustainable manner. Policy initiatives taken to afforest catchments has always proved right in controlling erosion and thus managing sediments.

References

Alagarsamy,R. and Zhang J., 2005. Comparative studies on trace metal geochemistry in Indian and Chinese rivers. *Current science*. Vol. 89, No.2, 299-309

Berner, R.A., 1971. *Principles of Chemical Sedimentology.* McGraw-Hill Book Company, New York.

Biksham, G., & Subramanian, V., 1988. Sediment transport of the Godavari basin and its controlling factors. *Journal of Hydrology.* 101, 275-290.

Chakrapani, G. J., & Subramanian, V., 1990. Factors controlling sediment discharge in the Mahanadi River basin, India. *Journal of Hydrology*. 117, 169-185.

Glymph, M. L., 1973. "Summary: Sedimentation of Reservoirs". *In.* William C. Ackermann et al., eds. Man Made Lakes, their Problems and Environmental Effects. *American Geophysical Unio.* Washington DC, p.343.

Haigh, J. M., L. Janský & J. Hellin, 2004. Headwater Deforestation: A Challenge for Environmental Management. *Global Environmental Change, Human and Policy Dimension*. 14(1):51-62.

Håkanson, L. & M. Jansson, 1983. P*rinciples of Lake Sedimentology*. Springer, Berlin, Heidelberg, New York, Tokyo.

Jones, B. F. & C.J. Bowser, 1978. The Mineralogy and Related Chemistry of Lake Sediments. *In*. Lerman, A., ed. *Lakes Chemistry Geology Physics*. Springer; New York, Heidelberg, Berlin. p. 179-235.

Krause, J., Schütt, B. & S. Thiemann, 2004. Hare River catchment – landscape character of a drainage basin in the southern Ethiopian Rift Valley. *In*. B. Wenclawiak & S. Wilnewski, eds. *Sedimentary Studies in Tropics and Subtropics.* Weiterbildung in Siegen, 14, 16-40.

Lal, R., 1990. *Soil erosion in the Tropics.* McGraw-Hill; New York.

Meyers, P.A. & R. Ishiwatari, 1993. The early diagenesis of organic matter in lacustrine sediments. -In: Engel, M.H. & Macko, S.A. (eds): Organic geochemistry - Priciples and applications. -Plenum Press, New York, p. 185-209.

Milliman, J. D., & Mead, R. H., 1983. World-wide delivery of river sediment to the oceans. *The Journal of Geology,* 91(1), 1-21.

Milliman, J. D., & Syvitski, J. P. M., 1992. Geomorphic/tectonic control of sediment discharge to the ocean: the importance of small mountainous rivers. *The Journal of Geology*, 100, 525-544.

Mohr, P.A., 1961. The geology of Ethiopia. Addis Ababa; Univ. College Addis Ababa Press.

Schütt, B., S. Thiemann & B. Wenclawiak, 2005. Deposition of modern fluvio-lacustrine sediments in Lake Abaya, South Ethiopia - A case study from the delta areas of Bilate River and Gidabo River, northern basin. Z.f.Geomorph. Suppl.-Bd./, 131-151.

Schwertmann, U. & R.M. Taylor, 1989. Iron oxides. In. Dixon, J.B. & Weed, S.B., eds. *Minerals in soil environments*. SSSA Book Ser., 1.-SSSA; Madison, WI.

Sednet. 2004. Contaminated Sediments in European River Basins - SedNet December 29th, 2004 http://www.sednet.org/materiale/SedNet_booklet_final_version.pdf

Sonnenfeld, P., 1984. *Brines and evaporites*. Academic Press; Orlando.

Transnational Monitoring Network, 2001. http://www.icpdr.org/pls/danubis/docs/folder/HOME/ICPDR/ICPDR_DOC_CENTRE/ANNUAL_PUBLICATIONS/W_Q_YEARBOOKS/TNMN2001/tnmn2001_4-1.html

United Nations Environment Program Global Environment Monitoring System/Water.

(UNEP/GEMS). 1995. *Water Quality of World River Basins*. Nairobi, Kenya: United Nations Environment Programme.

WHO/UNEP. 1997. *Water Pollution Control - A Guide to the Use of Water Quality ManagementPrinciples* http://www.who.int/docstore/water_sanitation_health/wpcontrol/begin.htm#Contents

World Commission on Dams, 2000. *Dams and Development: A New Frame Work for Decision Making*. pp (16-17) Earthscan Publications Limited, London

Coleman, J. M., 1969. Brahmaputra River: Channel processes and sedimentation. *Sedimentary Geology*. 3, 129-239.

Dendy, F.E; Champin W. A and R. B. Wilson, 1973. *"Reservoir Sedimentation Surveys in the United States"*. Willam C. Ackermann *et al*., eds. p.353.

Spatial-temporal variability of phytoplankton abundance and species composition in Lake Victoria, Kenya: implication for water quality management

Gikuma-Njuru Peter[1,] , Mwirigi, Peter [1], Okungu, John [1], Hecky, Robert [2] and Abuodha, Joseph[3],

[1]LVEMP, Water Quality Management Component, P.O. Box 1922 Kisumu, Kenya.
Email: njurupg@yahoo.com
[2]Biology Department, University of Waterloo, 200 University Avenue West Waterloo, Ontario, Canada. N2L 3G1
[3]Maseno University, Kenya.

Abstract

Study of phytoplankton abundance and species composition in Lake Victoria, Kenya showed a marked difference between the main lake and the semi-closed Nyanza (Winam) Gulf. Phytoplankton biomass and species composition showed both spatial and seasonal variation. In the main lake the highest chlorophyll concentration was during the short stratification season (August to September) but in the gulf it was after the short and long rainy seasons. Deep mixing depth (aprox. 30 m) in the main lake imposes light limitation to algal growth leading to the observed lower chlorophyll in the main lake, despite the higher soluble reactive phosphorus (SRP) in the main lake compared to the gulf. Blue green algae were the most abundant algal species, contributing between 45 and 65% of total abundance. *Cylindriospermopsis africana,* a nitrogen fixing and potentially highly toxic species was present in the gulf during the dry season but was absent during the wet season and in the main lake the nitrogen fixing *Anabaena sporoides* was present in high numbers during mixing period, which can be attributed to low availability of dissolved inorganic nitrogen during lake mixing. Rusinga channel, the transition area between the Nyanza Gulf and the main lake, had high diatom abundance and showed a different phytoplankton species composition compared to the other lake zones. The current dominance of cyanobacteria (blue-green algae) in the lake, and especially bloom forming and potentially toxic species calls for urgent management of pollution loading into the lake in order to stem further degradation of water quality and help in the restoration of Lake Ecosystem.

Key words*:* Nyanza Gulf, Phytoplankton, Water quality

Introduction

Phytoplankton productivity and species composition in a water body are mainly influenced by nutrient availability (mainly phosphorus, nitrogen and silica (for diatoms)), environmental factors such as temperature and light and the morphology of the water body (Kalff, 2002). Availability of light for phytoplankton growth in a lake is determined by water transparency, mixing depth and incident light intensity. Nutrients are normally present in very variable quantities from one water body to another and even within the same system may vary in space (especially during stratification) or seasonally according to rainfall, temperature, wind and the nature of the phytoplankton community (Maitland, 1990).

In lake Victoria, the influence of nutrient and light availability on phytoplankton abundance and species composition has be been reported and associated with the changes in phytoplankton assemblage that have occurred in the last 4 decades (Mugidde *et al.,* 2003). The shift from dominance by diatoms to that of blue green in the open waters of Lake Victoria has been associated with reduction of silica concentration in the water column due to increased burial of diatoms in the sediments, associated with increased productivity in the lake (Verschuren *et al.,* 2002). Increased nutrient loading into the lake, associated with increased human population in the lake catchment and resultant land degradation has been blamed for increased eutrophication and changes in phytoplankton productivity and assemblage in Lake Victoria (Botsma & Hecky, 1993; Hecky, 1993). The dominance of blue green algae and associated algal blooms in the lake (Ochumba & Kibaara, 1989; Krienitz *et al.,,* 2002) will continue to affect the lake water quality since some blue green species have been found to produce cyanobacterial toxins (Krienitz *et al.,* 2002).

Materials and methods

Study area

The Kenyan waters of Lake Victoria lie just south of the equator between 0^0 6' S/0^0 32'S and 34^0 13'E/ 34^0 52'E at an altitude of 1134 m asl and cover an area of 3600 Km^2 (approximately 6% of the whole lake) of which 1400 Km^2 comprises the Nyanza (Winam) Gulf (Figure 1). It has a catchment area of 3600 Km^2, which is drained, by 5 major rivers (Nzoia, Kuja, Nyando, Yala and Sondu) through which it contributes approximately 30% of total riverine inflow into Lake Victoria. The catchment falls in some of the most agriculturally productive areas in the country with extensive use of agro-chemicals and has several major urban centers including Kisumu.

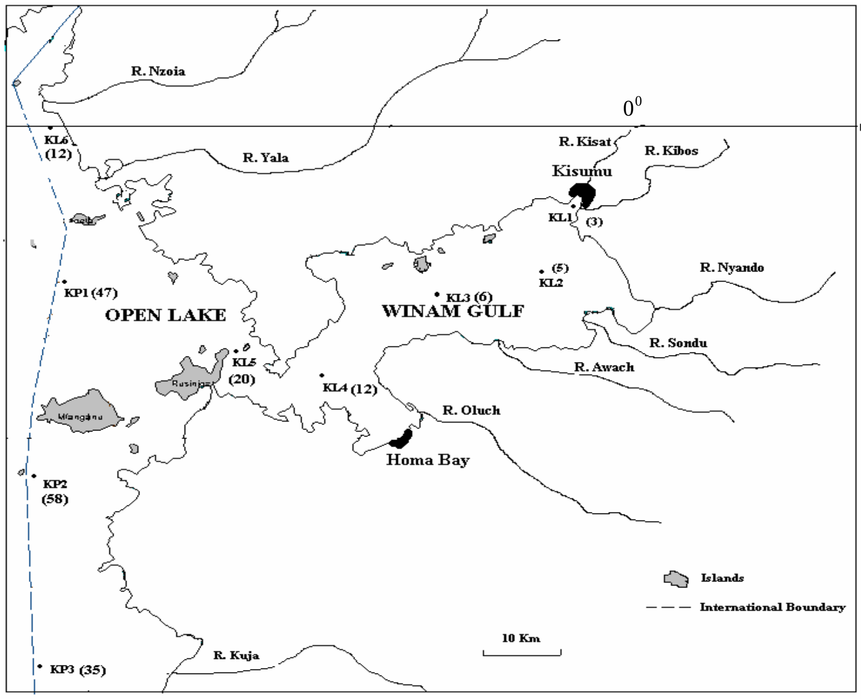

Figure 1. Location of the sampling stations in the Gulf (KL1-5) and in the open lake (KP1-3; KL6). Values in the parenthesis indicate the station depth.

Samples for analysis of phytoplankton biomass (chlorophyll) and species composition and nutrients were collected between 2000 and 2004 in stations within the main Lake Victoria and the Nyanza Gulf. The stations are part of the lake-wide LVEMP water quality monitoring network, chosen to fall within spatial transects covering both littoral and pelagic stations. Pelagic stations are the deep (>20 m) and offshore (>5 km from shoreline) whereas the littoral stations are those less than 20m deep and less than 5km from the shoreline.

Sampling and laboratory analysis

Water samples were collected at different depths using the 2-litre Van don sampler for the analysis of phytoplankton biomass (chlorophyll) and species composition and nutrients. Phytoplankton samples were fixed in the field with Lugol's iodine solution and analysis was done in the laboratory using standard inverted microscope techniques. Chlorophyll *a* was extracted using 95% ethanol and analyzed according to Wetzel and Likens (1990). Treatment and analysis of total and dissolved nutrients (Total phosphorus (TP), Orthophosphate (PO_4-P), Total nitrogen (TN), Nitrate (NO_3-N), Ammonium nitrogen (NH_4-N) and Silica (SiO_2-Si)) were carried out using spectrophotometric methods as described in APHA (1995). In the field, light penetration was estimated with a 25 cm diameter white Secchi disc and a Conductivity–Temperature–Depth (CTD) Probe with additional sensors (Hydrolab Survey 4a) was used to measure temperature, pH, dissolved oxygen, turbidity and conductivity at different depths down the water column.

Results

Observations in the field showed spatial and temporal variability of occurrence of algal blooms in the Nyanza Gulf and in the main Lake Victoria with heavy blooms observed in the eastern and southeastern zones near Kisumu City and rivers mouths in the gulf and near the mouths of major rivers in the main lake (Nzoia, Yala and Kuja). Heavy algal blooms in the gulf were found to occur after the rainy season (May to July), after the settling of inorganic turbidity associated with the rainy season. Physical observations and satellite imagery has shown the area north of Rusinga Channel, in the main lake, to have occasional algal blooms.

Phytoplankton biomass (chlorophyll a) showed a reducing trend along the gulf into the main lake with a highest average value of 21.1µg/l (KL2) and a lowest value of 6.9µg/l (KP3). KL6, a littoral station in the main lake, had an average value of 18.18 (Figure 2a). A comparison of 4 seasons, Jan-March (dry and long stratification); April-July (cool and mixing season); August-September (short stratification) and November-December (short mixing and onset of stratification) showed the maximum biomass in the gulf and the in the main lake to be during August-October (29.6µg/l and 15.8µg/l respectively) but in KL6, the littoral station in the main lake, it was during November-December (31.4 µg/l) (Figure 2b).

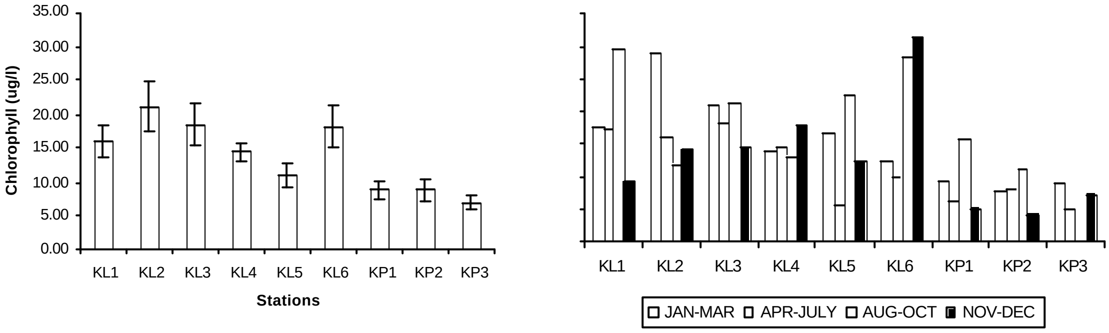

Figure 2. Spatial and seasonal variation (a and b respectively) of chlorophyll in the lake.

Species composition and abundance

Phytoplankton species composition varied within stations and between seasons. In both littoral and pelagic areas, cyanobacteria was the most abundant, contributing between 45 and 65% of the total phytoplankton abundance and diatoms contributed between 20 and 40% of total abundance, with the main lake having higher relative diatom abundance than the gulf (Figure 3 a and b). *Cyndrospermopsis africana* was present both in the littoral and pelagic stations but occurred in higher densities in March 2004 compared to other months sampled. Among the diatoms, *Nitzschia acicularis* was dominant in the main lake, whereas in the gulf *Aulacoseira nyassensis* dominated. *Anabaena sp* was more common in the main lake and *Microcystis aeruginosa* was more common in the gulf. Phytoplankton abundance was higher in the gulf than in the main lake and followed a similar spatial pattern with phytoplankton biomass (chlorophyll) (Figure 2a).

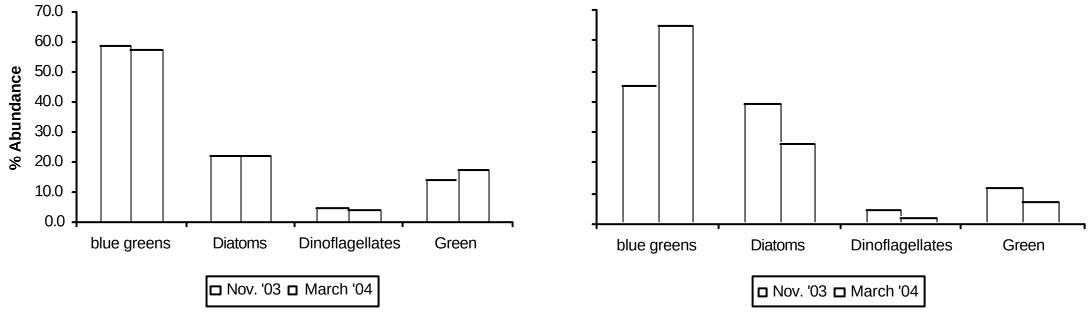

Figure 3. Relative abundance of different phytoplankton groups in the (a) gulf stations and (b) pelagic stations.

Productivity

Primary productivity rate in the lake ranged between 217 and 646 mg O_2 m^{-3} h^{-1} and showed seasonal variation within and between stations (Figure 4a).Photosynthetic efficiency as indicated by average productivity per unit biomass was low in the gulf station KL3 and in the main lake station (KP1) and ranged between 11 and 53 mg O_2 mg Chll^{-1} h^{-1} (Figure 4b).

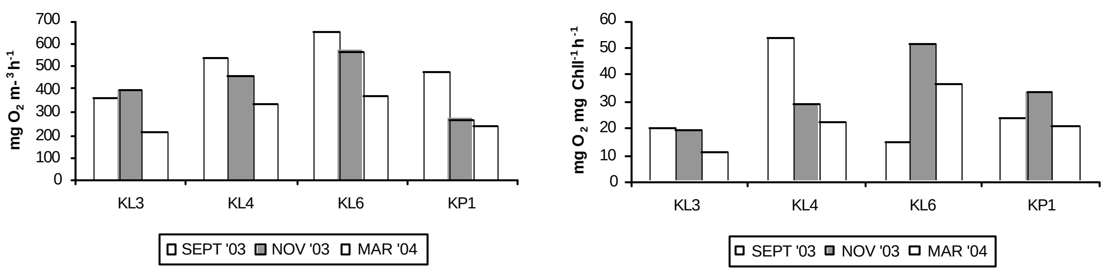

Figure 4. Primary productivity rate (a) and photosynthetic efficiency (b) in littoral and pelagic stations during September 2003, November 2003 and Mach 2004.

Nutrient concentrations varied spatially between the gulf and the main lake. PO_4-P concentration ranged between 19.4 and 56.5 µg/l and was higher in the main lake than in the gulf but SiO_2-Si was higher in the gulf than in the main lake with values ranging between 0.6 and 4.97 mg/l. The TN:TP ratio (molar) in the gulf stations was more than 50 whereas in main lake stations it was between 40 and 50. Water transparency (measured as Secchi depth) reduced from and an average value of 2m in the main lake to an average value of 0.5 m and showed a strong negative logarithmic relationship with the chlorophyll.

Discussion

Talling (1966) reported phytoplankton biomass (chlorophyll) values of 1.2-5.5µg/l in the pelagic waters which is less than the values measured during the present study (Figure 2) in the pelagic stations, confirming the reported changes in phytoplankton biomass in the past four decades (Kling *et al.,* 2001; Hecky, 1993). The dominance of cyanobecteria in lake, during the present study, is consistent with observations made by other scientists (e.g. Ochumba & Kibaara, 1989; Lung'aiya *et al.,* 2000) and in line with the reported changes of phytoplankton assemblage from diatom dominated to blue green dominated population (Hecky, 1993; Lehman & Branstrator, 1994; Kling *et al.*, 2001).

In the main lake, where the mixing depth is greater than the photic zone (Gikuma-Njuru & Hecky, 2005), phytoplankton growth is normally light limited (Mugidde, 2003), which may account for low algal biomass in this part of the lake despite the high dissolved nutrient concentration. In the gulf, phytoplankton may be light limited due to self shading and high inorganic turbidity associated with riverine inflows (Gikuma-Njuru & Hecky, 2005). As the water depth increases from approximately 2m in the eastern part of the gulf to more than 40 m in the main lake, the resultant changing light and nutrient regimes will result in spatial variability of phytoplankton assemblage since different species will respond differently to this changing environment.

The ratio between N and P can be used to indicate the phytoplankton nutrient status in a water body. According to Guildford and Hecky (2000), for TN:TP ratio <20 phytoplankton is normally nitrogen limited and in cases where the ratio is >50 phytoplankton will most likely be phosphorus limited. The TN:TP ratio in the lake shows that the Nyanza Gulf is consistently phosphorus limited whereas the main lake can be either nitrogen or phosphorus depending on the prevailing physico-chemical and nutrient status. Under nitrogen limited conditions, phytoplankton species with the ability to fix inorganic dinitrogen from the atmosphere, normally has an advantage over the other species and therefore dominate (Guildford et al., 2003; Mugidde *et al.,* 2003). This is in line with current status where of the nitrogen fixing *Anabaena sp* occur in the main lake and *Microcystis sp*, non nitrogen fixing blue green, is more common in the gulf. Since the gulf as a whole is P limited, continued P input to this semi-closed part of the lake will result in increased algal blooms and increased eutrophication and therefore negatively affecting the water quality.

The dominance of cyanobacteria in the lake present water quality challenges, as this algal group is bloom forming and has species which are known to produce phycotoxins, which can both cause fish kills and compromise drinking water quality (Hummert *et al.,* 2001; Kling *et al.,* 2001; Krienitz *et al.,* 2002). The occurrence, in mid 2004, of a large algal bloom in Kisumu Bay, which lead to the closure of the Prisons Department water treatment works for a number of days, is a good example of negative impact of algal blooms on water quality. Although this occurrence was thought to be as a result of reduced lake levels, due to long dry spell at the time, continued nutrient enrichment of the lake will probably increase the frequency and severity of such occurrences especially in the littoral hotspot areas (LVEMP, 2002).

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References

American Public Health Association (APHA), 1995. *Standard Methods for Analysis of Water and Wastewater, 18th edition*. Port City Press, Baltimore, MD.

Bootsma, H.A., Hecky, R.E. 1993, "Conservation of African Great Lakes: A limnological perspective", *Conservation Biology,* vol. 7, no. 3, pp. 644-656.

Gikuma-Njuru P. and Hecky R.E. 2005. Nutrient concentrations in Nyanza Gulf, Lake Victoria, Kenya: light limits algal demand and abundance. *Hydrobiologia*. 534: 131-140

Guildford, S. J., and R. E. Hecky, 2000. Total Nitrogen, total phosphorus and nutrient limitation in lakes and oceans. *Limnology and Oceanography* 6: 1213-1223

Guildford S. J., R. E. Hecky, W. E. Taylor, R. Mugidde, and H. A. Bootsma, 2003. Nutrient enrichment experiments in tropical Great Lakes Malawi/Nyasa and Victoria. *Journal of International Great Lakes Research 29* (supplement 2): 89-106

Hecky R. E., 1993. *The Eutrophication of Lake Victoria*. Verhandlungen International Vereingung Limnology. Sturtgart 25: 39-48

Hummert C., J. Dahlmann, M. Reichelt, and B. Luckas. 2001. Analytical techniques for monitoring harmful cyanobacteria in lakes. Lakes & Reservoirs: *Research and Management* 6:159-168

Kalff, J. 2002. *Limnology*. Prentice Hall, Upper Saddle River, New Jersey.

Kling H. J., R. Mugidde and R. E. Hecky, 2001. *Recent changes in the phytoplankton community of Lake Victoria in response to eutrophication. The Great Lakes of the World (GLOW): Food-web, health and integrity.* Backhuys Publishers, Leiden, The Netherlands: 47-65

Krietz, L., A. A. Ballot, C. Wiegand, K. Kotut, C. A. Codd and S. Phlugmacher, 2001. Cyanotoxin-producing Bloom of *Anabaena flos-aquae*, Anabaena discoidea and *Microcystis aeruginosa* (Cyanobacteria) in Nyanza Gulf Lake Victoria, Kenya. *Journal of Applied Botany*. Angewandte Botanik 76:179-193

Lehman J. T. and Branstrator D.K., 1994 Nutrient dynamics and turnover rates of phosphate and sulfate in Lake Victoria, East Africa. Limnol. Oceanogr. 39(2) 227-233

Lung'aiya, H. B. O., A. M'Harzi, M. Tackx, J. Gichuki, and J. J. Symoens, 2000. Phytoplankton Community Structure and Environment in the Kenyan Waters of Lake Victoria. Freshwater Biology 43. Blackwell Science Ltd: 529-543.

LVEMP, 2002. Integrated water quality/limnology study of Lake Victoria: *Final technical report*. COWI/DHI, Denmark.

Maitland, P.S. 1990, *Biology of fresh waters*, Second edn. Blackie and Sons Limited, USA.

Mugidde, R., R. E. Hecky, L. L. Hendzel, and W. D. Taylor. 2003. Pelagic nitrogen fixation in Lake Victoria (East Africa). *Journal of Great Lakes Research 29* (supplement2): 76-83

Ochumba, P. B. O., & D. I. Kibaara, 1989. Observations on blue-green algal blooms in the open waters of Lake Victoria, Kenya. *African Journal of Ecology* 27: 23-34

Talling J. F., 1966. The annual cycle of stratification and phytoplankton growth in Lake Victoria (East Africa). . *Inernationale revue der Gesamten HydroBiologie Und Hydrography* 51: 545 – 621

Vershuren D., Johnson T. C., Kling H. J., Edgington D. N., P. R. Leavitt, E. T. Brown, M. R. Talbot and R. E. Hecky, 2002. History and timing of human impact on Lake Victoria, East Africa. *The Royal Society,* London: 289-294

Wetzel, R. G. and Likens G. E., 1991. *Limnological analysis – 2nd edition.* Springer-Verlag New York, Inc.: 391

Ecosystem characteristics and the lower reach ecology rehabilitation practice of the Yellow River basin

Hao Fuqin[1], Huang Jinhui, He Zhijuan, Wang Xingong

Yellow River Water Resources Protection Institute, 450004, Zhengzhou, China

Abstract

According to characteristics of the Yellow River Basin such as topography, physiognomy, climate, hydrology and vegetation, the whole basin can divided into 3 ecotope and 10 ecology subregions, on the basis of analyzing characteristics of each ecological subarea, primarily identify main ecological environment issues of the Yellow River Basin. The lower reach of the Yellow River is typical Suspended River" and braided channel, at present it has many issues such as frequent dry, water environmental deterioration, channel sedimentation atrophy and wetland acreage atrophy of fan delta. This paper analyzes the effect of water quantity dispatching of the Yellow River and water and sand diversion on ecology rehabilitation in the lower reach river channel of the Yellow River, introduces estuary ecology rehabilitation works practice, and brings forward direction and objective of river ecology system rehabilitation of the Yellow River.

Key words: the Yellow River; ecological environment; ecology rehabilitation.

The stabilization and benign development of ecosystem are main symbols of river health, maintaining and recovering ecosystem health of the Yellow River Basin is the precondition and base of realizing health of the Yellow River. Presently, along with the increment of basin population and rapid development of economic society, the ecosystem of the Yellow River Basin has already suffered from all kinds of disturbances by nature and people, the bearing pressure of the Yellow River Basin increases gradually, take lower reach dry as symbol, the basin ecosystem presents the trend of integrated deterioration.

In order to slow and restrict the continuous deterioration of ecosystem, maintain and recover ecosystem health of the Yellow River Basin, all-round research and analysis must be made to ecosystem of the Yellow River Basin, analyze aiming at existing ecosystem issues one by one, then make the research and practice work of ecosystem rehabilitation. On the basis of identifying ecosystem characteristics of the Yellow River Basin, this paper identifies main ecology protection objective and existing ecological environment issues, illustrates the existing main issues of the lower reach ecosystem of the Yellow River, introduces the developed ecosystem rehabilitation practice work of the lower reach, and brings forward the direction and objective of ecosystem rehabilitation of the Yellow River Basin.

General situation of the Yellow River Basin[1]

The Yellow River is the second largest river in China, and is also the most important water source in northwest and north of China. It originates from Yueguzonglie basin at north foot of Bayankala Mountain of Qinghai-Tibet Plateau, and flows across nine provinces (areas) such as Qinghai, Sichuan, Gansu, Ningxia, Neimenggu, Shanxi, Shanxi, Henan and Shandong, it flows into Pohai in Kenli County of Shandong Province. It is 5464km long. The basin area is 795,000 km^2 (including 42,000 km^2 flow area).

River channel of the main stream of the Yellow River can be divided into upper, middle and lower reaches in accordance with basin characteristics. Estuary town from river head to Neimenggu is the upper reach, with basin area of 428,000 km^2, and the length of river channel is 3471.6 km. Here are many canyons, with large amount of water and abundant water resources. Taohuayu from estuary town to Henan Province is the middle reach, with basin area of 344,000 km^2, the length of river channel is 1206.4 km, and is the main source area of Yellow River flood and sediment load. From Taohuayu to estuary is the lower reach, with basin area of 23,000 km^2, the length of river channel is 785.6 km, sediment accumulation is severe, wandering of river channel is frequent, accretion surface inside bank is generally 3-5 m higher than ground surface outside bank, some reaches arrive upper 10m, and become the world famous “Suspended River”.

Terrain of Yellow River Basin is generally divided into three great ladders. The western ladder is located in east side of Qinghai-Tibet Plateau with altitude higher than 3000m; altitude of middle part Loess Plateau is generally 1000-2000m; the eastern ladder is a plain with altitude lower than 100m. The climate spans three climatic zones: High Plateau Climate Zone, the Medium Temperate Zone and the South Temperate Zone. The annual temperature for in the basin decreases from south to north and from east to west. The precipitation distribution decreases from south to north, and amount for the whole year mostly concentrates between July and October. Natural vegetation distribution is effected by oceanic monsoon. There are three vegetation type zones from southeast to northwest which are forest steppe, steppe and desert steppe.

Characteristics of ecological subarea and ecosystem in the Yellow River Basin

The Yellow River passes through plateau, mountainous region, hills, plains and various relief types. The watershed ecosystem has various types, and the ecosystem characteristics of different reaches and areas are different, so the characteristic of ecosystem of the whole Yellow River Basin is difficult to be understood comprehensively. In order to accurately understand the ecological characteristic of the Yellow River, we devide it into ecological subareas.

The ecological factors affecting watershed characteristics mainly include terrain, climate, soil, hydrology, biology, Human activity and other factors. Numerous factors and complex environment causes the difficulty of subarea, so the ecological subarea in the Yellow River Basin must be divided according to key factors. Terrain and topography are the main factors affecting the characteristics of the Yellow River Basin. The change of terrain and physiognomy causes the variance of other ecological factors, such as precipitation and temperature, so the grade-I ecological subarea in the Yellow River Basin shall be divided according to terrain factor. Grade-II key factors may have different representations in different ecotopes, some are soil erosion factors, some are vegetational cover degree factors, while others are probably regeneration of water resources and land use condition. Integrating the above analysis, and combining the characteristics of ecological factors in different regions, we can divide the Yellow River Basin into three great ecotopes and ten ecological subregions. The environmental characteristics of each ecotope is given ino Table 1.

Table 1. Characteristics of Each Ecotope and Ecological Subregion in the Yellow River Basin.

Ecotope	Ecological subregion	Location	Main environmental characteristics	Biological characteristics
Ecotope of upper reach of the Yellow River	Ecological subregion on Yellow River head	The Yueguzonglie basin, riverhead region above Maduo	Plateau climate region, cold and dry, little rainfall. Numerous rivers and lakes.	Characteristic of upland vegetation, the hardy spruce and fir tree is main vegetational types, greater biomass, and lower productivity.
	Ecological subregion of plateau and river valley	Section of river valley among mountains	Plateau climate region, cold and dry, steep mountain, narrow river channel.	Hardy upland vegetation are main vegetation, less artificial interference, short growth period, great vegetational cover degree, category of plant and animal is less, productivity of plant is lower.
	Ecological subregion of the Hetao plain	From Zhongwei of Ningxia to Tuoketuo of Mongolia, length of 750 km	Lie in semiarid region of Ning-meng with dry climate and greater evaporation capacity. It is the area with least precipitation amount in whole basin, width of river channel is 50km, and the altitude is 900-1200 m.	Herbaceous plant of tolerant to drought are main vegetation type, surface of river is larger, variety of biology is higher.
	Ecological subregion of Erdos plateau		It is arid region of upper reach of the Yellow River, altitude is 100-1400m, being table shape dry denudation plateau formed by airborne sediment land feature, directorate general of mineral resourcesdhands are numerous.	Vegetation coverage percentage is low, productivity is not high, variety of biology is less.
Ecotope of middle reach of the Yellow River	Ecological subregion of Loess Plateau	Start from the Great Wall north, extend to Qinling Mountains south and to Taihang Mountain east	Climate is dry with greater evaporation capacity. Altitude is 1000-2000 m; plateau form, girder, loess hill and channel are its principal part of geomorphy. Loose soil texture, sparse vegetation, and grave soil loss.	Less vegetation, unique habitat, biology category in directorate general of mineral resourcesdhand is the peculiar section of the Yellow River Basin.

	Ecological subregion of Fen-wei basin	Taiyuan, Yuncheng, and Guanzhong basin.	Altitude is 500-1000 m, the widest place of altitude is 40km, has abundant groundwater and spring river, fertile soil texture, rich products.	Agriculture vegetation mainly , the consumption of agricultural production to water resources is larger.
	Ecological subregion of Xiaoshan, Xiong'er and Taihang mountainous region		Altitude mostly exceeds 1000m, and is watershed of Yellow River, Changjiang River and Huaihe River.	Important position, higher biology variety, more species communion, various vegetation types.
Ecotope in lower reach of the Yellow River	Ecological subregion of alluvial plain in lower reach	It is formed by the alluviation of the Yellow River, Haihe and Huaihe Rivers, is the second largest plain of our country.	Flat physiognomy, wide river channel, grave sediment accumulation, the river channel is higher than surrounding plains, and become famous "Suspended River".	Large wandering of river, short water resources, greater population density, area of wetland is large, plentiful types, rich species, many key protective biology, ecological meaning is important and numerous key important protection biology.
	Ecological subregion of hills in Shandong Province	It is composed of Taishan, Lushan and Yishan Mountains.	Altitude is 400-1000 m, and is the lower reach right bank barrier of Yellow River, there are Laiwu, Xintai and other altitudes and plains with different sizes.	Vegetation cover degree is high, man-made influence is serious, variety of biology is higher, various types.
	Ecological subregion of fan delta	Ninghai in Lijin County	It is formed by modern sediment accumulation, wide zone, flat physiognomy, various species, seawater and freshwater connect, and become famous ecological interleaving region.	Variety of species is high, seawater and freshwater connect together, become famous estuary wetland of ecological interleaving region.

From the characteristics of each ecotope listed in the analysis Table 1, the Yellow River Basin basically belongs to an area with weak ecology and high sensitivity. The stability of the ecosystem is worse, and can easily be destroyed by external interference. According to investigation, at present the most serious damaged ecotope of the ecosystem of the Yellow River are: ecological subregion on Yellow River head, ecological subregion of Loess Plateau and ecotope in lower reach of the Yellow River. The main issues in each area are: pasture lands degradation and dry lakes in river head area, soil and water loss problem in Yellow River plateau, frequent dry channel sedimentation and fan delta wetland atrophy in lower reach ecotope, in which the lower reach ecotope faces the most and complex issues, we put stress on introducing main existing issues and ecology rehabilitation work of lower reach ecotope.

Main issues existed in the lower reach ecosystem of the Yellow River

Frequent dry of the lower reach

As the continuous improvement of water resources development strength of the Yellow River, insufficient control and management capacity, from 1972 to 1999, the lower reach of the Yellow River appears frequent dry phenomenon, in the end of 1990s, the dry became much serious, the last time of dry extended rapidly from former May and June to winter-spring season and summer-autumn season, even during the flood season. During 27 years from 1972 to 1998, Lijin Station in the lower reach of the Yellow River appeared dry for 21 years, accumulatively dry for 1050 days, the annual dry in dry year is 50 days, and dry extending to Henan Province borders is 5 years. Dry in 1997 is the most severe, and Lijin Station dry for 226 days.

The frequent dry in the lower reach of the Yellow River seriously damages the ecological balance, which makes unfavorable water environmental condition deteriorate furthermore. The estuary region is in state of dry or small water flow for a long time, with atrophy of river channel, underground water can not get sufficient freshwater supply, which aggravates seawater intrusion in estuary region, and makes salination areas increase. Dry also makes wetland water environmental condition in the Yellow River Delta out of balance, seriously threats the existence of aquatic lives, wild plants and birds in wetland protection area, and leads to degradation of ecological environmental system in estuary wetland and decrement of biology variety. Meanwhile, under the condition of decrement of water flow in river channel, self purification capacity of water decreases, however at the time of river dry, drainage and sewage water still drain into the Yellow River, massive contaminations accumulate in the river channel, and lead to serious deterioration of water quality when water flowing again.

Deterioration of water environment

Incoming flow of the upper and middle reach in the Yellow River last a little dry, water resources development and utilization degree improves year by year, and lead to rapid descend of water resources amount in the lower reach of the Yellow River. On the other hand, the serious polluted tributaries such as Luohe River, Qinhe River and Manghe River, and drainage and sewage water amount of pollution estuary into the Yellow River increases steadily. The two aspects' factors make water environmental deterioration conditions in the lower reach of the Yellow River urgent increasingly. Table 2 shows the water quality estimation results of key monitoring sections for total 3 years in the lower reach of the Yellow River from 2002 to 2004.

Table 2. Statistical Table for Water Quality Estimation Results of Key Monitoring Sections in the Lower reach from 2002 to 2004.

Year	Period	Huayuankou	Gaocun	Aishan	Luoshan	Lijin
2002	Non-flood season	Inferior V	?	-	-	?
	Flood season	Inferior V	Inferior V	-	-	?
2003	Non-flood season	Inferior V	?	?	?	?
	Flood season	?	?	?	?	?
2004	Non-flood season	?	?	?	?	?
	Flood season	?	?	?	?	?

Note: *According to environmental function and protection objective of Chinese surface water area, orderly divide into five classes in accordance with high and low functions:*

Class I is mainly applicable to head water and state natural protection area;

Class I is mainly applicable to grade I protection area of surface water source area of centralized living drinking water, habitat of rare aquatic lives, spawning site for fishes and shrimps, food for young fish etc.;

Class II is mainly applicable to grade II protection area of surface water source area of centralized living drinking water, wintering ground for fishes and shrimps, migration pathway, aquaculture area and other fishing zones and swimming areas;

Class IV is mainly applicable to general industrial water area and recreational water area that keeps indirect contact with human body;

Class V is mainly applicable to agricultural water area and common landscape required water areas.

Seen from Table 2, the water pollution conditions in the lower reach of the Yellow River in recent three years were quite severe, especially in years of 2002 and 2003 with a little dry of incoming flow, basically all the section water quality in the lower reach were at levels of Class V and interior Class V, pollution of Huayuankou reach was more serious, during the non-flood season of whole 2002 and 2003, the section water quality were all the interior Class V, and water body basically lost using function.

Aquatic environment is the base that aquatic lives relying on, especially the weak and small young fish, which have stronger dependence on fine aquatic environmental conditions. The river channel in the lower reach of the Yellow River is the spawning site for many rare fishes and pathway for oceanic migratory fish, if water quality deteriorates for a long time, the environment of fishes and other aquatic lives will suffer from damage, which seriously effects their existence and multiplication, decreases the biology variety in aquatic area, and quickens the deterioration of ecosystem.

Channel sedimentation and boost of flood level

The serious water and soil loss in the middle reach of the Yellow River and decrement of runoff in the lower reach, lead to grave sedimentation and atrophy in the river channel of the lower reach, riverbed rising gradually and the Yellow River becoming Suspended River", seriously threated flood control safety. Since 1986, the annual sedimentation amount of gross section was 245 million *Ton*, including 166 million *Ton* channel sedimentation, occupied 71% of gross section, during 10 years aggradations of river channel in the lower reach was 1.06~ 1.87m. The grave sedimentation of main channel made flood level raised, and appeared the historic highest flood level for many years.

Channel sedimentation makes flow capacity of channel of main stream decrease obviously, flow discharge of flat shoal decline, once appears the flood plain, it will lead to grave disaster. In August 1996, Huayuankou Station happened flood peak discharge of 7860m^3/s, the maximum sediment concentration is 126kg/m^3, the water level of Huayuankou Station reached 94.73m, great flood plain made water flow to high beach, flood went along bank, more than 3 million *Mu* beach areas were submerged, the population suffered from disaster arrived up 1 million, losses were greater than that was led by superflood in 1958 with flow discharge of 22300m^3/s.

Ecological issues in fan delta

Incoming water flow in the Yellow River estuary decreases, freshwater wetland atrophies

Since 1980s, along with rapid development of watershed economy and effect of global warm and dry climate, water quantity that entering into estuary district of the Yellow River decreased year by year: During 1986 to 2001, Lijin Hydrometric Station, more than 100km far from estuary, the measured annual runoff amount was just for 12 billion m^3, which only occupied 36% of mean value for many years, including merely 1.8 billion m^3 measured runoff amount of Lijin in 1997. The Yellow River is the main water supply source in estuary district, as the decrement of its incoming water flow, the domestic water and process water in estuary district is impracticable to ensure, and the ecological water was heavily insufficient. According to the analysis result of water resources amount in the Yellow River estuary and showing of TM image data, for recent years, the average supply water amount in estuary wetland is under 20% of normal year, large freshwater wetland appears dry and dying out as long term water lack, disappearance rate of estuary wetland has reached up 70%. Connectedness of clot passages in wetland ecosystem and ecological completeness are damaged, the Yellow River wetland environment relied by growth and existence of rare birds faces dying out, the ecological stability of the Yellow River delta suffers from grave threat.

Sediment of entering into sea decreases, sea coast erodes increasingly

From 1976 to 2000, the Yellow River delta eroded for 16.78km. During the 25 years, the annual average incoming sediment load amount decreased year by year, 827 million *Ton* from 1976 to 1986, 455 million *Ton* from 1989 to 1996, and 176 million *Ton* from 1996 to 2002.

Soil salinization aggravates

As the Yellow River estuary becoming into land is later, terrain is low and flat, with microtopography as hill, slope and depression, ground water elevation is high, degree of mineralization is great, evaporation is strong (rate of evaporation and rainfall is 3.24: 1), as the side leakage of the Yellow River water, amplitude of setup of sea water and tide aggression, native salinization as well as soil salinization are quite severe. The storm and tide disaster caused by gale, aggresses for scores kilometers at area without anti-tide dam, and makes new or aggravating brine salting of soil. The natural vegetation is also weak, native vegetation in grasslands and waste lands are mainly plants with strong salinity tolerance, if there is no effective protection measure, once these plants with salinity tolerance are damaged, especially on low and flat soil area, lower saline matter rises rapidly, makes soil translate into heavy halomorphic soil. Soil salinization leads to low yield of large cultivated lands because of salinization damage. The serious rejected cultivated lands shall recover to normal ecology by ten years.

Salinity of near waters improves, content of nutrition salt decreases, and productivity of commercial fishery declines

Nutrition salt is the substance base needed by aquatic lives, the important chemical substance base composing of estuary and near waters ecological environment. For recent 20 years, the decrement of the Yellow River water quantity of entering into sea makes reductive scope of interjunction area of river water and sea water, promotes toward estuary, at the same time, the flux of sea nutrition salt declines greatly, leads to the increment of estuary near waters salinity, but content of nutrition salt decreases for more than 50%, which directly effects the primary productivity of commercial fishery in sea area, leads to the great decline of fish kinds and quantities in estuary sea area, reaches up 95%, and seriously damages the ecological environmental balance of near waters.

Ecology rehabilitation practice in the lower reach of the Yellow River

In order to relieve the sharp supply and demand contraction of the Yellow River water resources, settle dry problem serious day by day in the lower reach, restrict aggradations atrophy of the lower reach river channel, the Managerial Department of the Yellow River Basin implements unified dispatching of the Yellow River water quantity and test of water and soil diversion, to a certain extent, the development of these works not only effectively settles dry and sediment load issues that troubled us for many years, but also has certain rehabilitation effect on ecosystem in the lower reach of the Yellow River. In addition, aiming at the wetland area atrophy problem in fan delta of the Yellow River, the Managerial Department of the Yellow River specially develops ecology rehabilitation works to estuary wetland.

Unified dispatching of the Yellow River water quantity

In order to relieve the supply and demand contraction of the Yellow River water resources and dry complexion serious day by day in the lower reach, in March 1999, the Managerial Department of the Yellow River formally implemented unified dispatching of water quantity from Liujiaxia Reservoir to Toudaoguai and from Sanmenxia Reservoir to main stream reach of Lijin. In 2001, they also carried out unified dispatching of water quantity from Toudaoguai to main stream reach of Sanmenxia.

On the basis of comprehensively considering industrial, agricultural and process water in each province and domestic water of masses in towns and villages, the unified dispatching of water quantity makes unified control to the five reservoirs of the Yellow River, preferentially ensure city life and industrial water supply and also can farthest satisfy agricultural water during key period, and increases ecological environmental water supply.

After implementing water quantity dispatching, realize no dry in the whole year of the Yellow River continuously for 6 years under the condition of a little dry even specially dry, turn the complexion of frequent dry in the lower reach, the main ecology rehabilitation functions are as follows:

1. Water quantity dispatching makes runoff amount in the lower reach in non flood season increase, the appeared months of inferior Class V water decreases obviously compared with former, water quality in the lower reach of the Yellow River improves, avoid the happening of water pollution events after returning flow of the Yellow River by the contamination accumulated in Changqing and Pingyin beach areas caused by dry of the Yellow River

2. Unified dispatching makes the river channel wetland obtain rehabilitation, which was damaged by dry of the Yellow River in 1990s with length more than 200km^2, some river channel wetlands are rehabilitated, and the functions of river channel wetlands exert.

3. The unified dispatching of the Yellow River water quantity ensures the ecological environmental water to a certain degree in the lower reach of the Yellow River, especially ecological environmental water of fish spawning young period, and is benefit for the recovery of fish resources in the Yellow River. After dispatching water, the copper fishes disappeared in 1980s in the Yellow River appear in crowd again, some migratory fishes disappeared for many years such as sea saury (also named as long-tailed anchory and anchory) also appear in the lower reach of the Yellow River again.

4. Water quantity dispatching relieves the acute atrophy trend of delta wetland areas, compared with that of 2001, the freshwater wetlands of the Yellow River delta increases 4389hm^2, promotes the consequent succession of estuary wetland, the variety of wetland biology system increases obviously, the bird quantities in country-level protection area in the Yellow River delta increases from 187 in the begin of 1990s to present 283, kinds of rare wildlife is 459, which increases by almost one time compared with former.

5. Water quantity dispatching reduces the flux of nutrition salt, decreases the incidence rate of red tide, improve the ecological environment of estuary near waters in non flood season, better the growing condition of phytoplankton and living environment of fish in estuary near waters area, and have certain positive effects on ecosystem rehabilitation of estuary and near waters environment.

Water and sand diversion

In order to restrict continuous sediment and atrophy of river channel in the lower reach, improve flow capacity of river channel, from July 4, 2002 to July 15, 2002, through the unified water and sand division of Xiaolangdi and Sanmenxia reservoirs, the Yellow River Conservancy Commission firstly made test of water and sand division in the Yellow River successfully. From September 6, 2003 to September 18, 2003, combined with flood control and advance release, through the unified water and sand division of Xiaolangdi, Sanmenxia, Luhun and Guxian reservoirs, made the second test of water and sand division, in the subsequent 2004 and 2005, respectively make the third and fourth water and sand divisions.

Water and sand division means in the precondition of adequately considering sediment transport capacity of the Yellow River lower reach, take use of balancing storage of reservoir, make effective control and adjustment to water and sand, so as to relieve channel aggradations in the lower reach, even reach the effect of scouring or none aggradations, and realize objective of no rising of the lower reach riverbed.

Results of four times' water and sand division show that, water and sand division is one of the most effective methods for keeping health lives in the Yellow River, it restricts the continuously deteriorated trend of river pattern in the Yellow River lower reach, and has certain positive effects on ecosystem rehabilitation, mainly embodies as following aspects:

1. Water and sand division can reduce channel sedimentation atrophy, increase flat shoal flow discharge of main stream channel, and make flow capacity of main stream channel improve for a certain degree. Since 2002, the three times' water and sand division prototype tests realized erosion along whole line of river channel in the Yellow River lower reach, 260 million *Ton* sediment load were translated into sea, and flow capacity of river channel improved from less 2000 m^3/s before test to 3000 m^3/s;

2. The river flow of lower reach increases greatly, flood plain water flow offers timely supply for river channel wetland, restricts area atrophy of river channel wetland, many silver sands in the water and high organic content provide mass nutritive substances for growth and development of animals and plants in the wetland;

3. Water and sand division makes the flow of the Yellow River reach to the last hydrometric station Lijin Station improves greatly, and provides advantageous conditions for over irrigation of estuary wetland. During water and sand division period in 2005, the maximum flow of Lijin Station reached to 3000m^3/s, under the condition of massive over irrigation, about 1000 m^3 water of the Yellow River injected into wetlands in the Yellow River estuary, improved wetland ecological environment of fan delta, and had certain positive effects on ecosystem rehabilitation;

4. Water and sand division increases sediment load amount, makes wetlands area in estuary increase, at the same time, nutritive salts and organic substance contained in sediment load are benefit for multiplication and growth of fishes in estuary district and Bohai River water area, board water surface and continuous large flow provide fine living and multiplication spaces for migratory fishes;

5. Mass flow water of the Yellow River injecting into Bohai River, also has positive meaning for restricting long term encroachment of sea water, controlling salinization and protecting lands.

Ecology rehabilitation works of estuary wetland

Freshwater supply of the Yellow River is the basic guarantee of keeping ecosystem in estuary district. The common effects such as decrement of water and sand quantities of entering into sea in the Yellow River, sustainable and rapid development of estuary district economy and other factors, leads the estuary ecosystem in the Yellow River out of balance seriously. Water resources of the Yellow River is the important basis of rehabilitating and rebuilding ecosystem function in the Yellow River estuary, unified dispatching of water quantity, makes water flow in estuary increase in non flood season, and offer water condition for wetland rehabilitation.

In 2001, the state began to implement 300,000 *Mu* wetlands rehabilitation work at both sides of existing river channel of the Yellow River delta natural protection area at state level. The wetlands rehabilitation work does not affect flow path of the Yellow River north anabranch and flood capacity of existing river channel of the Yellow River. According to recent years' circumstances of incoming water and sand of the Yellow River and terrain characteristics of wetland rehabilitation area, adequately taking use of natural terrains of large back differences of longitudinal dam and great transverse slope of beach area, the works adopts sediment ejection scheme combined with bailing and artesian flow and given priority to artesian flow. While great river level can not reach the minimum water level of artesian flow diversion, adopt bailing sediment ejection; while great river level is higher than minimum water level of artesian flow diversion, adopt two modes of artesian flow and bailing sediment ejection. Main vegetations in recovered wetlands are chionese tamarisk and bulrush, depth of underground water is different, the shallow for 20-30cm, the deep for 80-100cm, and average for 50cm.

The works was finished in September 2003. According to site investigation, since the implementation of works, wetland ecosystem of the Yellow River delta improves obviously, freshwater wetlands areas of estuary increases obviously, growth of vegetation is flourishing, quantities and kinds of birds coming here for living through the winter and multiplying increase steadily, some rare birds within the country and overseas also appear here.

Work and objective for the future

The ecology rehabilitation work developed in the lower reach of the Yellow River relieves or restricts the deterioration of ecosystem to a certain degree, however, seen from ecological issues faced by the whole Yellow River Basin, the work is far not enough. In order to protect the weak ecological environment in the Yellow River Basin, develop research and practice work better for the future, we establish the ecological protection direction and objective:

1. Near term objective: Develop ecology investigation, explore evolution rules, establish protection planning, adopt protection measures, relieve or restrict deterioration of ecosystem in the Yellow River.

2. Medium term objective: Make research and practice work of ecology rehabilitation. Adopt structural measures and nonstructural measures, recover or rebuilt damaged ecosystem.

3. Long term objective: Establish and perfect protection and supervision system of ecological environment, and realize benign maintenance of the Yellow River ecosystem.

The realization of the Yellow River Basin ecosystem objective needs that we throw much more energy, at the same time, we also need broad attention and recognition of international and domestic society, pool the wisdom and efforts of everyone, jointly strive for stability and benign development of the Yellow River Basin ecosystem, and construct the Yellow River into river with friendly environment.

References:

Chief Compilation Office of the Yellow River Annals of the Yellow River Conservancy Commission, 1998. *Overview of the Yellow River Basin*. People's Publishing House of Henan Province.

Special key project on Yellow River harnessing, 2005. *Study on Ecological Environmental Water Demand in the Main Streams of the Yellow River*.

Seasonal change in fish species in a paddy field ditch linked to Lake Biwa, Japan

Masahiro Nakagawa and Takashi Suzuki

Department of Fisheries, Faculty of Agriculture, Kinki University, Nakamachi 3327-204, Nara 631-8505, Japan.
Email:gf998002@yahoo.co.jp

Abstract

As a step toward understanding the biodiversity in a paddy field ditch linked to Lake Biwa, the fish fcommunity were surveyed monthly from January 2000 to May 2005. The sampling site (4.0 m^2 and approximately 0.6 m deep) is at an intersection of the concrete paddy field ditch. A total of 3,362 fish specimens representing 24 species were collected by two people by using hand nets for 20 min during the day. Two cyprinid species, *Gnathopogon elongatus elongatus* and *Acheilognathus rhombeus*; spine loach *Cobitis biwae*; and gobiid fish *Rhinogobius* sp. were the most abundant species, comprising 66.0% of the total fishes. These were followed by three minors—*Zacco platypus* (8.5%), *Phoxinus lagowskii steindachneri* (4.6%), *Zacco temminckii* (4.0%). Far Eastern brook lamprey *Lethenteron reissneri* (1.0%) and Medaka *Oryzias latipes* (1.1%), which are described in the Japanese Red Data Book, were collected. Three invasive species, one cyprinid fish *Rhodeus ocellatus ocellataus* (0.42%), *Micropterus salmoides* (0.14%), and one gobiid fish *Tridentiger brevispinis* (0.03%) were rarely collected all year around. *G. e. elongatus* and *A. rhombeus* were frequently collected from spring to late autumn, while spine loach and *Rhinogobius* sp. were collected every season. Minors were occasionally collected mainly in late autumn and winter. *Z. platypus*, *P. l. steindachneri*, and *Z. temminckii* were largely collected in November–December 2001 and 2002, in November 2000, and in July 2001 and October 2002, respectively. However, these three minors were collected simultaneously from October 2004 to January 2005. This suggests that the paddy field ditch, which contained few invasive species, functions as a habitat and/or breeding ground for parent fish, a cradle for offspring, and a corridor for migration, and supplies endangered fishes with a refuge for *in situ* conservation. Moreover, a stable paddy field ditch could be an interesting subject for synecological studies by ecologists and ichthyologist.

Key words: endangered species, *Lethenteron reissneri*, species diversity

Introduction

Lake Biwa is the largest lake in Japan (surface area, 670.5 km^2; maximum depth, 104 m) and is a hotspot of biodiversity (with more than 50 fish species) that contains some endemic species. It is also one of the most important lakes from the viewpoint of natural heritage. Understanding the biodiversity of the lake has been a major focus of research long before the 1970s. However, after the 1980s, when the invasive alien fish species largemouth bass (*Micropterus salmoides*) and bluegill (*Lepomis macrochirus*) began occupying the coastal area of the lake, conservation and production of the lake's biodiversity became an essential focus area. The coastal area of the lake consists of sand, rock, parts of the river, attached lakes called "Naiko" that are geologically lagoons, and paddy water systems. These various structures have been essential in creating the abundant biodiversity observed in Lake Biwa.

In particular, the paddy water systems around lakes and rivers have been receiving increasing attention in ecology, ichthyology, and conservation biology. This is because it is believed that these systems might have effectively generated fishery products and thus maintained biodiversity, including the conservation of several endemic and endangered species. Fish community of a paddy water system was first systematically investigated by Saitoh *et al*. (1988) in the Kinki region, following which there was an explosion of research in the last several years, and this has produced mixed results (Katano *et al*., 2001, 2003; Iguchi *et al*., 2003; Nakamura & Oda, 2003).

However, very little information is available on paddy water systems in literature related to Fish community around Lake Biwa, and there are few long-term studies on it. This will hamper the sound maintenance of the ecosystem of such systems. Hence, this paper aims at contributing information on the fish community in a paddy field ditch linked to Lake Biwa. In this study, we chose a model area and examined a number of species and individuals, their maturity, the monthly and annual changes in Fish community, and the transition of species diversity.

Materials and methods

Study area

The study area is an intersection (4.0 m^2 in area and approximately 0.6 m deep; 35°24' N, 135°14') of a paddy field ditch in Higashiazai County in the northern part of Lake Biwa, in Shiga Prefecture, Central Japan. Water flows from the Ane River, one of the major rivers in the prefecture, into Lake Biwa through a connected lake. Water flows into the ditch throughout the year, and the ditch has never run dry. Water temperature is highest (approximately 30°C) in July and lowest (7°C) in February; the average temperature is 16–18°C. The side and bottom of the study area and the ditch linked to it were made of concrete. The bottom of the ditch was then covered with mud and it was the habitat of bivalves such as *Anodonta woodiana*, *Unio douglasiae*, *Inversidens japanensis*, and *Corbicula leana*.

Sampling methods

In order to quantitatively examine the Fish community, specimens were collected every month from January 2000 to May 2005 with the exception of February 2000. The collection was done for 20 min between 1400 and 1530 hours by two individuals who used hand nets. Specimens were generally identified according to Nakabo (2000) and some were partly fixed with 10% formalin solution for confirming their identity. The remaining specimens were released immediately after examination. In order to facilitate an understanding of the pattern of utilization of the study area from the viewpoint of each growth stage, the maturity of the specimens was synthetically examined by checking the body size, nuptial coloration, and pearl organs.

Species diversity

The species diversity of the specimens was examined by means of the Shannon-Wiener Index in order to record the monthly and annual changes in the biodiversity of the Fish community. The index (H') was calculated using the formula $H' = - Sp_i \log_2 p_i$ (MacArthur, 1955), where p_i is the proportionality of abundance of species *i* in the sample. This index was used because it highlights the relative abundance of rare vs. dominant species, the consideration of which is important in an ecosystem that does not contain a large number of species.

Results

Number of species and individuals along with their maturity

A total of 3,362 fish specimens representing 24 species and 9 families were collected in this study (Table 1). Two cyprinid species, field gudgeon *Gnathopogon elongatus elongatus* and flat bitterling *Acheilognathus rhombeus*; spine loach *Cobitis biwae*; and common freshwater goby *Rhinogobius* sp. OR were found to be the most abundant species and comprised 66.0% of the total fish. These were followed by three minors—pale chub *Zacco platypus*, Japanese fat minnow *Phoxinus lagowskii steindachneri*, and dark chub *Zacco temminckii*. Rosy bitterling *Rhodeus ocellatus ocellatus*, largemouth bass, and short-spined Japanese trident goby *Tridentiger brevispinis* were the invasive species and comprised only 0.6% of the total fish. On the other hand, Far Eastern brook lamprey *Lethenteron reissneri* and Medaka *Oryzias latipes*, which are described in the Japanese Red Data Book, comprised 2.1% of the total fish.

The pattern of utilization of the study area was different among fish species and was categorized as accidental invasion, growth, or spawning. We rarely collected mature specimens of Ayu *Plecoglossus altivelis altivelis* and short-spined Japanese trident goby, whereas immature specimens of carp *Cyprinus carpio*, pike gudgeon *Pseudogobio esocinus esocinus*, Far Eastern catfish *Silurus asotus*, and largemouth bass were found. Accidental invasion by these fishes is a strong possibility due to the remarkably low frequency of collection (1.2%). Both mature and immature specimens were collected from the other species. These species use this area at least for their growth. Some species of these fish utilize this area for spawning because a fertilized egg of oily gudgeon *Sarcocheilichthys variegatus microoculus* was also found in a bivalve during the study period. The abundant presence of bivalves in the area is probably advantageous to the bitterlings—slender bitterling *Tanakia lanceolata*, oily bitterling *Tanakia limbata*, rosy bitterling, and flat bitterling—which deposit bivalves in the breeding season.

Table 1. Number (%) and maturity of fishes collected from January 2000 to May 2005 in the paddy field ditch connected to Lake Biwa.

	Species	No. of fish (%)	Mature	Immature
1	*Lethenteron reissneri*	32 (1.0)	+	+
2	*Plecoglossus altivelis altivelis*	32 (1.0)	+	–
3	*Zacco platypus*	286 (8.5)	+	+
4	*Zacco temminckii*	133 (4.0)	+	+
5	*Phoxinus lagowskii steindachneri*	156 (4.6)	+	+
6	*Gnathopogon elongatus elongatus*	651 (19.4)	+	+
7	*Sarcocheilichthys variegatus microoculus*	3 (0.1)	+	+*
8	*Pseudogobio esocinus esocinus*	2 (0.1)	+	–
9	*Cyprinus carpio*	3 (0.1)	–	+
10	*Carassius auratus grandoculis*	75 (2.2)	+	+
11	*Carassius auratus langsdorfii*	91 (2.7)	+	+
12	*Tanakia lanceolata*	83 (2.5)	+	+
13	*Tanakia limbata*	26 (0.8)	+	+
14	*Rhodeus ocellatus ocellatus*	14 (0.4)	+	+
15	*Acheilognathus rhombeus*	324 (9.6)	+	+
16	*Misgurnus anguillicaudatus*	121 (3.6)	+	+

17	*Cobitis biwae*	514 (15.3)	+	+
18	*Silurus asotus*	1 (0.1)	–	+
19	*Oryzias latipes*	38 (1.1)	+	+
20	*Micropterus salmoides*	5 (0.1)	–	+
21	*Odontobutis obscura obscura*	19 (0.6)	+	+
22	*Rhinogobius* sp. OR	731 (21.7)	+	+
23	*Tridentiger brevispinis*	1 (0.1)	+	–
24	*Gymnogobius urotaenia*	21 (0.6)	+	+
	Total	3362		

+ and – represent presence and absence, respectively. * represents the presence of fertilized eggs in bivalve.

Monthly changes in fish community

The dominant species were native fish species found throughout the year, although seasonal changes in fish community were clearly observed in this study (Figure 1). In winter, field gudgeon and flat bitterling generally disappear, whereas spine loach and freshwater common goby are found in every season. The percentages of the latter two species in the total specimens increased sharply during winter because many fishes also disappeared during this season. Many of the cyprinid fishes such as crucian carps and bitterling were frequently collected from April to September (Figure 1., portions shaded by dots,) whereas the three minors—pale chub, Japanese fat minnow, and dark chub—were mainly collected in late autumn (Figure 1., portions shaded by slashes). Far Eastern brook lamprey was frequently collected from the area around the weeds at the water's edge and in the mud at the bottom of the lake between November and April, although adult varieties were rarely collected in July. Ammocoetes of lamprey were frequently collected in January and February. Medaka was collected from July to September. Invasive species were randomly collected in this study.

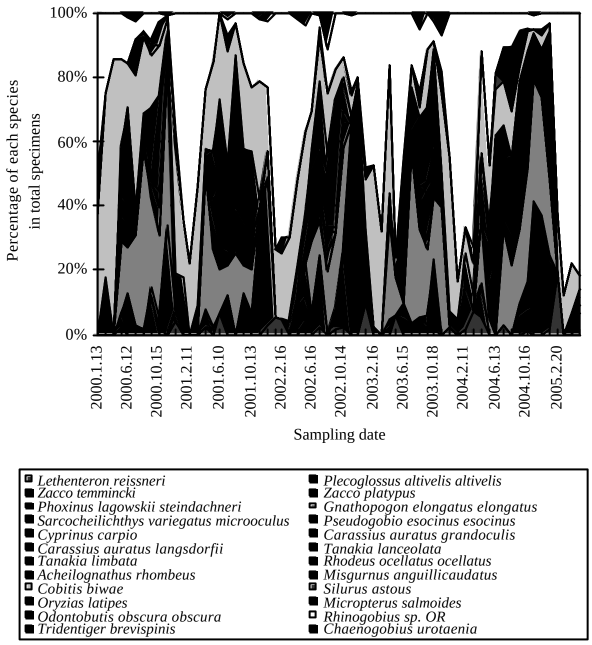

Figure 1. Annual changes in the percentage of each species in the total specimens collected in the paddy field ditch connected to Lake Biwa.

Annual changes in fish community

The occurrence of many of the species exhibited regular increases and decreases during the study period (Figure 1.) However, the population of the three minors exhibited dramatic changes every year. Pale chub, Japanese fat minnow, and dark chub were mainly collected in November–December 2001 and 2002 (16–85 specimens), in November 2000 (57 specimens), and in July 2001 (13 specimens) and October 2002 (12 specimens), respectively. Mass collection of pale chub in 2002 established it as the dominant species. However, these three minors were collected simultaneously from October 2004 to January 2005. Far Eastern brook lamprey (Figure 1., black) and Medaka (Figure 1., polka dot), which are mentioned in the Red Book, did not tend to decrease during the course of this study.

Monthly and annual changes in species diversity

Table 2 shows the values of the Shannon-Wiener Index of species diversity in the study area. The value was 3.20 at the highest point in June 2000 and 0.53 at the lowest point in March 2005. The average was 2.77 at the highest point in October, and it showed the lowest annual dispersion. It was 1.10 at the lowest point in March. There was a large dispersion in April from 0.59 in 2000 to 2.46 in 2004. The indexes were more than 2.50 from June to October and less than 2.01 during the other months. However, the index was exceptionally high in December 2004 and January 2005.

Table 2. Monthly and annual changes in the Shannon-Wiener Index of species diversity in the paddy ditch connected to Lake Biwa.

Year	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
2000	1.41	–	1.41	0.59	2.66	3.20	2.29	1.83	2.73	2.89	1.86	1.94
2001	1.29	0.76	1.32	2.24	2.50	2.66	2.94	2.07	2.92	2.70	1.94	2.04
2002	1.02	1.26	1.07	1.88	2.54	2.81	3.15	2.86	2.37	2.66	1.57	0.85
2003	2.04	1.14	0.90	2.19	1.14	2.30	1.99	2.98	2.77	2.75	2.34	1.36
2004	0.79	1.57	1.36	2.46	2.11	2.09	2.87	2.77	2.81	2.85	2.24	2.73
2005	2.93	1.32	0.53	1.01	1.07	–	–	–	–	–	–	–
Aver.	1.58	1.21	1.10	1.73	2.01	2.61	2.65	2.50	2.72	2.77	1.99	1.78
S.D.	0.78	0.30	0.34	0.75	0.72	0.44	0.49	0.51	0.21	0.10	0.31	0.71

– represents “not investigated.”

Discussion

The paddy water systems around lakes and rivers in Japan have been receiving an increasing amount of attention in the field of *in situ* conservation (Iguchi *et al*., 2003) because the population size began decreasing after the 1980s. In this study, in order to understand the monthly and annual changes in the fish community and the mechanism for conserving biodiversity in paddy water systems, the Fish community in the paddy field ditch connected to Lake Biwa was quantitatively examined from January 2000 to May 2005 using hand nets. A key distinguishing feature of this paper is its focus on a monthly examination with quantitative collection for approximately six years. The important findings of this study are as follows: (1) Two cyprinid species, field gudgeon and flat bitterling; spine loach; and freshwater common goby were the most abundant species and comprised 66.0% of the total fish. (2) The two cyprinid species were frequently collected from spring to late autumn, while spine loach and freshwater common goby were collected during every season. (3) Three minors—pale chub, Japanese fat minnow, and dark chub were occasionally collected mainly during late autumn and winter; however, the population size of these groups changed dramatically during this period. (4) Species diversity was high from June to October and low during the other months. These finding have been discussed below.

(1) Paddy water ditches connect core areas such as lakes and rivers and yield more species than the core area itself (Hosoya, 1982). In this study, 24 species, which is half the total number of species found around Lake Biwa, were collected, notwithstanding the concrete ditch. Katano *et al*., (2003) examined species diversity and the abundance of freshwater fishes in the paddy field ditch and showed that the natural stream bed, which had not been covered by concrete, had a greater number and biomass of fishes. On the other hand, a part of the triple concrete ditch sometimes continued to produce a wide variety of species (Katano *et al*., 2001), as shown in this study. The wide variety of species in this area may be due to the velocity of water, the structure of the ditch, and the volume of mud accumulation.

It is also remarkable that the current dominant species is not an introduced but some native species. Introduced largemouth bass and bluegill have presently occupied the area around Lake Biwa (Minobe & Kuwamura, 2001), and the two species usually exclude native species (Nakajima *et al*., 2001). Only five largemouth bass individuals and no bluegill were collected in this study. Because the two species prefer lentic water to lotic water like that in this study area, the species would be minor species.

On the other hand, judging from their maturity, many of the other species utilize the study area for their growth and/or spawning. The presence of some endangered fish species also indicates that the ecosystem of this area is sound and is important in terms of *in situ* conservation. Far Eastern brook lamprey and Medaka, which are sensitive to environmental destruction and are excellent indicators of biodiversity, were regularly collected in this study. The individual numbers collected in the area did not decrease, at least during the period of this study. This indicates that the area functions as a shelter from invasive species and as a part of the life cycle. Further, existing concrete ditches without fishes can recapture biodiversity if proper improvements are carried out.

(2) In this study, many cyprinid species, including the two dominant species, were frequently collected from spring to late autumn, whereas the other dominant species, spine loach and freshwater common goby, were collected during every season. Nakamura and Oda (2003) examined seasonal changes in the number and maturity of fishes ascending a paddy field ditch and showed that most individual field gudgeons, spine loaches, and freshwater common gobies were collected from April to June, and most of these were mature. These

results indicate that almost all species, including spine loach and freshwater common goby, seasonally migrate between paddy field ditches and core areas such as lakes and rivers. They also suggest that one of the major threats to fishes inhabiting the area around the paddy water system is evidently a fragmentation of the systems. Therefore, we should prevent any disturbance that causes fragmentation. (3) The three minors may be identified as being of the same guild from the viewpoint of their body forms as well as their seasonal migration from late autumn to winter, even though the fish generally exhibit different behavior patterns in terms of feeding, migration, etc. (Kawanabe *et al*., 2001). The guild sometimes became a temporal dominant and excluded the field gudgeon and/or the flat bitterling that were originally dominant in the study site, for example, during the period of November–December 2001 and 2002. These three minors have recently been collected simultaneously. However, a complete understanding of the simultaneous appearance and interpretation of their niche and guild requires further study.

(4) Species diversity is high from June to October every year. This indicates that the various fishes have appeared in the study area during the periods. Increase in the species diversity in December 2004 and January result from the simultaneous appearance of the three minors. The species diversity of the study area changes dynamically during the seasons and changes slightly during the years. The fluctuation in species diversity is probably a character common to other paddy water systems. Therefore, investigation of Fish community for environmental assessments should be performed during several different seasons over several years.

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References

Iguchi K., Yodo T. & Katano O., 2003. Habitat condition for freshwater fishes found in a paddy water system on Iriomote Island. *Jpn. J. Ichthyol*. 50, 115–121 (in Japanese, with an abstract in English).

Hosoya K., 1982. Freshwater Fish community of the Yoshii River, Okayama Prefecture. *Bull.Giogeogr. Soc. Japan*. 37, 23–35.

Katano O., Hosoya K., Iguchi K. & Aonuma Y., 2001, Comparison of Fish community among three types of rice fields in the Chikuma River basin. *Jpn. J. Ichthyol*. 48, 19–25 (in Japanese, with an abstract in English).

Katano O., Hosoya K., Iguchi K., Yamaguchi M., Aonuma Y. & Kitano A., 2003. Species diversity and abundance of freshwater fishes in irrigation ditches around rice fields. *Envr. Bio. Fish*. 66, 107–121.

Kawanabe H., Mizuno N. & Hosoya K., 2001. *Freshwater fishes of Japan*. Yamatokeikokusha, Tokyo (in Japanese).

MacArthur R. H., 1955. Fluctuation of animal populations and a measure of community stability. *Ecology*. 36, 533–536.

Minobe H. & Kuwamura K., 2001. Analysis of the change in Fish community and ecological environment in “naiko” around Lake Biwa. *Ecol. Civil Eng*. 4, 27–38 (in Japanese, with an abstract in English).

Nakajima T., Fujioka Y., Maehata M., *et al*., 2001. Relationship between fish distribution pattern and geomorphology in the Konan area surrounding the southern part of Lake Biwa. *Jpn. J. Limnol*. 62, 261–270 (in Japanese, with an abstract in English).

Nakabo T., 2000. *Fishes of Japan with pictorial keys to the species, 2nd ed*. Tokai University, Tokyo (in Japanese).

Nakamura T. & Oda N., 2003. Seasonal changes in fishes ascending a paddy field ditch in the Naka River system, central Japan. *Jpn. J. Ichthyol*. 50, 25–33.

Saitoh K., Katano O. & Koizumi A., 1988. Movement and spawning of several freshwater fishes in temporary waters around paddy fields. *Jpn. J. Ecol*. 38, 35–47 (in Japanese, with an abstract in English).

The primary production of an inland artificial lake in Zimbabwe

Ndebele Mzime Regina and Magadza Christopher C.H.

Department of Biological Sciences, University of Zimbabwe. PO MP167, Mount Pleasant, Harare, Zimbabwe

Email:ndebele@science.uz.ac.zw

Abstract

The primary production of a shallow, eutrophic inland lake in Zimbabwe was investigated during the period of March 2003 to May 2003. The study was carried out at three sites along the length of the lake. The average primary production in the lake was 8.2 mg h^{-1} (2 334 g C $m^{-2}h^{-1}$). There was no significant difference in primary production among the sites (P<0.05). Primary production levels at site 2 were found to be significantly higher than those of the last long-term primary production study recorded by Robarts (1979) at P<0.05, twenty-four years before the present study. The presence of dominating cosmopolitan phytoplankton species *Microcystis* sp. together with the high primary production values indicated that the lake is now hyper-eutrophic. The primary production values ranged from 1.1 mg h^{-1} (275 g C m^{2} h^{-1}) to 25.4 mg h^{-1} (6 350 g C m^{2} h^{-1}). There is an urgent need to address the deteriorating health of the lake.

Key words: eutrophication, Lake Chivero, primary production

Introduction

Lake Chivero (formerly Lake McIlwaine, 1952) is a man-made reservoir that was created along the Hunyani River mainly as a supply of potable water for the city of Harare. The lake has a capacity of 247 megalitres, a length of ten kilometers and surface area of 26.3 km^2. Since its creation, the lake became hypereutrophic (1960s) and temporarily recovered during the 1970s to early 1980s to a metastable state (Thornton 1980, Magadza 1994). However, the mesotrophic state of the lake was short-lived as from the late 1980s onwards, the water quality began to deteriorate and the lake became eutrophic once again with infestations of water hyacinth and algal blooms (Magadza 1994, 2003). In 1996, fish deaths prompted an investigation into the causes and possible mitigation measures of the lake’s eutrophication, when the concentrations of most nutrients were higher than during any other period as reported by Marshall (1995). In Lake Chivero, the primary production levels have generally increased with time, reflecting the increased nutrient loading in the lake. Tables 1 and 2 show primary production values of Lake Chivero and also that of some tropical lakes for comparison.

Table 1. Productivity-biomass levels in Lake Chivero and a few warm freshwater lakes*P.P. refers to the primary production of each lake, computed as the integrated primary production per square meter.

Lake	Kariba	Malawi	Tanganyika	Victoria	Chivero
Year of study	1988	1980	1975	1990-1991	1977-1980
Algal biomass mg/l	0.2-2.9	0.02-0.03	0.02-0.93	-	-
Chlorophyll a µg/l	2-11	-	0.1-4.5	8.4-24.5	2-45
*P.P. g Cm^{-2} d^{-1}	0.01-0.5	0.24-1.14	0.4-3.1	3.3-13.5	1.64-6.03
Species no.	155	504	92	602	-
Source	Cronberg 1997 1987	Hecky & Kling 1981	Hecky & Kling 1993	Mugidde Robarts 1979	

Table 2 Primary production values of freshwater lakes in southern Africa and the tropics (after Robarts 1982)

Lake or water body	Country	Primary production range	Source and year ($g C m^2 d^{-1}$)
Castanho	Brazil	0.05-1.50	Schmidt, 1973
Chad	Chad	0.70-2.69	Lemoalle, 1973
Cresent Is. Crater	Kenya	1.13-3.15	Melack, 1979
George	Uganda	1.95-5.80	Ganf, 1975
Hartbeesport	South Africa	0.40-30.9	Robarts, 1984
Naivasha	Kenya	1.39-2.33	Melack, 1979
Oloiden	Kenya	1.58-4.54	Melack, 1979
Sibaya	South Africa	0.23-1.85	Allanson, 1979
Winam Gulf	Kenya	1.61-3.68	Melack, 1979

Lake Chivero is one of the most productive and enriched lakes in the world and in the southern African region only Hartbeesport dam is comparable to it in terms of the primary production range (Tables 1 and 2). The Hartbeesport dam in South Africa is one hypereutrophic lake that is similar to Lake Chivero in terms of high pollution levels, proliferation of algal and aquatic weeds as well as a high primary production rates (up to 30.9 g C m^2 d^{-1}) as shown in Table 2 (Robarts, 1984). As a result of eutrophication and algal blooms, frequent fish deaths have been reported in both Lake Chivero and Hartbeesport dam. However the last systematic assessment of primary production rates in Lake Chivero was performed in the late 1970s by Roberts (1982). This study was to evaluate changes in primary production and relate it to current nutrient concentration status of the Lake.

Materials and methods

The study area

Figure 1. Sketch of Lake Chivero in relation to the city catchment and the main tributaries.

The lake is situated in southern Africa, Zimbabwe and is located some 35 kilometers southwest of Harare (17° 54' S, 30° 54'E) (figure 1.). The lake has a surface area of 26 km^2 and an average depth of 9 m, whilst the maximum depth is 27 m (Burke &Thornton, 1982). The lake is hypereutrophic, mainly because of the city sewage effluent that enters the lake, mainly due to over burdened waste water treatment facilities and unmanaged diffuse source pollution from storm water runoff and non compliant industries.

The sampling sites

The three sampling sites, covering the length of the lake, are located at the upper section of the lake (Site 3: S 17° 53.057', E 030° 46.405'), the middle section of the lake (Site 2: S 17° 54.551', 30° 49.063') and at the downstream section of the lake (Site 1: S 17° 54.895', 30° 50.701') near the dam wall.

Primary production was assayed on a depth profile of 0, 0.5, 1, 2, 3, 4, 5, 6 and 7 meters respectively on a fortnightly basis using the light and dark bottle method, after Vollenweider (1969). At each site and depth, one dark and two light bottles were suspended in the water for four hours, between 10:00 hrs and 14:00 hours.

A one-way ANOVA was used to test significance of site in primary production. A t-test was used to ascertain whether there had been any significant change in the primary production of Lake Chivero since 1976 (Robarts, 1979) to the time of the present study.

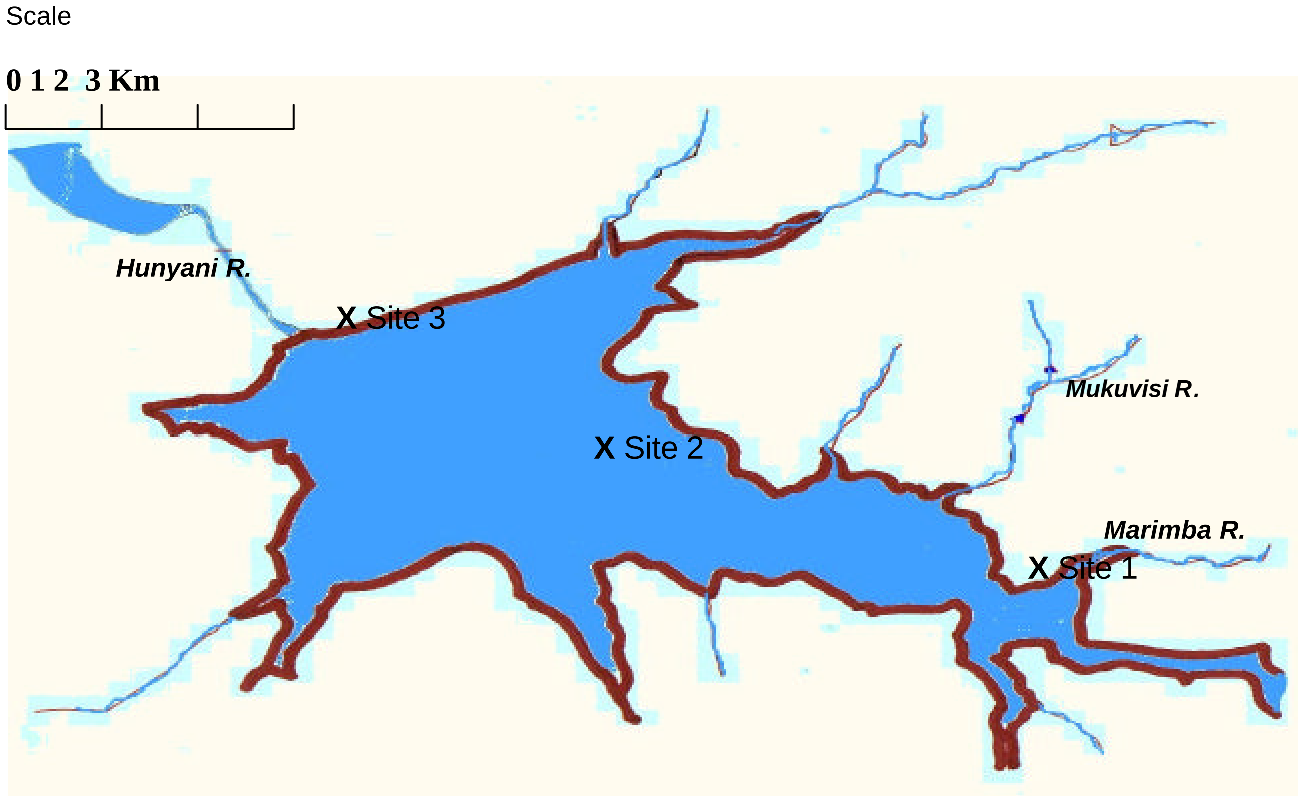

Figure 2: Map showing sampling sites in Lake Chivero.

Results

The primary production levels in the lake were generally high. The highest recorded level was 25.4 mgl^{-1} (6 350 g C m^{2} d^{-1}) whilst the lowest value was 1.1 mgl^{-1}, (275 g C m^{2} d^{-1}). These values were recorded at site 1, time 1,1m and at site 3, time 1, and 4m respectively. On average, the highest productivity levels recorded were at 0.5 m whilst high productivity was within the first three meters of the water column and from three metres downwards, the productivity levels generally decreased with depth (figure 3). The analysis of variance test showed that there was no significant difference in primary production between the sites in the lake (P = 0. 828 at 5% sig. level).

The primary production values were measured in mgl^{-1} and were computed to g C m^{2} h^{-1} in order for them to be comparable to previous studies. The compared primary production values are shown in Table 3. After the computation, a two-tailed t-test, assuming equal variance, (since variance data were not available for the Robarts study) was done to compare the primary production mean of site 2 with that obtained by Robarts in 1979 during the same period (Mid-March to April) at site M-4. This is because site M-4 is the approximate mid-point of the lake and this is where site 2 of the present study was also located. The t- test showed that there was a significant difference (P= 0.0089 at 0.05 sig. level) between the primary production levels of the present study at site 2 and that during the Robarts study. The computed average values at each site are shown in Table 4.

Discussion

The significantly higher primary production levels as compared to Robarts' (1979) showed that there has been a significant increase in the primary production of the lake (approximately eight times higher: 445 to 3370 g C m2 h^{-1}) in the past 24 years. Lake Chivero at the time of sampling, was more productive and possibly, more eutrophic than before, particularly when compared to other tropical impoundments (Robarts, 1979). Lakes Malawi and Victoria for instance, had primary production levels that ranged from 1.14 to 234 C m^{-2} h^{-1} and 9.8 to 31 C m^{-2} h^{-1} (Hecky & Kling, 1987; Mugidde, 1993). However, it is important to note that these lakes are deeper and clearer than Lake Chivero. This could possibly explain the lower productivity levels, as the nutrients are not as concentrated as in a shallow lake like in Lake Chivero. Compared to Hartbeesport dam, a water body that is more physiognomally similar to Lake Chivero, and where primary production levels of up to 185 C m^{-2} h^{-1} have been recorded (Robarts, 1984), the primary production levels in Lake Chivero were very high suggesting that Lake Chivero's eutrophication level has increased. The solution of reducing the productivity levels lies in the control of the growing conditions of phytoplankton, especially the nutrient levels.

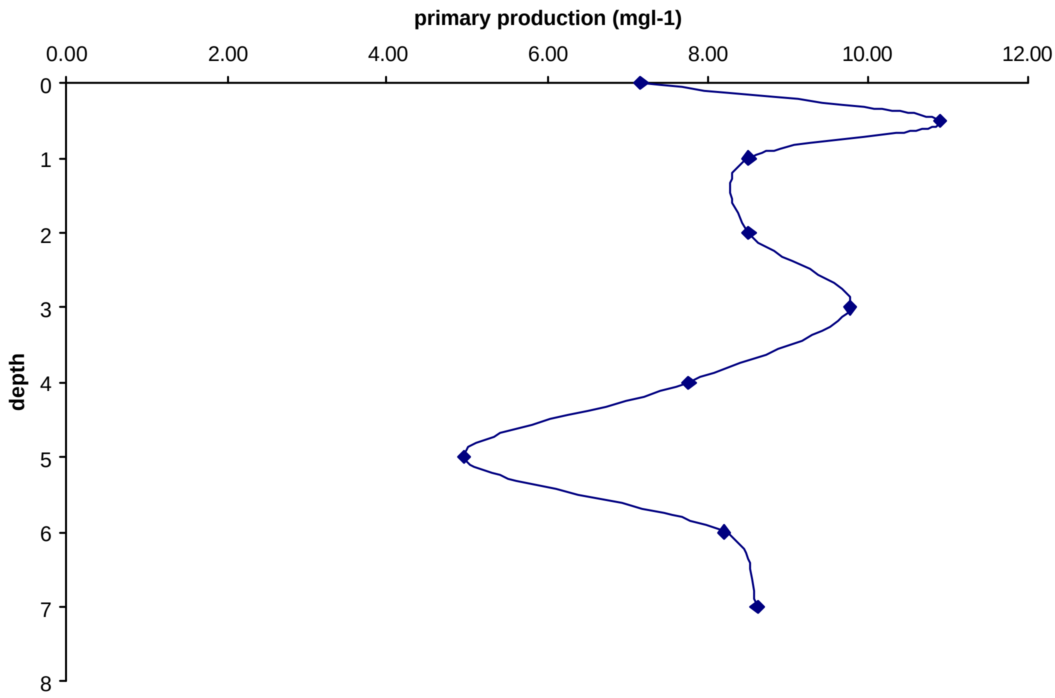

Figure 3. Average primary production levels at each depth in the lake. Table 3. Primary production levels in Lake Chivero: comparison between the Robarts (1979) and the present study.

Date	Primary production (g C m^{-2} h^{-1})	**Average**
23/03/76	652.7	
07/04/76	248.1	
21/04/76	461.3	**454**
20/03/03	2 860	
04/04/03	4 569	
17/04/03	2 60	**3 370**
T-Stat		**4.76**
P value		**0.0089**

Table 4. Average primary production levels in g C m^{-2} h^{-1} (first column) converted to mg l^{-1} h^{-1} (second column).

Date	Site	Primary production (g C m^{-2} h^{-1})	Primary production (mgl^{-1})
1	1	1 968.75	6.4
1	2	2 860.00	9.93
1	3	3 183.66	8.52
2	1	827.00	8.52
2	2	4 569.00	9.78
2	3	2 680.72	7.77
3	1	3 130.41	7.36
3	2	557.61	8.2
3	3	1 228.29	7.16
Average		2 333.88	8.18

Recent studies (Nhapi, 2004; Mukwashi, 2001; Magadza, 2001) indicate that there are now various routes of nutrient entry into the lake, comprising waste water effluent as well as storm water and sewage transport breaches. A nutrient management strategy now has to take a multifaceted dimension, including socio-ecological issues arising form urban poverty that has lead to increased urban agriculture (Magadza, 2003).

References

Burke N. A. and Thornton J.A., 1982. The creation of Lake McIlwaine: history and design. In: Thornton, J.A. and Nduku W.K. Dr. W., eds. *Lake McIlwaine. The Eutrophication and Recovery of a Tropical Man-made Lake*.. Junk Publishers, The Hague, The Netherlands.

Hecky R.E. and Kling H. J., 1987. The phytoplankton and protozooplankton in the eutrophic zone of Lake Tanganyika: Species composition, biomass, chlorophyll 'a' content and spatiotemporal distribution. *Limnology and Oceanography* 26, 548-564.

Magadza C. H. D., 1994. Evaluation of eutrophication control in Lake Chivero, Zimbabwe, by multivariate analysis of zooplankton. *Hydrobiologia* 272**,** 277-292.

Magadza C.H.D., 2001. Immerging issues on water resources management in Africa. In Jansky L., Nakayama M and J.H. Uitto (Rdt). *Lakes and Reservoirs as International Water Systems.* United Nations University Publications 2001.

Magadza C.H. D., 2003. Lake Chivero: A Management Case Stud**y**. *Lakes and Reservoirs: Research and Management* 28, 69-81.

Marshall B. E., 1995. Changes in the benthic fauna of Lake Chivero, Zimbabwe over thirty years. *Southern Africa Journal of Aquatic Sciences* 21, 22-28.

Mugidde R., 1993. The increase in phytoplankton activity in Lake Victoria (Uganda). *Verhandlungen Internationale Vereinigung fur limnologie* 25, 846-849.

Mukwashi M., 2001. *Characterisation of stream outflow in the city of Harare*. BSc thesis, University of Zimbabwe, Harare

Nhapi I., 2004. *Options for wastewater management in Harare, Zimbabwe*. D. Phil Thesis, Wageningen University, The Netherlands

Robarts R. D., 1979. Underwater light penetration, chlorophyll a and primary production in a tropical African lake (Lake McIlwaine, Rhodesia). *Archive of Hydrobiology* 86, 423-444.

Robarts R. D., 1982. Primary production of Lake Chivero. *In*. Thornton J.A. and Nduku W. K.. Dr. W. eds. *Lake McIlwaine. The Eutrophication and Recovery of a Tropical Man-made Lake*. Junk Publishers, The Hague, The Netherlands.

Robarts R. D., 1984. Factors controlling primary production in a hypereutrophic lake (Hartbeesport Dam, South Africa). *Journal of Plankton Research* 6, 91-105.

Thornton J.A., 1982. The algal community. *In* Thornton J.A. and Nduku W. K., Dr. W., eds. *Lake McIlwaine. The Eutrophication and Recovery of a Tropical Man-made Lake*. Junk Publishers, The Hague, The Netherlands

The occurrence of microcystin in Lake Chivero, Zimbabwe

Ndebele Mzime Regina and Christopher H.D. Magadza

[1]Department of Biological Sciences, University of Zimbabwe, PO MP167, Mount Pleasant, Harare, Zimbabwe
Email:ndebele@science.uz.ac.zw

Abstract

Lake Chivero is a eutrophic reservoir built to supply water to the city of Harare, Zimbabwe. Blooms of blue-green algae have been a problem for many years and there has been concern about the toxins produced by *Microcystis* spp. The level of the toxin microcystin produced in cultures of *Microcystis* from Lake Chivero was investigated during the period of March 2003 to May 2003. Microcystin was found in algal cells cultured from the lake water in concentrations ranging from 18.02-22.48 $\mu g\ l^{-1}$ with a mean of 19.86 $\mu g\ l^{-1}$. These concentrations are the highest so far recorded from the lake and raises concern over the possible effects of the toxin on the health of people drinking the water. There is a need to control eutrophication and reduce algal blooms to prevent their potentially detrimental effects.

Key words: health, Lake Chivero, microcystin

Introduction

Lake Chivero (formerly Lake McIlwaine) is a reservoir created on the Manyame River to supply water to the city of Harare in Zimbabwe. It has a capacity of 247 x 10^6 m^3 when full; its mean depth is 9m (maximum = 27 m) and it is 10 km long. Sewage effluent from Harare is discharged into the lake and it became eutrophic in the early 1960s (Munro, 1966). Water quality improved in the 1970s after sewage effluent was used for irrigation before being discharged into the lake (Thornton, 1982) but these measures failed in the 1980s and it is again highly eutrophic (Magadza, 2003; Nhapi, 2004).

The most important consequence of the eutrophication of Lake Chivero is the dense blooms of blue-green algae that cause problems with water purification as, well as the presence of *Eichhornia crassipes,* which now covers a large portion of the lake surface and several kilometres of the reaches of the lake's inflowing streams. Blue-green algae are known to produce toxins and algal blooms and have caused livestock deaths in various countries (WHO, 2001). Algal toxins, such as microcystin (produced by *Microcystis* spp.) may also be a threat to public health because they are hepatotoxic, carcinogenic and teratotoxic in both humans and aquatic organisms (Bourke & Hawes 1983; Lawton *et al*., 1994; Falconer, 1999) and may also induce sterility (Irvine *et al*., 2003).

Circumstantial evidence linked a seasonal increase of gastroenteritis cases in Harare to algal blooms in Lake Chivero (Zilberg 1966; Marshall, 1991) while algal toxins may have contributed to an extensive fish kill in 1996. The only fish species killed was the cichlid *Oreochromis macrochir*, which feeds on blue-green algae. Many of the dead fish had enlarged livers, a possible symptom of algal poisoning (Moyo, 1997). In Harare, cases of gastrointestinal infections and liver cancer have risen over 300% and 400% respectively in the past decade; between 1991 and 2001. Documented cases of gastroenteritis rose from <100 to 300 per thousand while liver cancer rose from 30 to 130 per thousand (Zimbabwe National Health Profile (Z.N.H.P.), 1990-2001; Harare City Department of Health 1991-2001) (Figure 1). The extent to which this is linked to algal toxins is unknown but Johansson and Olsson (1998) found up to 13.9 $\mu g\ l^{-1}$ of microcystin in Lake Chivero and also detected it in the city's tap water. This is a matter of concern because microcystins have been linked to hepatocellular carcinomas (HCC), a major type of primary liver cancer (Ueno *et al.,* 1996; Welker *et al*., 2004). This study is a report on limnological conditions and the concentration of blue-green algae and microcystins in Lake Chivero during the period of March to April of 2003.

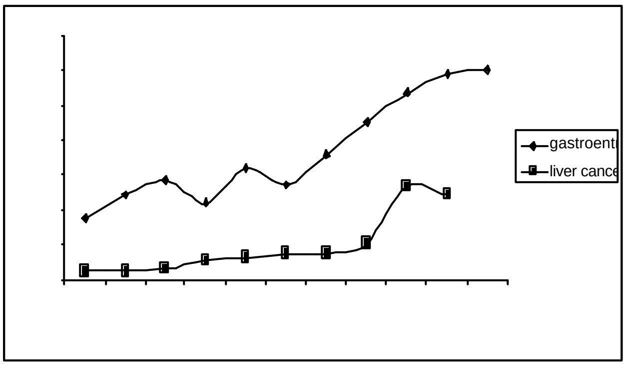

Figure 1. Prevalence of gastro-enteritis and liver cancer per thousand people in Harare. (Source: Zimbabwe National Health Profile (Z.N.H.P.), 1990-2001; Harare City Department of Health 1991-2001).

Material and methods

Samples were collected every fortnight from three stations located along the length of the lake in March and April 2003 (figure 2). Phytoplankton samples integrated from 0 to 5m were taken with a 5m hose at the same stations. From these samples; cultures of the phytoplankton were inoculated into Bold's basal medium (Bischoff & Bold, 1963) and put under a non-heat emitting fluorescent light for two weeks. *Microcystis aeruginosa* culturing was done by firstly sub-culturing the isolated samples of *Microcystis* sp. (over a two-week period) using the same type of stock solution. To obtain purified *Microcystis aeruginosa* cultures, that is, the second *Microcystis* sp. cultures were inoculated into a WC medium, also Woods Hole medium (Guillard & Lorenzen, 1972) and cultured over another two week period.

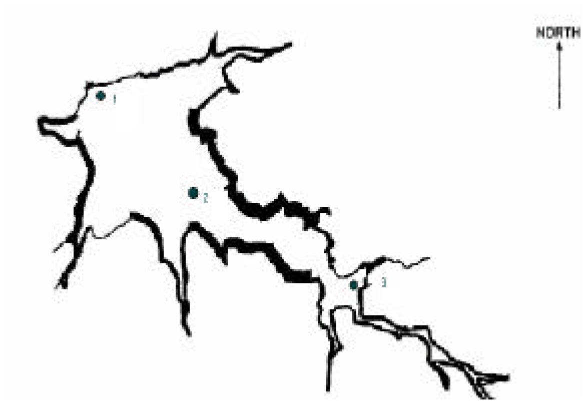

Figure 2.The sampling stations in Lake Chivero.

They were then used in the extraction of the toxin, microcystin-LR through the use of alcohol as a solvent (chloroform in atropine after Trease & Evans, 1978). From there, the determination of the toxin's concentration was done by high performance liquid chromatography (Lawton *et al*., 1994). Dissolved oxygen, temperature, conductivity, chloride and pH were measured on site using a Horiba U-23 multi-meter (Horiba Instruments Ltd) every two weeks at the three sites. The measurements were taken to 7m depth at one-metre intervals, but at station 1, which was at the mouth of the Marimba River, the maximum depth was 4m because of the shallow water. A Nansen water sampler was used to collect water for the measurement of total nitrogen and total phosphorous, which was done in the laboratory with a DR/2010 Spectrophotometer (HACH Company) kit. A multiple analysis of variance (MANOVA) and a Scheffe method of multiple comparisons of treatment effects (site, depth and time) were used to locate areas of significant changes in the physico-chemical parameters

Results

The average concentration of microcystins in the cultures was 19.86 µg/l, with the highest value coming from the culture developed from station 1 and the lowest at site 2 (Table 1). There was no significant difference in the average concentration of microcystin among the sites (analysis of variance, $p > 0.05$).

Table 1. The mean concentration (± standard deviation) of microcystin (µg l^{-1}) in cultures taken from three sites in Lake Chivero, March-April 2003.

Station	Concentration (µg l^{-1})
1	22.48 ± 2.5
2	18.02 ± 1.6
3	19.08 ± 3.0
Mean	19.86

Temperature ranged between 22.6°C (site 1, 4m, time 3 and site 3, 7m, time 3) and 28.2°C (site 3, 0m, time 1). On average, the temperature decreased with depth at a rate of about 0.26°C per meter. The pH values ranged between 6.66 (site 2, 2m, time 3) and 8.75 (site 3, 2m, time 1). The pH values recorded were within the acceptable limits of 6 to 9 (Zimbabwe Water and Waste Water Standards, 2001) and together with the DO values generally decreased with depth. The DO and conductivity values ranged between 2.83 to 10.5 mgl^{-1} and 94.5 to 571 mSm^{-1} respectively. However, with the exception of the highest value of 571 mSm^{-1} at site 1, 1m, the conductivity levels were within the acceptable limit of 500 mSm^{-1}. The chloride depth profile did not follow a particular trend and the values range was within 16 to 191 mgl^{-1}.

Table 2. The mean concentration of physico-chemical values taken from three stations in Lake Chivero, March-April 2003.

	Date	Temp (°C)	DO (mgl^{-1})	pH	Cond (mSm^{-1})	P (mgl^{-1})	N (mgl^{-1})	Chloride (mgl^{-1})
Station 1								
	28/03/2003	23.73	7.63	8.07	143.32	2.42	3.2	164
	01/04/2003	23.27	6.71	6.76	142	2.55	2.64	100.77
	15/04/2003	23.35	5.56	6.46	367.83	2.99	2.68	75.43
Station 2								
	28/03/2003	26.66	6.46	7.80	148	2.94	2.31	174.78
	01/04/2003	23.56	7.51	6.93	264.44	1.98	2.94	75.80
	15/04/2003	23.37	4.07	6.66	149.89	2.73	2.26	64.67
Station 3								
	28/03/2003	26.22	7.07	8.16	147.56	2.98	1.41	155.33
	01/04/2003	23.89	8.69	7.34	226.69	2.74	1.35	44.09
	15/04/2003	23.87	5.58	6.36	325.56	2.67	2.02	71.03

The concentration of total nitrogen values ranged between from 0.3 to 8.4 mgl^{-1} whilst the total phosphorous values' range was 1.01 to 5.01 mgl^{-1}. Most of the phosphorous values were above 2 mg/l whilst more than half of the nitrogen values were above the same reading. The phosphorous levels in the lake were high and above the acceptable limit according to the Zimbabwe water and waste water standards (2001), whilst the total nitrogen levels were below the recommended value of 10 mgl^{-1}. The nitrogen to phosphorous ratios were low, ranging from 1:0.1 (site 3 at 6 and 7m, time 1) to 1: 3.5 (site 2, 7 meters, time 2) with most of the N: P values being below 1:1. The averages of the physico-chemical data are shown in Table 2.

The MANOVA showed that there were no significant differences among the sites, the depth and the time of sampling for the temperature, pH, conductivity, chloride, total P and total N with the exception of time comparisons of temperature, DO and chloride; site comparisons of DO, pH and chloride and depth comparisons of temperature and DO (Table 2). The Scheffe method of multiple comparisons reflected that overall; there were no statistically significant differences in all the physico-chemical parameters between the three sites, the three sampling periods and the different depths within the water column (Table 3).

Table 3. The comparison of the physico-chemical constituents of Lake Chivero at each depth, site and time of sampling. Figures represent probability levels (P values) from the MANOVA and Scheffe analysis, statistically significant values are highlighted in bold (P < 0.05).

Parameter	Temp	DO	pH	Cond	Cl	Tot P	Tot N
Date	**0.0**	**0.02**	0.87	0.16	**0.0**	0.99	0.70
Site	0.5	**0.0**	**0.0**	0.34	**0.05**	0.96	0.45
Depth	**0.0**	**0.0**	0.7	0.94	0.54	0.37	0.34
Scheffe Sig.	0.5	0.21	0.26	0.49	0.28	0.09	0.65

Discussion

The fact that the microcystin levels recorded in this study exceeded the recommended levels of 0.01 µg/l and 1.0 µg/l for drinking (tap) water and lake water respectively (Ueno *et al.*, 1996; WHO, 2001) and were also higher than the maximum of 13.9 µg l^{-1} recorded in an earlier study (Johansson & Olsson, 1998) is a grave cause for concern. This is because the values exceed safety recommendations by factors of approximately 2000 and 20 times respectively. These values are the highest microcystin concentration levels recorded to date from Lake Chivero. It can be suggested that the high microcystin concentration levels found in the present study can be a reflection of increased toxin production with increase in algal biomass and standing crop in the lake (algal blooms). It can also be a possibility that such toxin production is a result of increased eutrophication due to pollution.

The thermal stratification and pollution of Lake Chivero has been discussed by several workers such as Nduku (1978), Thornton and Nduku, (1982) who have demonstrated that the lake was polymictic. In the present study this would be confirmed by the breakdown of stratification at the time of sampling. The evidence of this lack of detectable thermal stratification was the presence of localised algal blooms that occurred with the temporary upwelling of nutrients at the time of sampling. The stratification observed here is that due to dissolved material, especially chloride salts. This type of stratification in the lake is a natural feature of most shallow, saline lakes such as Lake Naivasha in Kenya (Melack, 1979) but its presence in Lake Chivero reflected the high ionic input, most probably from sewage effluent. Magadza (2003) detected a thermohaline stratification at station 3 during September in 2001. Lake Chivero originates from granites rocks which yield low conductivity water of the order of 0.25 to 0.5 mSm^{-1}. The high ion content is therefore of artificial origin, when these data are compared to the conductivity of 5.5 mSm^{-1} in the upstream reservoirs, Cleveland Dam (Magadza, 2003). Marshall and Falconer (1973) found that conductivity did not vary with depth in Lake Chivero. This was consistent with the present study. In comparison, conductivity and chloride levels of Lake Chilwa during the dry years were very high (Magadza, 1994) and comparable to Lake Chivero. This shows the high pollution in the latter lake. Lake Chilwa is a saline, endorheic lake which shows periodic desiccation.

The high concentration of microcystin in the lake is a matter of concern and appears to be a potential threat to the health of the citizens of Harare as it exposes the residents to great risk of contracting gastroenteritis and liver cancer as well as other toxin related diseases. It has been suggested that the microcystin strain in Lake Chivero may not be poisonous (Moyo, 1998). This is because livestock and direct human deaths have not been reported in Zimbabwe but in Hartbeesport Dam (South Africa), Alexandria dam (Austria), Haimen city, Jian-Su province, Fusui county, Guangxi province (China) and the Whitewater Lake (Canada) cattle, sheep, bird kills and increased incidences of liver cancer have been linked to algal toxin contact, which usually coincide with algal blooms Park *et al*., 2001; Ueno *et al*., 1996; Lepisto *et al*., 1992, 1994). The toxicity testing was beyond the time scope of the present project but the simultaneous increase of toxin related diseases presented in this study over the past decade as well as the eutrophication levels due to pollution in the lake may mean increased microcystin levels can be linked to the diseases. In Harare, only one study (Zilberg, 1966) presented evidence of such a link between Lake Chivero toxins and cases of gastroenteritis in infants.

The results of this study show that it is imperative to ascertain the proportion of toxic microcystins in the waters of Lake Chivero. There is an urgent need to control the levels of microcystin in the lake and in the drinking water supplied to Harare especially since microcystin has been detected in the drinking water of the city (Johansson & Olsson, 1998). The removal of microcystins can be done during the water purification process, by extracting the toxin from the water through the use of solvents such as alcohol, chloroform and atropine (Trease & Evans, 1978). Also, since the toxins are biodegradable, a sand filtration can be used to remove any dissolved toxins from the water during water purification (Johansson & Olsson, 1998). Microcystin can be eliminated through the use of activated carbon as well as ozone and potassium permanganate but this method has the disadvantage of corroding the water pipelines (Lepisto *et al.,* 1992). Another measure of eliminating microcystins in Lake Chivero would be through the elimination of algal cells using algaecides but this would result in the accumulation of the toxin in the water as the cells lyse and release the toxin into the water. The most effective measure of eliminating microcystins and other toxins would be the control of algal blooms in the lake through the reduction of nutrient supply to the lake. This would require a major capital investment in sewage treatment works and the disposal of sewage effluent, which may not be possible given the current economic situation in Zimbabwe where the local authorities responsible for treating sewage lack the resources to do so. Reduction of pollution and eutrophication would ensure reduced and possibly relatively safe levels of algal blooms and toxins.

However, results from self purification studies of the Mukuvisi River, a tributary of Lake Chivero, offer possibilities of use of constructed wetlands for tertiary treatment of processed waster waters as well as interception of urban surface run off.

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References

Bischoff H. and Bold H.C., 1963. *Phycology studies. IV. Some soil algae from enchanted rock and related algal species*. 95 pp.University of Texas Publication, Texas.

Bourke A. T. C. and Hawes R. B., 1983. Freshwater cyanobacteria (blue-green algae) and human health. *Medical Journal of Australia* 11, 491-492.

Falconer, I. R., 1999. An overview of problems caused by toxic blue-green algae (Cyanobacteria) in drinking and recreational water. *Environ Toxicol.* 14, 5-12.

Irvine, K.N., T.P. Murphy, T. Tang, L.M. Romano, and D. Walters. 2003. Spatial and temporal trends in chlorophyll a, nutrients, and sediment in relation to avian mortality, Whitewater Lake, Manitoba, Canada. *In.* M. Munwar ed. *Sediment Quality Assessment and Management: Insight and Progress.* 271-286 pp. Ecovision World Monograph Series, Aquatic Ecosystem Health and Management Society.

Guillard R.R.L. and Lorenzen C.J., 1972. Yellow-green algae with chlorophyllide c. *Journal of Phycology* 8, 10–14.

.Johansson S. and Olsson M., 1998. *Investigations of toxic algal blooms in a drinking water reservoir of Harare, Zimbabwe*. Technical Report, University of Lund.

Lawton L. A., Edwards C. and Codd G.A., 1994. Extraction and High Performance Liquid Chromatographic method for the determination of microcystins in raw and treated waters. *Analyst* 119, 525-1530.

Lepistö L., Lahti K., Färdig M. and Neimi J., 1992. *Removal efficiency of algae and especially cyanobacteria in different waterworks.* National Board of Waters and Environment, Finland Report.

Magadza C.H.D., 1994. Evaluation of eutrophication control in Lake Chivero, Zimbabwe, by multivariate analysis of zooplankton. *Hydrobiologia* 272, 277-292.

Magadza C.H.D., 2003. Lake Chivero: A management case study. *Lakes & Reservoirs: Research and Management* 8, 69–81

Marshall B. E. and Falconer A. C., 1973. Eutrophication of a tropical African impoundment (Lake McIlwaine, Rhodesia). *Hydrobiologia* 43, 109-123.

Marshall B.E., 1991. Toxic cyanobacteria in Lake Chivero: A possible health hazard. *Transactions of Zimbabwe Scientific Association* 65,16-19.

Melack J.M., 1979. Photosynthetic rates in four tropical African fresh waters. *Freshwater Biology* 9, 555-571.

Moyo N.A.G., 1997. The causes of massive fish deaths in Lake Chivero. In: *Lake Chivero A Polluted Lake*. (ed Moyo N.A.G.). University of Zimbabwe Publications, Harare, Zimbabwe.

Munro J. L., 1966. A Limnological survey of lake McIlwaine, Rhodesia. *Hydrobiologie*: 28, 181-308.

Nduku W. K., 1978. The thermocline stability in Lake McIlwaine, Rhodesia. Paper presented at "*The Limnological Society of Southern African Congress*", Pietermaritzburg, Republic of South Africa.

Nhapi I., 2004. *Options for wastewater management in Harare, Zimbabwe*. D. Phil Thesis, Wageningen University, The Netherlands

Thornton J.A. 1982. The algal community. *In.* Thornton J.A. and Nduku W. K. eds.. *Lake Mcllwaine*. *The Eutrophication and Recovery of a Tropical Man-made Lake.* Dr. W. Junk Publishers, The Hague, The Netherlands

Thornton J.A. and Nduku W. K. 1982. Water chemistry and nutrient budgets. *In.* Thornton J.A. and Nduku W. K. eds. *Lake Mcllwaine. The Eutrophication and Recovery of a Tropical Man-made Lake.* Dr. W. Junk Publishers, The Hague, The Netherlands.

Trease E.G. and Evans W.C., 1978. *Pharmacognosy*, *11th ed*.. Macmillan Publishing Co., Inc. New York.

Ueno Y., Nagat S., Tsutsumi T., Hasegawa A., Watanabe M.F., Park F. D., Chen G. and Yu S.Z., 1996. Detection of microcystins, a blue-green algal hepatotoxin, in drinking water sampled in Haimen and Fusui, endemic areas of primary liver cancer in China, by highly sensitive immunoassay. *Carcinogenesis* 17, 1317-1321.

Welker M., Chorus I. and Fastner J., 2004. *Occurrence of cyanobacterial toxins (microcystins) in surface water of rural Bangladesh*: pilot study: Federal Environmental Agency, Germany, WHO/SDE/WSH/04.06 © World Health Organization

World Health Organisation. 2001. *Guidelines for drinking-water quality,* 2nd ed. (Addendum to Vol. 1.) Recommendations. Geneva. 13-14 pp

Zilberg B., 1966. Gastro-enteritis in Salisbury European Children – A Five-Year Study. *The Central African Journal of Medicine* 12, 164-168.

Zimbabwe Government., 2001. *Water (Effluent and Waste Water standards Regulations)* Statutory Instrument of 2000 [CAP: 20: 24].

Zimbabwe National Health Profile (ZNHP), 1990-2000. *Ministry of Health and Child Welfare, Epidemiology and Disease Control.* National health information and surveillance unit, Harare, Zimbabwe.

Effects of physical mixing on the environment of satellite lakes and dams of lake Victoria, Kenya

Osumo Wilfred [1], Enock Wakwabi[1], Lewis Sitoki[1], Fredrick Guya[1] , Tsuma Jembe[1] and Zadock Ogutu[2]

[1]Kenya Marine & Fisheries research Institute, P.O Box 1881, Kisumu, Kenya.
E-mail: wosumo@yahoo.com
[2]Interuniversity Council for Eastern Africa, P.O Box 7110, Kampala, Uganda.

Abstract

In the Lake Victoria catchment are found several small lakes and dams. The satellite lakes and dams are important aquifers and buffer zones for the lake. Besides the water bodies are important water sources both for livestock and domestic use. A study conducted in the water bodies between January 2003 and December 2004 showed that effects of eutrophication are wide spread in the lakes and dams. An investigation of primary productivity and effects of physical mixing and material transport was done. In Lake Sare, Lake Victoria waters are transported to the satellite lake during both day and night through Goye causeway that connects the two. The water bodies have higher rates of primary productivity than Lake Victoria. Spatial extent of oxygen depletion due to respiration of organic matter in the water column is lower than the rates of primary productivity. Variability of oxygen depletion is dependent on local thermal and hydrodynamic processes that appear to be controlled by seasonal and short-term wind regimes. Advective transport and convective mixing augmented by wind effects control distribution of nutrients. An analysis of sediment grab samples showed that nutrient fluxes and loading is primarily external rather than from sediments. From the chemical parameters that were measured, soluble reactive phosphorus concentrations ranged from 4.44 to 35.28, Nitrate/nitrite ranged from 4.69 to 335.88 µg/l, ammonia from 29.18 µg/l to 473.14 µg/l and silica from 5.01µg/l to 50.06 µg/l. Fish catches particularly indigenous fish species like *Schilbe intermedius, Labeo Victorianus, Protopterus aethipicus and Clarius gariepinus* depicted stunted growth possibly due to heightened eutrophication and reduced dissolved oxygen levels due to reduced mixing on the water/air interface as a result of macrophyte cover.

Key words : dams, eutrophication, satellite lakes

Introduction

Lake Victoria is the second largest (surface area, 68, 800 km^2) freshwater lake in the world. It lies across the equator between 0^{o}20'N-3^{o}0'S and 3^{o}0'S and 34^{o}53'E at an altitude of 1135 m above sea level (Crul, 1992). It is shared by Kenya (4 100 km^2), Uganda (31 00 km^2) and Tanzania (33 700 km^2). The lake contains numerous islands (Welcomme, 1972) and a catchment of about 192 200 km^2.

In the Kenyan sector of the lake's catchment are located several small lakes and dams. These water bodies include both natural freshwater wetlands and constructed/transformed water supply dams. Some of the studied dams include Futro, Maranda, Masawa, Mwer, Ochilo, Ochot, Ugege, Uranga among others. Most of the dams were communally constructed mostly in the 1950s and 1960s to provide water supply for domestic use and livestock. Others were formed naturally by rivers and under ground water sources that filled natural depressions. Areas in which they are located contain hydric soil with a high water table subject to water logging which is a common indicator of wetland hydrology. The most notable of the water bodies include Lake Sare (500 ha), Lake Kanyaboli (1050 ha), and Lake Namboyo (200 ha) located at 00^{o}02'36''S 034^{o}03'32''E, 00^{o}04'30''N 034^{o}09'36''E and 00^{o}00'23''N 034^{o}05'09''E respectively. These lakes are located within the Yala swamp which covers an area of 17 km^2. They are vital life support and natural assets to communities that live in their vicinities. They are vital sources of water for domestic use and livestock and fishing.

Worldwide observations indicate that water quality continues to deteriorate everyday (Falkenmark, 2000). The situation is even worse in third worlds countries. Populations surrounding the water bodies are some of the poorest. A rapid increase in human and livestock population, open access system and weak legislation have led to a situation in which the satellite lakes and dams are either fully or heavily exploited or over exploited. Water abstraction levels have recently reached an asymptote level and have thus led to high primary productivity. Conflicts over ownership of watering points both for livestock and domestic are a common feature in some of the water bodies. These are manifestations of over-exploitation of the water resources. The effects on the ecosystems have been exacerbated by land-based resource alterations. These have had negative effects on the livelihoods of the surrounding communities and a risk in food security of the poorest. The changes taking place in the ecosystems are likely to lead to reductions in maximum sustainable yields expected from the water bodies, modifications of the resource species composition, health and diversity, increases in the ecosystem instability and variability and a reduction in water quality and safety.

Fallow areas surrounding the water bodies are covered by woody vegetation mainly savanna woodland made of Acacia, Albizzia and Butyrospermum. Others consist of herbaceous vegetation (*Cymbopogon, Hyparrhenia, Londetia* and *Cyperus papyrus*). The areas under natural vegetation are decreasing due to high population pressure and resultant excessive cultivation. The areas are utilized in growing a variety of crops maize, cotton, sisal, tobacco, beans, sugar cane, coffee, sorghum, millet, wheat and root crops mainly cassava with varying levels of fertilizer application containing high levels of Nitrogen, Phosphorus and potassium.

Management of the water bodies has been inadequate or even poor. Though it is generally agreed among the communities on the importance of the water bodies, no serious management measures are in place to control pollution both from run-off from the surrounding farms, direct watering of livestock and washing of house-ware and bathing in the water bodies.

The paper recognizes that, if the water bodies are to be restored to and managed for optimal benefit, there is no alternative to rationalization of the water bodies. This must ensure that the communities that depend on the resources bear the costs of the ecosystems impacts as far as possible. There is also need to ensure that land-based and shore-based activities such as agriculture and washing respond in a similar manner and are held accountable for their impacts on the ecosystem and take appropriate measures to reduce these to the required limits.

Materials and methods

Study area

Sampling for various physical and biological parameters was done at specific sites in several small lakes and dams found in the Kenyan Lake Victoria water catchment (Table 1). Identification of sampling sites was done by a GPS Magellan Model 315 on board a 25-horse power fibre-optic dingy. Sampling was done at specific sites in Lakes Sare, Namboyo and Kanyaboli and in several dams in the Lake Victoria Kenyan water catchment.

Methodology

A portion of water sample collected with a 3 litre van Dorn water sampler was preserved in Lugol's solution and a 2 ml phytoplankton sub-sample was placed in an Utermol sedimentation chamber and left to settle for at least 2 hours. Phytoplankton species were then identified and enumerated using a Zeiss Axioninvert 35 inverted microscope at 400X magnification. Phytoplankton taxa were identified using methods of Huber-Pestalozzi (1968) and Cocquyt *et al.*, (1993). Water temperature and dissolved oxygen were measured with a Hanna HI 9143 oxygen meter, conductivity with a LF 96 meter, pH with a Hanna 8014 pH meter. In Lake Victoria these parameters were determined with a Hydrolab water quality-measuring instrument. Water transparency was determined by a 20 cm diameter *Secch*i disk. Total hardness was determined by EDTA titration with Eriochrome T black and HCL mixture as indicator. Water samples in 500 ml plastic bottles were fixed with acid for laboratory analysis.). Wind speed and direction were recorded with an Anderaa wind speed sensors model 3070 and 2070 respectively. In Lake Victoria, *in situ* current velocities and direction (cms^{-1}, a) were determined with an Anderaa™ minstrommer current meter model SD 4A. Measurements were taken at 1 m depth intervals. The wind velocities were then separated into the four direction components by equaling negative values to the west and south in order to investigate relative contribution of each four-wind direction to wind direction prior to averaging. Surface area and water volumes of the water bodies were calculated from several GPS positions taken on the water bodies' perimeter fence. The water bodies were then stratified and random bottom depths determined by a graduated mast. These were averaged and used to calculate maximum and average depths. The average depths and surface areas were used to determine average water volumes.

Water exchange across the Goye causeway that connects Lake Sare with Lake Victoria was made by taking depth and widths to determine the cross-sectional area of the causeway. Measurements were taken to calculate water exchange (cms^{-1}) using the following formula described in Jeffries and Mills (1990):

$$Dsc=AV$$

where DSc = water discharge (cm^{3}s^{-1}), V= velocity (cms^{-1}), and A= cross-sectional area of the causeway (cm^2).

Results

From the study it was found that most of the water bodies are communally owned. These communites practice no management regimes and the water bodies are openly accessed. The communities living in the vicinity of the ecosystems, majority who are extremely poor, fetch water for domestic use, bath, wash clothes and water their livestock directly on the watering points. Human settlement, livestock rearing, subsistence agriculture, both planned and unplanned residential areas, informal industrial activities such as sand harvesting characterize utilization of areas surrounding the wetlands.

Macrophytes consisting of *Cyperus papyrus, phragmites australis,* have formed thick walls along the perimeter fences of the satellite lakes. The macrophytes have made L. Namboyo virtually inaccessible and impenetrable.

Some water bodies exhibited anoxic ($<$ 1 mgl^{-1} DO) features in the bottom waters while others had near super saturation throughout the water column. In the dams dissolved oxygen ranged from 6.25 mgl^{-1} to 10.25 mgl^{-1} and surface total ions measured as conductivity ranged from 150 µScm^{-1} to 220 µScm^{-1} while in the satellite these ranged from 6.73 mgl^{-1} to 8.95 mgl^{-1} and 98 µScm^{-1} to 169 µScm^{-1} respectively. During the same period in Winam Gulf, surface dissolved oxygen and conductivity ranged from 8.25 mgl^{-1} to 5.74 mgl^{-1} and 158 µScm^{-1} to 98 µScm^{-1} respectively.

Rainy months were between April to June. May recived the highest amount of rainfall an average of 125 mm while December 2004 had the lowest 14mm. January to March received no rainfall. During the rainy months of April to June DO In Winam Gulf becomes depleted in the lower column layers.

In all the water bodies studied there was dissolved oxygen super-saturation in the surface waters possibly a result of the increased primary productivity enhanced by eutrophication. However, at L. Kanyaboli, Kalenjuok and Ochillo dams there were observed anoxic levels (<1.9 mg^{-1}) in the deeper waters especially in areas covered by macrophytes. Soluble reactive phosphorus concentrations ranged from 4.44 to 35.28, Nitrate/nitrite ranged from 4.69 to 335.88 µg/l, ammonia from 29.18 µg/l to 473.14 µg/l and silica from 5.01µg/l to 50.06 µg/l.

Wind measurements indicate a general northeasterly (average 3 cms^{-1} at L. Sare and 6 cms^{-1} at L. Kanyaboli) and a southwesterly pattern (average 4 cms^{-1} at L. Sare and 8 cms^{-1} at L. Kanyaboli) in the morning and afternoon hours, respectively. Westerly winds were highest on a 24 hour-cycle. The results indicate westerly and easterly wind patterns in the morning and afternoon hours respectively. Winds systems do not indicate major hourly shifts. Diel wind speeds were highest between 14.00 and 18.00 hours. In the satellite lakes water movement exhibited an horizontal water movement corresponding to wind direction. At L. Kanyaboli, the largest of the satellite lakes, an average current speed of 4 cms^{-1} and 6 cm^{-1} was recorded in the afternoon and afternoon hours, respectively.

Table 1. Showing some of the small water bodies and their geographical locations, sizes and other features.

Water body	Latitude	Longitude	Maximum depth, m	Average depth, m	Approximate surface area (ha and or x 1000m^2	Approximate water volume X 1000 m^3
L. Sare	00°01’45’’S	034°03’01’’E	5.5	3.5	500 h	
L. Kanyaboli	00°04’30’’N	034°09’36’’E	6	4	1050 ha	
L. Namboyo	00°00’25’’N	034°05’32’’E	11	7	200 ha	
Ochot dam			2.9	1.9	47.780 m^2	90.145
Maranda dam	00°05’42’’S	034o13’35’’E	1.7	1.05	16.674 m^2	17.507
Ochilo dam	00°00’30’’S	034°16’06’’E	1.9	1.2	17.732 m^2	21.279
Uranga dam	00°05’19’’N	034°16’22’’E	2.5	0.6	3.653 m^2	2.192
Masawa dam	00°04’54’’N	034°14’00E	1.5	1.1	4.764 m^2	50.446
Futro dam	00°04’07’’N	034°16’03’’E	1.2	0.7	23.919 m^2	16.673
Kalenyjoouk dam	00°04’31’’N	034°13’05’’E				
Mwer dam	00°07’11’’N	034°10’15’’E	5	1.5	122.051 m^2	186.564
Mauna dam	00°12’31’’N	034°09’21’’E	4	2.3	46.685 m^2	110.693
Yenga dam	00°13’00’’N	034°12’36’’E	4.5	2.1	18.732 m^2	39.337

Concentration of chlorophyll-*a* in various dams found in Lake Victoria basin is shown in table 2. An analysis of biological (chlorophyll, productivity and algal biomass) and chemical parameters (temperature, DO and pH levels) suggests occurrence of intensive biological activity. In the dams chlorophyll-*a* ranged from 6 µgl^{-1} to 300 µgl^{-1} with an average of 42 µgl^{-1}.

Table 2. Chlorophyll-*a* concentration µgl^{-1}. in two major satellite lakes in the Lake Victoria catchment.

Water body	Ecological site	Concentration, µgl^{-1}. during the day	Concentration, µgl^{-1}. at night
Lake Sare	River mouth	15.1	11.1
Lake Sare	mid-lake	24.4	17.8
Lake Kanyaboli	River mouth	27.3	15.1
Lake Kanyaboli	mid-lake	17.4	13.4

Algal species richness in the water bodies varied from 12 to 43 with the highest number recorded at L. Namboyo. Variations in blue green algae were high as opposed to other groups (Table 1). Dinoflagellates were represented mainly by *Strombomonas* sp. Concentrations of algal cells were particularly high at Maranda dam located at Bondo District. Concentrations of up to 5000 cells ml^{-1} (Figure 2) were recorded. In most of the dams algal cells were stunted, small in size and unhealthy. Algal species composition in some selected water bodies is given in Figures

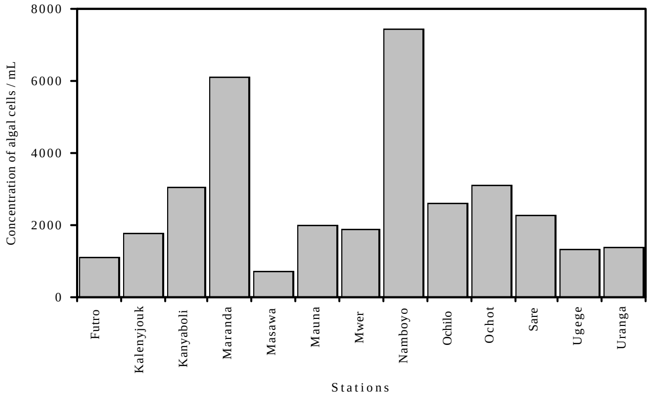

Figure 1. Algal composition of major taxa and concentration cells (cell mL^{-1}) in some satellite lakes and dams of the northern catchment of L. Victoria, August/September 2004.

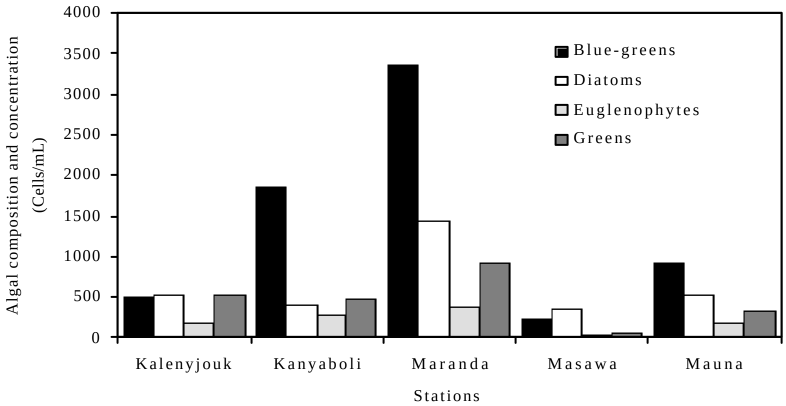

Figure 2. Concentration of algal cells (Cell mL^{-1}) in stations in the satellite lakes and dams of the northern catchment of L. Victoria in the period August-September 2004.

In many stations blue-green algae were found to be predominant in the composition by over 50%. There was a wide range in the concentration of chlorophyll-*a* varying from with an average of (Figure 5). Diel changes in chlorophyll-*a* concentration were minimal indicating that there is no significant difference (Table 2). The chlorophyll-*a* concentration was higher in the dams (6 μgL^{-1} to 300 μgL^{-1}, with an average of 42 μgL^{-1}) compared to those recorded in Lake Victoria (5 - 230 μgL^{-1} with an average of 36) during the same period.

The fish, mainly *Lates niloticus* and *Oreochromis niloticus* caught in Lake Victoria were of comparatively longer lengths than those caught in the satellite lakes and dams (Table 3). Fish caught in the dams particularly endemic fish species like *Schilbe intermedius, Labeo Victorianus, Protopterus aethipicus* and *Clarius gariepinus* depicted stunted growth characteristics. In Lake Victoria fish were observed to mature at much smaller sizes compared to those in Lake Victoria (Table1).

Table 3. Sex and maturity levels of fish in various water bodies in the Lake Victoria catchment and those in Lake Victoria.

Water body	**Fish, n=30**	**Sex**	**Mean length, cm**
Kalenjuok Dam,	Nile Perch *Lates niloticus*	male	28.4
Kalenjuok Dam	Nile Perch *Lates Niloticus*	female	41.5
Kalenjuok Dam	Nile Tilapia, *Oreochromis niloticus*	male	38.6
Lake Victoria, Nyanza Gulf	Nile Perch *Lates niloticus*	Male	32.3
Lake Victoria, Nyanza Gulf	Nile Perch *Lates Niloticus*	female	43.1
Lake Victoria, Nyanza Gulf	Nile Tilapia, *Oreochromis niloticus*	male	41.5

This can be attributed to both natural and human induced factors. The water bodies in the lake's water catchment are mainly re-charged by rainfall and seasonal rivers..Evaporation in the region (located close to the equator) coupled with the shallow nature of the water bodies and heightened eutrophication may have resulted in deterioration of water quality thus rendering the ecosystems inhabitable especially in the epilimnetic waters.

Flow rates across the Goye causeway were between 0.3 cms^{-1} and 40.2 cms^{-1} with a maximum of 120 cms^{-1}. The latter was observed to flow in the direction of the lake. Thus water exchange across the causeway with a cross-sectional area of approximately 533 000 cm^2 was approximately between 15900 cm^3s^{-1} and 2 142 660 cm^3s^{-1} with a maximum o f 63 960 000 cm^3s^{-1}. Swift flow rates were recorded especially in the afternoon hours. The direction of flow was observed to depend on wind direction and differences in water level between the lake and Lake Victoria. An increase in water level in Lake Victoria triggered a flow into Lake Sare and vice-versa.

Besides run-off rainwater contributes significantly to eutrophication of the water bodies. Studies carried out by the Kenya- Belgium joint project in the early 1990s revealed that rainwater contributes an appreciable amount of nutrients in the Lake Victoria basin (table 4)

Table 4. Mean chemical composition of rainwater onto Lake Victoria (Kenya). N=9 (values in parentheses indicate the range)

Parameter	pH	Conductivity µScm^{-1}	Total alkalinity $CaCO_3$ mgl^{-1}	Total hardness $CaCO_3$ mgl^{-1}	NO_3-N µg NL^{-1}	SRP µgPL^{-1}	Reactive sil;ica (Rsi) mgl^{-1}
Constituent	(5.6-7.1) 6.58	(6.29-31.5) 17.9425	(5-24) 14.5	(5-17) 11	(0.607-5) 170.2	(6.3-50.3) 9.064	(0-2.5) 0.944

Discussion

The principal seasons in Lake Victoria and its catchment consist of dry and rainy period. (Ogalo, 1981). Most cooling and mixing take place in the dry season and maximum stratification during the rainy season. Most influential factors are rainy and wind as opposed to the earth's rotation (Lewis, 1987).

Wind fields play an important role in the hydrodynamic process. In the Lake Victoria water catchment, these winds blow with considerable force especially in the afternoon hours.It creates surface and internal waves in the water, which result in currents and water movements, transport of dissolved substances, mixing of the water column and suspension of bottom sediments.The prevailing winds tend to alternate between morning ad afternoon hours possibly influenced by diurnal heating and cooling of the land surface.

In almost all the water bodies studied, blue-green algae were dominant by over 50%. This is thought to be a direct result of direct supply of nutrients from agricultural lands that surround the water bodies. Livestock that water the water bodies directly deposit high amounts of waste that is fodder for the rapid proliferation of the phytoplankton. The situation is compounded by the shallow nature of the water bodies and the constant perturbations from winds, human and livestock causing re-suspension of nutrients from the bottom sediments. High algal proportions as was recorded in the water bodies can produce and release sufficient toxins causing deterioration in both water quality and safety.

The spatial variability of the oxygen depletion ($\leq$ 1 mgl^{-1}) that some water bodies exhibited is partly due to respiration due to decomposing organic matter and the nature of the soils.) The hydric soil of wetlands are saturated, flooded and tend to develop anaerobic conditions in the upper part (Gichuki, in press) Possible consumption of DO by bacteria that decompose organic matter entrapped in the bottom water contributes to this anaerobic condition. Reduced mixing on the water/air interface as a result of macrophyte cover is also a contributory factor in the anoxic condition.

Main causes of pollution in the small water bodies include effluents and run-off containing nutrients such as nitrates and phosphates. These nutrients have stimulated a massive growth of aquatic plants and algae. The vegetation are subsequently clogging watering points our waterways. Decomposition of dead vegetation due use up dissolved oxygen as they decompose and block light to deeper waters. This, in turn, proves very harmful to aquatic organisms particularly fish as it affects the respiration ability or fish and other invertebrates that reside in water. Hence fish caught in most of the water bodies depict stunted growth.

Anthropogenic activities on the terrestrial environment of the water bodies have resulted in increased run-off, sedimentation and nutrient fluxes into the water bodies. Wash-off from plowed fields, charcoal and logging sites and eroded riverbanks when it rains have led to high levels of sedimentation. When the sediments enter the various water bodies lead make fish respiration becomes difficult, plant productivity and water depth become reduced, and aquatic organisms and their environments become suffocated. This is possibly one of reasons for reduced fish catches in the small water bodies.

Lake Kanyaboli, the largest of the three satellite lakes is charged by direct precipitation, backwash waters from Lake Victoria through the Goye causeway and R. Yala. As in the other satellite lakes, winds induce a corresponding horizontal water movement. A change in direction in the wind direction results in a similar response in the water mass Water exchange between the Lake and Lake Victoria seems to be controlled by water level changes between the two. A rise in Lake Victoria water level leads to a backwash flow into Lake Kanyaboli and vice versa.

An annual series of fish deaths were reported in Lake Victoria (Ochumba, 1987) caused by de-oxygenation. Studies to determine the cause of the deaths attributed them to sudden upwelling of deep anoxic waters to the surface thereby suffocating the fish (Ochumba, 1987). Though no fish deaths have been reported in the satellite lakes and dams, the lack of life exhibited by minimal fish catches, stunted growth characteristics, severity of algal blooms and infestation of macrophytes e.g water hyacinth, *Eicchornia crassipes,* (Mart Solms), papyrus, *Cyperus papyrus* that are often known to inhabit extremely polluted and unsafe water sources are common.

In Winam Gulf, which is relatively shallow and comparable to the satellite lakes, water circulation system is responsive to the large-scale monsoon system and the cooling rainy season. Isothermal features and high oxygen levels throughout the water column characterize the water column. This scenario is replicated in the satellite lakes. However in the dams, which are extremely shallow, the surface waters exhibit dissolved oxygen saturation in the surface waters and anoxic tendencies in the deeper waters. This can be attributed to the possible high levels of primary productivity in the surface waters due to algal blooms. However the sub-surface waters get depleted of oxygen due to inhibition by macrophyte cover and high oxygen demand due to respiration of decomposing organic matter.

The underlying reason why most communities who live in the vicinity of the satellite lakes and dams have been slow to adopt effective management regimes is probably because, firstly, it takes time to realize that the water bodies are exhaustive and secondly, because transitions towards new management systems can be costly. For a long time the water bodies were considered inexhaustible and the perception was that controls were not necessary. However slowly and painfully, the communities have realized that this is necessary. This scenario is compounded by a lack of comprehensive environmental policies as well enough and reliable information (Gitonga, 1992; Odera *et al*, 2000; El-Fadel *et al*., 2001)

These ecosystems constitute vital life support and natural assets that should be conserved. Though the water bodies in the Lake Victoria basin are fully exploited, access remains practically open with no or minimal restrictions on water use and conservation. Though human and livestock populations in the Lake Victoria Basin have grown at an unprecedented rate, conservation legislations and management regimes have not been enacted to keep pace with their usage.

Conclusion

The proportion of blue-green algal cells in many water bodies around Lake Victoria is alarmingly high possibly as a consequence of abundant supply of nutrients from rich agricultural lands near them. The situation is exacerbated by the shallow nature of many of the water bodies and the constant perturbations from both livestock and humans causing re-suspension of nutrients. Similarly, chlorophyll-a values observed during this survey were high and strong indicated that most of the water bodies are eutrophic. This situation calls for urgent intervention measures to avert health and environmental problems.

The open access and communal ownership of the water bodies pose a major handicap in the management of the water resources. This is mainly because at the lowest levels of communities, the village, weak legislations are applied across the board. This is mainly because communities that live within vicinities of the resources are often the poorest of the poor and are therefore pre-occupied with survival. Since the water resources are often small water masses with no obvious national importance to the central government, implementation water management regulations are minimally felt.

Nutrient enrichment run-off from the surrounding farms remains the main cause of eutrophication in the small water bodies. Contribution by rainwater is a also a possible contributor.

As a management measure, better agricultural methods should be practiced with a view to protecting the water bodies from pollution effects. As a first step, surrounding communities should be encouraged to make water points for livestock away from the dams. A return to optimum environmental conditions should see an improvement in ecological efficiency thereby enabling higher fisheries production in addition to enabling and sustaining diversity.

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References

APHA (American Public Health Association). 1995. *Standards for the examination of water and wastewater, 19th ed*. Port City Presss, Baltimore, Maryland, USA.

Crul, R. C. M. 1992. *Limnology and Hydrology of Lake Victoria. Monographs of African Great Lakes*. 90 p. UNESCO, Paris.

Cocquyt C and Vyerman W., 1993. *A check-list of the algal flora of the East African Great Lakers (Malawi, Tanganyika and Victoria).* National Botanic Garden of Belgium, Meise. 55p.

El-Fadel, M. L., El_Jisr, K., Jamali, D., 2001. Industrial waste management in developing countries: The case of Lebanon. *J. Environ. Mange.* 61, 281-300.

Falkenmark, M. 2000. *No freshwater security without major shift in thinking*. Ten-year message from Stockholm water symposia. Stockholm, Sweden.

Gitonga, J. K. 1992. Agricultural chemicals and wetlands: a review. *In.* S. A. Crafter, S. G. W Howard, eds. *Wetlands of Kenya. Proceedings of the KWWG Seminar on wetlands of Kenya*. National Museums of Kenya, Nairobi, Nairobi, Kenya.

Huber-Pestalozzi G. 1968. Cryptophyceae, Chloromonadophyceae, Dinophyceae. *Das Phytoplankton des Susswassers, 1. Teil* (ed. G. Huber-Pestalozzi), 2. Aufl., pp.I-IX + 1-322. Schweizerbart'sche-Verlagsbuchhandlung, Stuttgart.

Jeffries, M., Mills, D., 1990. *Freshwater Ecology: Principles and Application*. Belhaven Press, London and New York.

Gichuki N. (in press). Wetland Research In the Lake Victoria Basin, Kenya. Lake Victoria Research (Vicres). *Initiative Synthesis repor*t (in press)

Ogallo L. 1981. Trend of rainfall in East Africa. *Kenya J. Sc. Tech.* (A)2:83-90

R. W. Miller & D.T. Gardiner, 1998. S*oils in our Environment, 8th Edition.* Prentice-Hall Publishers, Upper Saddle River, NJ.

Ochumba, P.B.O., 1987. *Periodic massive fish kilss in the Kenyan portion of Lake Victoria. Water Qual*.. Bull. 12:119-122, 130.

Odera, D. R., Semu, E., Kamau, G. N., 2000. Assessment of cottage industry derived heavy metal pollution of soils with Ngara and Gikomba areas of Nairobi city, Kenya. *Afican J. Sci. Technol*. 1(2), 52-62.

Welcomme, R. L., 1972. The *Inand wates of Africa*. CIFA Tech. Pap. 1, 117 p. Committeee for Inland Fisheries of Africa (CIFA), FAO, Rome.

Eco-restoration of Chilika lake: A journey from Montreux record to the Ramsar Wetland Conservation Award

Pattnaik, A.K. IFS,

Chief Executive, Chilika Development Authority, C-11, BJB Nagar, Bhubaneswar-14, Orissa, India, Email:ajitpattnaik@hotmail.com

Abstract

Chilika lake, situated along the east coast of India, is a unique assemblage of marine brackish and fresh water ecosystem with amazing biodiversity and a Ramsar site. It shelters a number of endangered species listed in the IUCN red list of threatened species and an avian grandeur for more than one million migratory birds. The highly productive lake eco-system with its rich fishery resources sustains the livelihood of more than 0.2 million-fisher folk and 0.8 million people who live in the catchment of the lake. The lake was facing multidimensional ecological and anthropogenic pressures leading to an overall loss of biodiversity and productivity adversely affecting the livelihood of the local communit. The root cause of the degradation of the ecosystem of the lake were identified as; siltation, shifting of the inlet channel and shoal formation along the outer channel, fall in salinity, decline in fish productivity, proliferation of fresh water weed and invasive species, unauthorized shrimp culture etc. Ramsar Bureau added it to the list of the Montreux record (threatened list) in 1993 due to the changes in its ecological characters. Being concerned with this the Government of Orissa created Chilika Development Authority (CDA), for the restoration of this important wetland. The issues associated with restoration of the lake were vast in scale and also ecologically, hydro-logically and socio-economically complex, that warranted a meticulous integrated planning to address them. CDA initiated key targeted studies essential to a management-directed understanding of the ecosystem. An adaptive restoration plan to deal with the inherent complex and dynamic nature of the lake was developed. The inbuilt monitoring program, has been the uniqueness of the adaptive management which was designed to provide a feedback loop that enabled the planning team to assess the restoration assumptions, improve the model of the lake and its basin system, assess progress towards the targets, and adjust the plan to reflect what has been learned from the expanding knowledge base. While formulating the restoration plan through a consultative process the social and economic needs of the community were accommodated within the ecological capabilities of the system without compromising on the integrity of the ecosystem. The participation of local communities was made central to management of the lake and its drainage basin. To achieve this it was attempted to blend the wisdom, innovations and practices of the local community with the appropriate scientific reasoning into the restoration plan. Adequate emphasis was given to make the community understand the values and functions of the lake ecosystem through a meticulously planned out reach programme. The restoration strategy adopted was for a more ecologically beneficial hydrologic regime by way of opening of a new mouth to improve the exchange of water. It succeeded not only in rejuvenating the ecosystem of the lake but also immensely benefited the local community by way of enhancing their per capita family income due to increase in the productivity. There has been a steady increase in the fishery resources after the intervention that has gone up by eight times with reappearance of the native species. Other significant improvements are; expansion of the sea grass meadows with increase of their species diversity, increase in the population of Irrawaddy dolphins a flagship species with expansion of their habitat, significant reduction of invasive species like water hyacinth etc. Considering the sensitivity of the lake ecosystem restoration was carried out through an adaptive assessment process which are evaluated, refined, and supported by a strong continuous input, in form multi-agency scientific research, targeted studies to develop performance measures which collectively represent the response of the system to restoration efforts over a range of spatial, temporal and ecological scales. The restoration approach adopted by CDA is considered as a successful coastal wetland restoration model. The core value of the restoration model is its global relevance. This is testified by the fact that Chilika was removed from the Montreux record with effect from 11th November 2002 due to successful restoration of the lake. CDA is also conferred with the prestigious Ramsar Wetland Conservation Award. The paper will provide a glaring example of the holistic integrated participatory approach adopted for successful restoration of the wetland and its drainage basin with ecosystem approach which has immensely benefited the local communities by way of improvement of their livelihood.

Introduction

Chilika lake is situated along the east coast of India and a Ramsar site (Figure 1). The wetland is a unique assemblage of marine brackish and fresh water ecosystem with amazing biodiversity, that shelters a number of endangered species listed in the IUCN red list. It is an important wintering ground for more than one million migratory birds. The highly productive eco-system with its rich fishery resources sustains the livelihood of more than 0.2 million-fisher folk and 0.8 million people who live in the lake basin. The lake is influenced by three subsystems viz. Mahanadi river system, rivers flowing in to the lake from the western catchment and the Bay of Bengal (Figure 2). These, together with certain climatic factors such as rainfall, evaporation and wind patterns influence the two key factors that shape the geomorphology and determine the biological productivity and diversity of the lake i.e. the salinity regime and sedimentation patterns. A complex combination of freshwater discharge, evaporation, wind condition and tidal inflow of seawater govern the salinity profile, which exhibits both temporal and spatial variations. Due to the shallow depth of the lake, there is very little stratification. The lake had been facing multidimensional ecological and anthropogenic pressure leading to an overall loss of biodiversity and productivity adversely affecting the livelihood of the local communities who have been depending on the wetland. Constructions of major hydraulic structures on the river systems and the change in the land use pattern in the catchment are also responsible for the alteration in the flow pattern into the lake. Ramsar Bureau added it to the list of the Montreux record (threatened list) in 1993 due to significant changes in its ecological characters.

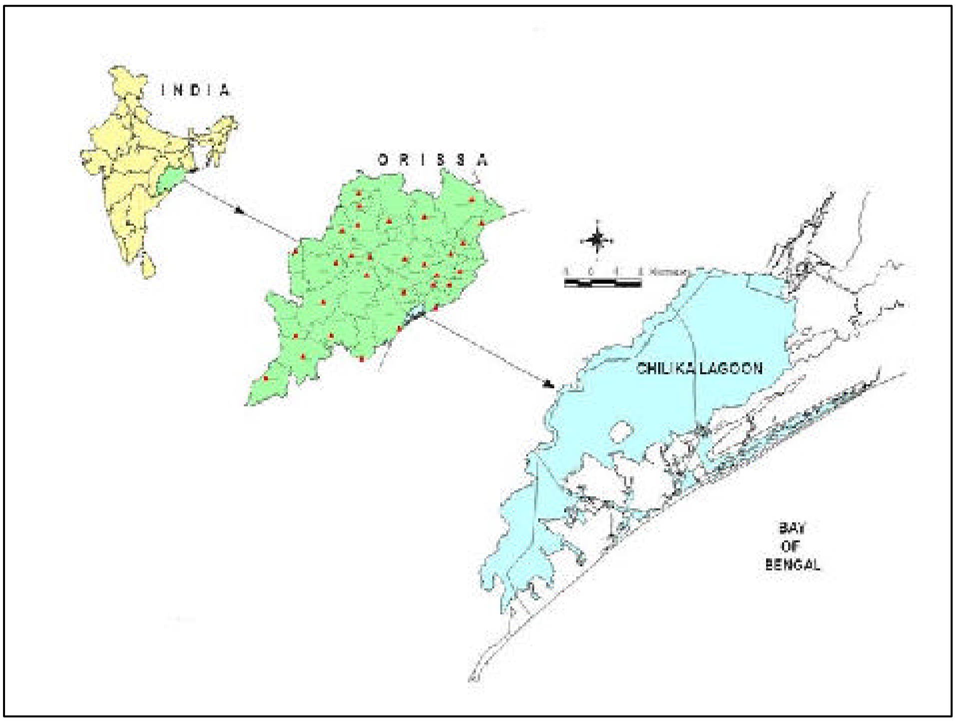

Figure 1. Location map of Chilika.

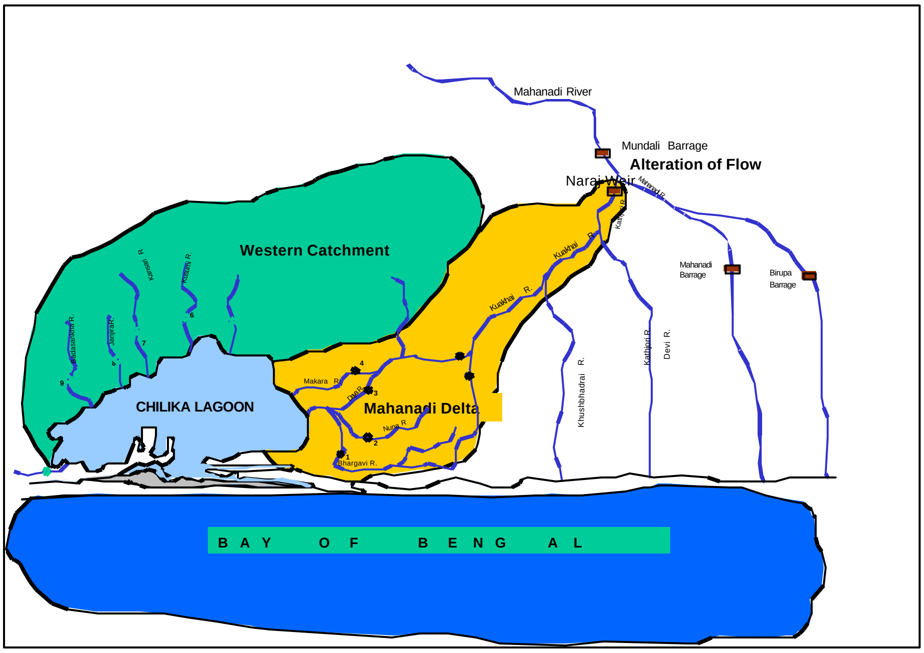

Figure 2. Hydrological set-up of Chilika lake.

Being concerned with this the Government of Orissa created Chilika Development Authority (CDA), for the restoration of this important wetland. The root cause of the degradation of the lake were identified as; siltation, shifting of the inlet channel and shoal formation along the outer channel, fall in salinity regime, decline in fish landing, proliferation of fresh water invasive species, poor discharge of flood water leading to water logging in the peripheral crop land, unauthorized shrimp culture etc. For a clear understanding of this complex and sensitive ecosystem and to trace the root cause of degradation, CDA commissioned the services of the premier institutes of the country like National Institute of Oceanography (NIO) and Central Water and Power Research Station (CWPRS), to carry out the targeted studies. Based on their findings, they recommend the measures to reverse the process of degradation of the lake ecosystem. The main thrust of the restoration plan has been to maintain the structural and functional integrity of the lake ecosystem and maintain its productivity. This implies that the flow of benefits from the system should commensurate with the carrying capacity, so that the ecological integrity of the system can be maintained. Through a meticulously designed out reach programme this key concept is disseminated to the local communities. The local communities are considered as an integral component of wetland eco- system.

As more than 0.20 million fisher folk depend on the lake, and about 0.80 people live in the lake basin, while formulating the management plan, wide stakeholders consultation was given utmost priority to make it as inclusive as possible. The issues involved in restoration of a complex ecosystem like Chilika with multitude stakeholders, who had lost their hope on the system due to sharp decline in productivity, was a tough job. So involvement of local communities has been made central to the management of the lake and its basin. The communities were empowered and facilitated to share knowledge and experience; they formed community level institutions for management of the resources and they provided support by way of self initiated good practices and norms for sustainable management of the lake. Restoration of such complex and sensitive ecosystem system needs very careful planning wide consultation and lot of scientific research and input. To achieve this it was attempted to blend the wisdom of the local community with the appropriate scientific reasoning into the restoration plan. The approach adopted by CDA demonstrates that active community participation is vital for halting the processes of degradation and achieving social equity and sustainable management of the wetland and its drainage basin. This was achieved through the village level institutions, women self help groups, community based organizations and networking of the NGOs.

Considering the complex and sensitive eco-system of the lake, it was realized that it argues for drawing on the best available scientific knowledge and targeted research and studies to obtain chunks of knowledge that are critical to formulate the restoration plan. To achieve this, there was need for a proper conceptual hydrological and hydrodynamics model. So, CDA initiated extensive hydrological studies and hydrodynamics modeling by commissioning the services of the premier institutes of the country to trace the root cause of the degradation of the lake ecosystem and find out the best options to restore it. Through an extensive hydrological study the useful data for setting the boundary conditions for the hydrodynamics and hydrological modeling were generated. Based on the out come of the above study, the Central Water and Power Research Station (CWPRS), recommended for opening of a new mouth to restore the hydrological regime of the lake. Considering the complex multitude niche of the lake ecosystem, a critical hydrological intervention of this dimension, EIA, was carried out by multidisciplinary professionals from National Institute of Oceanography (NIO) to generate base line information to predict the possible impacts of the intervention and to suggest possible mitigation measures. Interestingly during the process of consultation, it emerged that the opening of an artificial mouth, was a long-standing desire of the local community. The strategy adopted by CDA was to opt for a more ecologically beneficial hydrologic regime to; improve water quality, recovering the lost habitat of the important species, enhancement of fishery resources and controlling the invasive species. With the input from the experts, scientific findings and the community endorsement a very convincing restoration plan could be formulated, which enabled the Chilika Development Authority to receive funds from Govt. of India to implement the most appropriate restoration intervention.

Finally the historic hydrological intervention by way of opening of the mouth was carried out on 23rd September 2000. After the hydrological intervention there had been a significant improvement of the ecosystem of the lake, which is considered as one of the most successful restoration initiative globally. As predicted by the, CWPRS, after the completion of the desiltation and opening of the mouth, there has been increase in the salinity flux and tidal flux by 40% and 45% respectively. The improvement in the tidal flux turned the lake in to pulsing mode. It succeeded not only in rejuvenating the ecosystem of the lake but also immensely benefited the local community by way of enhancing their per capita family income due to increase in the productivity. After the hydrological intervention, there has been a marked improvement in the exchange of water between the sea and the lake with restoration of salinity regime. The gradual reduction in the salinity gradient from the lake mouth to the lake interior after the opening of the mouth is providing the desirable sense of direction for the euryhaline forms to enter into the wetland from the sea. This is facilitating the auto-recruitment of the fish, prawn and crab juvenile into the lake. There has been a significant improvement of the fish production. The per capita family income of the fisher community has gone up by 1000 US $ per annum .The fish landing which was a mere 1,600 Metric tons before opening of the mouth increased to 14,000 Metric tons during 2004. Six species of native fish of economic importance reappeared after the intervention, which had disappeared due to degradation of the ecosystem. The other ecological performance indicators which showed improvement are; expansion of the sea grass meadows with increase in species diversity, increase in the population of Irrawaddy dolphins a flagship species, with expansion of their habitat, reappearance of six native fish species, significant reduction of invasive species like water hyacinth are few such examples. It is heartening to note that after the intervention the local community has started a number of self-initiated good practices as they regained their faith on the lake ecosystem after the restoration intervention.

To track the changes in the lake ecosystem, an intensive monitoring protocol is developed. Through this process the restoration interventions are evaluated, refined, and supported by a strong continuous input, in form of multi-agency scientific research, targeted studies to develop performance measures and a comprehensive close monitoring of key components. In addition to this to track the system a set of biological and physical parameters most appropriate to the ecosystem of the lake were meticulously identified as performance measures. These performance measures are being used starting from the implementation phase to evaluate how, specifically, the system is responding to various interventions and meeting the fundamental restoration objectives. This includes long term close monitoring of the key physico-chemical and biological parameters of the lake from 30 monitoring stations, stream gauging to assess the silt and the fresh water inflow in to the system, monitoring of nutrient load from point and non point sources, fish stock assessment, monitoring of avian fauna, study of habitat and distribution of Irrawaddy dolphin, monitoring of the sea grass meadows etc.

The lake basin, spreading over more than 4000 square kilometers was the logical starting point for planning and management actions for sustainable management. The environmental flow assessment provided necessary clues regarding the significance of the freshwater flow from the drainage basin to maintain the ecological integrity of the lake. The large-scale silt flow from catchments (0.365 million cubic meters, assessed through the stream flow measurement) was identified as one of the biggest management problems (Pattnaik, 2002). Further assessment revealed that land degradation in the drainage basin not only leads to enhanced silt flow into the lake but also causes poverty, due to low productivity in the drainage basin. The chief livelihood strategy adopted by the watershed community is rain-fed paddy cultivation once a year. The average annual rainfall received is 1400 millimeters, but because it is not well spread, total or partial crop failure is a common phenomenon. In spite of the endowment of natural ecosystems, which could have constituted a livelihood provider for the lake basin communities with sustainable income generation and adequate employment opportunity, due to degradation of the basic life support system, the agricultural productivity is low. Poor crop productivity had been adversely affecting the livelihood of the watershed community, consequently triggering migration in search of employment. The production was as low as 5-8 quintals per acre even during the good crop year. The depletion of natural resources and loss of their productive capacity had imparted huge costs on the local communities due to declining agricultural productivity and acute shortage of water. This has forced the local communities to adopt tree felling from the nearby forest as a means of livelihood. An innovative participatory micro-watershed management concept was adopted to address this problem with a “sustainable livelihood” approach for holistic management of natural resources. The lake basin management is conceived as a long-term participatory process. The objective of this concept has been to facilitate the community through empowerment to take decisions and build capacity to work collectively. The participation of local communities and stakeholders in planning and implementing management of natural resources and in sharing the responsibilities of decision-making is a key feature of the ecosystem approach. They have considerable, relevant knowledge of the ecosystem and ways in which it can be sustainably managed. The basic approach was to create an enabling environment, through capacity building of the community, community based organisations and NGOs at the outset, and a series of need-based training programmes to facilitate an integrated and holistic management of micro-watershed by the community. The goal was to enable the community to manage and reverse degradation of life support systems within the watershed, particularly land and water, to enhance the productivity, resulting in alleviation of poverty and promoting improvements in livelihood of agricultural communities. The focus was on restoration and conservation of a degraded life support system within the micro-watershed. In general, many find it difficult to identify with the concept of eco-system processes. An analysis of needs, value and perspectives of local communities are fundamental to ecosystem management. To achieve this, an innovative grassroot approach was adopted by the CDA, by formulating a micro-plan, blended with indigenous knowledge and appropriate experts’ input, for optimum utilization of the natural resources in a sustainable manner and to increase productivity and provide equal opportunity for livelihood for the landless, marginal farmers and women. To ensure the involvement of the community and sustainability, it was ensured that the watershed agricultural community share a part of the costs of the treatment towards the watershed development fund which would be utilised for maintenance and further improvements of the watershed assets created after the project period is over. The watershed association and the user groups had been able to efficiently implement the micro-plan in consultation with the community. One of the most successful initiatives was a series of rainwater harvesting structures, which they designed and installed. They succeeded in recharging aquifers and transforming local ecosystems as well as their surrounding economies. The advantage of the system is that along with arresting rainwater; it improved the moisture regime in the field, particularly downstream. After the rainwater harvesting structures had been constructed, the production of rain-fed paddy improved and there has been no crop failure due to an erratic rainfall. The farmers have started growing a second cash crop like wheat, sunflower and pulses after the main crop is harvested due to improved moisture regime. The yield per hectare has improved to 8-10 quintals per acre. The small rainwater harvesting structures are swift to show regenerative results. The most visible change is the presence of water as indicated by the recharged wells and the greenery in the village. The villagers say that after 2001 there have been a rise in agricultural productivity and two crops can now easily be grown annually (Pattnaik, 2001). This is believed to be a result of both the water harvesting and regeneration of forests. Emigration also significantly decreased with the increase in agricultural production and creation of employment opportunities for the landless labourers. Now the intricate link between vegetation water and livelihood is more apparent to the local communities. The holistic management of natural resources at the grass-roots level also facilitated conflict resolution. The longstanding village level conflicts and differences of opinions within the micro-watershed area were resolved as a result of the participatory initiative. Additionally, it reduced the ecosystem's vulnerability to drought, improved agricultural incomes for small farmers, increased wage-labor opportunities for the landless, provided an equitable distribution of benefits to the most impoverished, and reduced environmental degradation and drudgery. Notably, there have been increased earnings from land and non-land activities for the poor, reduced debt, and improved livelihood and food security leading to further poverty alleviation, reduced environmental degradation and reduction in the silt load into the lake. The outside migration in search of employment has reduced by 70%. The long-standing inter-village conflict was ploughing these non- descript villages for the past 17 years, with 16 criminal cases of grievous nature pending in the court of law and the loss of two precious lives. With the launch of the watershed programme four years ago, complete inter-village conflict resolution has been achieved. The micro-watershed became the model in the context of social integration. It has also taken the lead in integrating women from all communities into the mainstream by way of empowerment through self-help groups and their active participation in the watershed management. The project brought a silent revolution and the local community now lives in harmony, as they say, "We are now an extended watershed family and there is no question of discrimination."

The women of the community benefited in a special way through the formation of the women self-help groups (SHG) and capacity building training for skill improvement. The training was organized with the assistance from National Bank of Agriculture and Rural Development (NABARD). Through a micro-credit mechanism, the members of the SHGs adopted income-generating activities to supplement their family's income. By working to earn for themselves, the women empowered themselves against the prevailing social taboo, now they are better placed to take the decision on financial matter.

Due to fall in salinity, there was proliferation of the fresh water invasive species. The weed spread area, which was mere 20 square kilometers. in 1972 increased to 523 square kilometers by October 2000, leaving a weed free area of bare 334 square kilometers. After the hydrological intervention, the weed free area improved to 506 sq.km (Pattnaik-2001). A wetland research and training centre is established by CDA to carry out required targeted studies, elective research and monitoring and assessment. The research centre is equipped with the state of art laboratory, modelling infrastructure, GIS and Image processing software and hardware as well as the infrastructure for the training.

It is a perfect example of how the restoration of a wetland with most appropriate strategy can not only restore the ecological integrity of the wetland, but also, how it can contribute significantly towards the improvement of livelihood of the local community due to increase in the productivity. The valuation of the enhanced fishery resources alone after the restoration stands at 13.5 million US$ annually. The increase in the productivity, both in the wetland as well as the watershed due to the good environmental practices, has facilitated the poverty alleviation of the community. The community participation, linkage with the various national and international institutions, intensive monitoring and assessment system are some of the uniqueness of the management practices adopted by CDA for restoration of this unique wetland. It is worthwhile to mention here that the lake is located in a developing country and in a province with severe resource crunch. The restoration task could be completed with the limited resource available indigenously without any overseas funding or loan from any financial institutions. With the strategic planning and most efficient and appropriate utilisation of the very limited available resources in from of grants from the Government of India (equivalent to mere 11 million US $), the vast task of restoration of this wetland could be accomplished . Chilika is removed from the Montreux record by the Ramsar bureau with effect from 11th November 2003 for successful restoration of the lake. The prestigious Ramsar Wetland Award and the Indira Gandhi Payavaran Purashakar are also conferred on CDA for the impressive way in which the restoration was carried out with the active participation of the community. One of the core values of the restoration model adopted by CDA is its global relevance.

References

Pattnaik, A. K., 2001. *Participatory watershed management of Chilika lake : A Ramsar site of India*. 9th Inter. Conf. On the conservation and management of lakes held at Shiga, Japan from 11-16th Nov. 2001.

Pattnaik, A. K., 2001. *Opening of a new mouth: A step to restore the ecosystem of Chilika lake- A Ramsar site of India*. Proc. Workshop on Envi. Mana & Wiseuse Brackish waterlakes, lake and Bays. 9th Int. Conf. on Cons. & Management of lakes. P.14-18.

Pattnaik, A. K., 2002. *Rejuvenation of Chilika lake: An example of ecological restoration with community participation*. Workshop on Envir. Governance and sustainable Dev. The contribution of the Ramsar Convention. GBF-17 – Valencia. P.1-4.

The role of the Yala swamp lakes in conservation of Lake Victoria region haplochromine cichlids: evidence from molecular genetic and trophic ecology studies

Abila Romulus[1] ,Walter Salzburger[2] , Millicent Florence Ndonga[1] , Dickson Otieno Owiti[1] and Marta Barluenga[2]

[1] Department of Zoology, Maseno University. P.O. Box 333 Maseno Kenya.
[2] Department of Biology, Laboratory of Evolutionary Biology, University Konstanz, Universitaetsstrasse 10, 78457 Konstanz, Germany.
[1]Email-owinoroma@yahoo.com

Abstract

Lake Kanyaboli, a satellite lake of Lake Victoria, has been suggested as a potential refugium for haplochromine cichlids that have gone extinct in Lake Victoria.

We employed mitochondrial DNA and microsatellite DNA molecular markers as well as feeding ecology studies to re- evaluate the evolutionary and ecological significance of Lake Kanyaboli haplochromines. The mitochondrial DNA and microsatellite markers revealed high genetic diversity in the endangered *Xystichromis phytophagus* and also the presence of mtDNA haplotypes that may have either gone extinct in Lake Victoria or have arisen *in situ*. Lake Kanyaboli thus acts as a 'genetic reservoir' for the Lake Victoria species flock.

Gut content analysis revealed six trophic groups among the six haplochromine species. The haplochromine community in Lake Kanyaboli therefore exhibits trophic specializations. The relatively high trophic diversity in this cichlid community contrasts with the currently simplified trophic relationships of Lake Victoria. This high trophic diversity contributes to high energy flow and overall ecological efficiency of the lake.

Lake Kanyaboli and similar satellite lakes therefore provide an opportunity for conservation of both genetic and trophic diversity threatened by introduction of exotics in the Lake Victoria basin. Lake Kanyaboli should thus be recognized as an important Evolutionary Signficant Unit (ESU) for Lake Victoria region haplochromine species. Basin wide molecular genetic characterization of the other tilapiine cichlid species as a basis of identifying genetically robust stocks that can be used in aquaculture or to restock Lake Victoria should be undertaken.

Key words: cichlids, conservation, trophic ecology, genetics, Victoria, Yala swamp.

Introduction

The haplochromine cichlids of Lake Victoria have been noted to have one of the highest rates of speciation and adaptive radiation among living vertebtates (Seehausen, 2002). This extraordinary adaptive radiation has been attributed to sexual selection as well as feeding specializations. Feeding specializations of the haplochromine cichlids have been instrumental in resource partitioning and therefore in shaping the cichlid community structure (Seehausen and Bouton, 1997) and maintaining the high diversity (Bouton *et al*., 1999). This trophic differentiation contributed to the evolution and adaptive radiation of the cichlid flock of Lake Victoria.

The cichlid fauna of Lake Victoria formed an important component of the fisheries of the lake and therefore played a critical role in provision of protein requirements to the riparian communities (Sumaila, 2000). However, in the 1980's the haplochromine cichlid fauna of Lake Victoria experienced unprecedented rate of extinction. Loss of the haplochromines have been attributed to pollution that hinders sexual selection (Seehausen *et al*., 1997) as well as predation from the exotic Nile perch (*Lates niloticus*, L) (Ogutu – Ohwayo, 1990).

The conservation of the remaining cichlid species is therefore of utmost importance. In order to protect the remaining populations it is important to be able to characterize them genetically. Such genetic information can be used as basis for future re-stocking or aquaculture. Some of the extinct cichlid species still thrive in small isolated water bodies (commonly referred to as satellite lakes) scattered around the Lake Victoria basin (Loiselle, 1996, Aloo, 2003). These range from small lakes to dams and reservoirs. These water bodies have been recognized to have special significance in the conservation and the future survival of these cichlids since they act as 'refugia' (Kaufman and Ochumba, 1993; Maithya, 1998).

The aim of this work was to evaluate the conservation significance of Lake Kanyaboli, the largest Yala wetland lake by studying trophic ecology of the six common haplochromine cichlid species as well as population genetics of the endangered *Xystichromis phytophagus* (Greenwood, 1965) using neutral molecular markers.

Materials and methods

The study area

The study was carried out in Lake Kanyaboli, a small (10.5 km^2) and shallow freshwater lake (average depth: 2.5 m; maximum depth: 4.5 m) situated in the Yala wetlands in Western Kenya (Fig. 1). The Yala swamp is Kenya's largest freshwater wetland and covers about 175 km^2 along the northern shores of Lake Victoria. It is bordered to the North by the Nzoia River and to the South by the Yala River. Three main lakes exist in the Yala wetlands (Kanyaboli, Namboyo, Sare), of which Lake Kanyaboli is the largest and most remote from Lake Victoria. Lake Kanyaboli is separated from Lake

Victoria by massive papyrus swamps that presently inhibit faunal exchanges between the two lakes. No Nile Perch has ever been observed in Lake Kanyaboli, corroborating that it has been isolated from Lake Victoria at least since the 1950's. The fish fauna of Lake Kanyaboli is dominated by cichlids – three species of tilapia (*Oreochromis esculentus,* (Graham, 1929) *Oreochromis variabilis* (Boulenger, 1904), and *Oreochromis leucostictus* (Trewavas, 1983) and haplochromine cichlid species (Kaufman & Ochumba 1993; Aloo, 2003). Besides its cichlid fauna, the Yala swamp is home to a rich and complex community of animals including the endangered Sitatunga antelope (*Tragecephalus spekei*) as well as papyrus endemic birds.

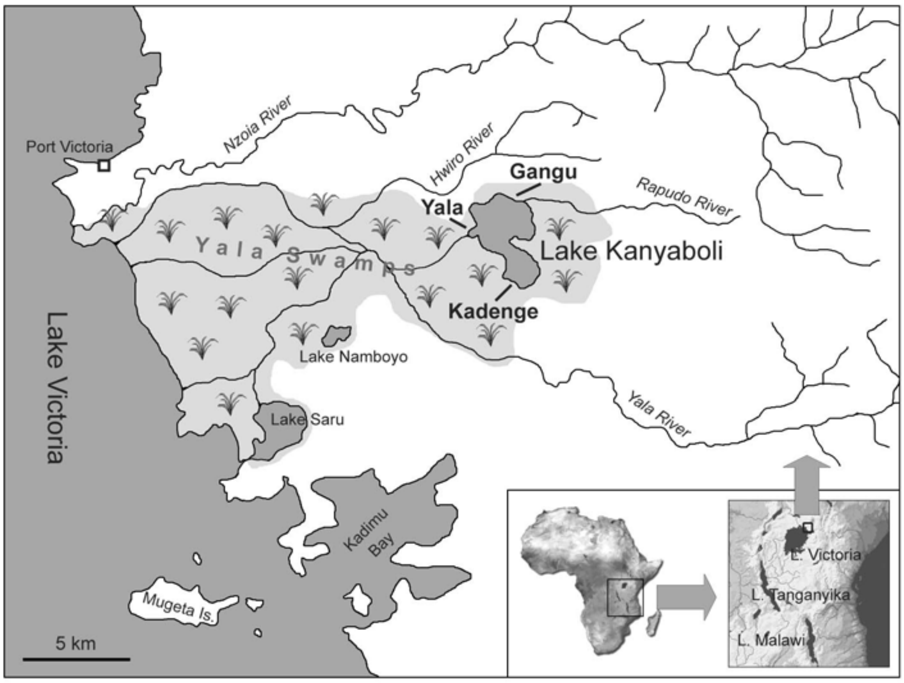

Figure 1. Map of yala swamp showing the position of Lake Kanyaboli and other associated lakes. (Re-drawn fro Crafter *et al*., 1992).

Fish sampling and trophic relationships of the haplochromine cichlids

Six species of haplochromines namely *Astatoreochromis alluaudi* (Pellegrin, 1903), *Lipochromis maxillaris* (Greenwood, 1965), *Astatotilapia nubila* (Boulenger, 1906), *Xystichromis phytophagus* (Greenwood, 1965), *Pseudocranilabrus multicolor victoriae*

(Seegers, 1990) and *Astatotilapia* 'big eye' (Kaufman, 1996) were collected from Lake Kanyaboli using a 3.81 cm multifilament gill net. Due to its small size, *P.m. victoriae* samples were obtained by angling along the papyrus fringing swamps. *X. phytophagus* specimens were grouped into three distinct populations – Kadenge, Gangu and Yala for population genetic analysis. The fish were gutted and preserved in 90% ethanol. Contribution of each food item to the diet was determined by determining the relative abundance, percentage occurrence and prominence values as described in Hyslop (1980).

DNA extraction, amplification via PCR, purification and sequencing

Muscle tissue from 90 % ethanol preserved specimens was used as source of DNA.

Total DNA was extracted by sodium chloride extraction and ethanol precipitation after initial proteinase K digestion (Bruford *et al.,* 1998).

For polymerase chain reaction (PCR) amplification of the first section of the mitochondrial control region, the fastest evolving segment of the mitochondrial genome, the published primers L-Pro-F and TDK-D were used. PCR amplification was performed in a reaction volume of 21.1 µL (9.9 µL HPLC water, 2 µL buffer, 1.6 µL 10mM dNTPs, 1.4 µL 10 mM $MgCl_2$, 2 µL of each primer/2nM, 0.2 µL TAQ DNA polymerase and 2 µL of diluted DNA) under the following conditions: 35 cycles with a denaturation phase at 94 °C for 30 s, an annealing phase at 52 °C for 30 s, and an extension phase at 72 °C for 90 s. PCR products were visualized by mini-gel electrophoresis using Ethidium-Bromide staining and 1 % agarose gels.

Two micro-liters of purified PCR product were used as template in the cycle sequencing reaction. The reaction mixture for cycle sequencing was made up of 1 µL of 10 µM L-Pro-F primer, 1.5 µL of the Big Dye termination reaction mix (Applied Biosystems) and 5.5 µL of HPLC water. The annealing temperature for cycle sequencing was adjusted to 50 °C. The cycle-sequenced products were purified with an ethanol – sodium acetate precipitation, re-suspended in 15 µL of HPLC water and analyzed on an ABI 3100 capillary DNA sequencer (Applied Biosystems).

Microsatellite survey

Six microsatellite loci, which were developed for the cichlids *Copadichromis cylicos* (Kellog *et al*., 1995), *Tropheus moori* (Zardoya *et al.,* 1996) and the East African mollusc crusher *Astatoreochromis alluaudi* (Wu *et al*., 1999) were initially tested by sequencing to assess their utility for population studies in the haplochromine cichlids. The six microsattelites loci tested were chosen for population analyses on the basis of the repeat number in the sequenced alleles. The selected loci show atleast 10 uninteruppted CA or AC dinucleotide repeats and were, therefore, considered to exhibit sufficient potential for polymorphism for population analysis. The selected loci were TMOM11, TMOM5, UNH001, UNH002, OSU20D and TMOM27A. For sequencing, Polymerase Chain Reaction (PCR) were carried out in 21.1µl volumes (9.9 µl HPLC water, 2.0 µl buffer, 1.6 µl dNTP, 1.4 µl 10mm MgCl2, 2.0µl of each locus specific primer, 2.0 µl of diluted DNA and 0.2 µl Taq DNA Polymerase) using the above PCR conditions. The PCR product was diluted 1:10. 1 µl of the diluted PCR was added to 0.125 µl 500bp Size Standard and 0.9 µl HPLC water. Denaturation was performed for 4 minutes at 94^{o}C and the samples immediately placed on ice. Direct sequencing was carried out on an ABI PRISM^{R} 3100 automatic sequencer. Detection of microsatellite alleles in genomic DNA was achieved by end – labeling the forward primer of the pair with FAM or HEX dyes. The sequencer output was automatically analysed with an adapted GENESCAN® ANALYSIS programme (Applied Biosystems) and the fragment size (genetic loci polymorphism) determined (scored) using GENOTYPER® software.

Data analysis I: Mitochondrial control region

For the mtDNA data genetic variability was estimated by calculating the haplotype diversity in each population. The genetic differences between the sampled populations of the *Xystichromis phytophagus* were tested using F – statistics (The fixation index) (Weir and Cockerham, 1984) as calculated by ARLEQUIN^{R} 2.000 sofftware (Schneider *et al*., 2000). The fixation index, F, serves as a convenient and widely used measure of genetic differences between populations (Wright, 1978). Theoretically F_{ST} has a minimum of 0, indicating no genetic difference and a theoretical maximum of 1, indicating fixation for alternative alleles/ haplotypes in the sub – populations.

Data analysis II: Microsatellite markers

For the microsatellite data the genetic polymorphism was estimated for each population with GENEPOP 3.1d (Raymond and Rousset, 1995) as implemented in the ARLEQUIN^{R} 2.000 sofftware (Schneider *et al*., 2000) as the number of alleles per locus (N_A), the observed (H_0) and the expected heterozygosity (H_E). Departure from Hardy – Weinberg expectation for every locus was calculated in each population and between populations using a test analogous to Fisher's exact tests (Guo and Thompson, 1992) estimated with a 100 000 step, 1000 iteration, Monte Carlo Chain series of permutation, as implemented in the software ARLEQUIN^{R} 2.000 (Schneider *et al.*, 2000). Genotypic linkage disequilibrium was evaluated with GENEPOP 3.1d as implemented in the ARLEQUIN^{R} 2.000 software (Schneider *et al*., 2000). Differences between populations in their haplotypic (mtDNA sequences) and allelic (microsatellites) distributions was tested with an exact test of population differentiation (Raymond and Rousset, 1995) using the software ARLEQUIN^{R} 2.000 (Schneider *et al.,* 2000) as F_{ST}, the pairwise fixation indices, based on haplotype or allele frequency variations. The significance of genetic subdivision was assessed using 1000 permutations.

Results

Feeding and trophic relationships of the haplochromine cichlids

Gut content analysis revealed that eight food items comprised the diet of the six haplochromine species. These food items are algae (both blue green algae and diatoms), chironomid and *Chaoborus* larvae, other unidentified insects, mollusks, fish embryos, fish eggs, plant remains and detritus. Based on frequency of occurrence, chironomid and chaobarus larvae was the main food taken, occurring in 64.3 % of the guts examined, followed by plant remains (43.8%), detritus (37.5%) and other insects (33.5%). All the six haplochromine species examined fed on chironomid/*Chaoborus* larvae. Molluscivory and paedophagy (egg and embryo feeding) were restricted to *Astatorechromis alluaudi* (Pellegrin, 1903) and *Lipochromis maxillaris* (Greenwood, 1980) respectively. The two food items therefore contributed less to the total amount of food items taken by the haplochromines. Only 15.2% of the guts examined contained mollusks and 16.1% and 6.25% contained fish embryos and eggs respectively.

Based of relative abundance chironomid and chaobarus larvae constitutes the highest amount of food taken (29.3%). The low values of relative abundance shows that each type of food is taken in low quantities. The overall diets of the six haplochromine species expressed as frequency of occurrence and relative abundance are given in table 1. *Astatotilapia nubila* (Boulenger, 1906) can be classified as an insectivore, *Astatotilapia* 'big eye' (Kaufman, 1996) as an algivore, *Lipochromis maxillaris* as a paedophage, *Xystichromis phytophagus* (Greenwood, 1965) as a plant feeder, *Astatoreochromis alluaudi* as a molluscivore and *Pseudocranilabrus multicolor victoriae* (Seegers, 1990) as an algivore.

Population genetic structure inferred from the mitochondrial DNA sequences

DNA sequence of the 400 base pair segment of the mtDNA revealed 11 distinct haplotypes present in the 205 specimens of *Xystichromis phytophagus* from Lake Kanyaboli. Mitochondrial DNA thus revealed high genetic diversity within this species. Private haplotypes were only found in two populations, Gangu and Kadenge, however in very low abundance. The haplotype frequency especially of the three main haplotypes occurring in 83.9% of all specimens was similar in the three populations (Table 2).

Population genetic structure inferred from the microsatellites

The six microsatellite markers exhibited high polymorphism in *X. phytophagus* characterized by multiple numbers alleles. A total of 152 alleles were found in 191 individuals. At each of the six loci the total number of alleles observed ranged from 12 to 20 in TMOM5, 13 to 19 in TMOM11, 21 to 24 in UNH001, 11 to 14 in UNH002, 30 to 34 in OSU20D and 10 to 12 in TMOM27A. Except for the locus TMOM27A, the allele frequencies for the most frequent alleles were low (less than 0. 3). The microsatellite alleles exhibit similarities across the three populations although some alleles were restricted to only one of the three populations (Tables 25 - 30). Levels of allelic diversity were consistently high in each of the three populations (19.2 ±9.6). For all the loci, the same alleles were the common alleles in all populations.

Genetic diversity as measured by expected heterozygosity was high (0.89±0.00025) (Table 3). The observed heterozygosity was high and ranged from 0.83 to 0.96 in TMOM5, 0.84 to 0.85 in TMOM11, 0.89 to 1.00 in UNH001, 0.83 to 0.97 in UNH002, 0.85 to 0.93 in OSU20D and 0.48 to 0.79 in TMOM27A. The genotype frequency at locus TMOM27A showed relatively high deficiency of heterozygotes in the three populations. This locus thus probably has null alles. The locus OSU20D exhibited the highest variability (i.e. most polymorphic) with 30 to 40 alleles per population while the locus TMOM27A with 10 to 12 alleles per population was the least variable. Genotype frequencies at the loci UNH002 and TMOM27A showed deviations from Hardy – Weinberg expectations in the Yala population while the loci TMOM11, UNH002, OSU20D and TMOM27A showed deviations in the Kadenge populations. All loci were in accordance with the Hardy – Weinberg expectation in the Gangu population ($P < 0.05$). There was also no evidence of linkage disequilibrium between any pair of loci in any of the three populations.

Discussion

A relatively high number of trophic groups was observed in this study compared to the lower number of trophic groups in the Nile perch impacted lakes of Victoria region. The absence of Nile perch in Lake Kanyaboli can be attributed to the presence of the massive swamp separating Lake Kanyaboli and Lake Victoria. Similar higher trophic diversities have also been made in the satellite lakes Nawampasa, Gigati and Agu in Uganda that have been impacted by the Nile perch (Mbabazi *et al*., 2004). Greenwood (1981) and van Oijen (1990) showed that the pre Nile perch Lake Victoria haplochromines exhibited diverse feeding habits including detritivory, insectivory, higher plant feeding, zooplanktivory, molluscivory, paedophagy and piscivory. The presence of Nile perch has however simplified the trophic structure of the haplochromines from the above diverse trophic groups to only two belonging to a single trophic level (Namulemo, 1998). Most of the pre Nile perch trophic groups are however still represented among the six Lake Kanyaboli haplochromines and here they occupy three trophic levels i.e primary, secondary and tertiary levels. Lake Kanyaboli therefore has a more direct flow of energy from primary to tertiary consumers through the haplochromines and as such the haplochromines play an important role in energy flow and overall ecological efficiency of the lake system.

Understanding the spatial and temporal dynamics of endangered species and populations are critical to effective conservation and management (Soule, 1987). One goal of conservation biology is to conserve genetic diversity and evolutionary processes. Quantification of levels of genetic biodiversity in extant endangered species is being recognized as important in the recognition of taxonomic units in need of protection (Avise, 2000, Moritz, 1994). Most endangered species exhibit very low genetic variabilities probably due to genetic drift and historical bottlenecks in population size. In some case studies, plausible arguments have been advanced for a direct association between observed molecular variability and the long – term viability of an endangered taxon (Soule, 1987). An understanding of the genetic structure of a population is also key to our understanding of the importance of genetic resources and the importance of genes for the conservation of species and biodiversity. In the broadest sense, conservation of the genetic diversity and intergrity of a species relies on identifying the critical genetic units and then managing these units in a co-ordinated manner (Lesica and Allendorf, 1995).

Both molecular markers employed in this study have revealed high genetic variability within *Xystichromis phytophagus*. The high genetic variability exhibited in this species augurs well for the species and implies that genetic variability has been conserved in this species. This reflects a historically large effective population size and lack of population bottlenecks in the past. The theory of genetic drift predicts that bottlenecked populations should exhibit very low heterozygosities. Large effective population sizes are important in minimizing the effects of drift which may lead to population differentiation even in the presence of gene flow. The species is thus not in immediate danger of extinction in Lake Kanyaboli.

One strategy that has been proposed to restore the cichlid populations in the Lake Victoria basin has been aquaculture and captive breeding (Maithya and Okeyo-Owuor, undated abstract). The success of such aquaculture ventures rely on identifying genetically pure and robust populations as a source. The finding of a genetically robust population of *X. phytophagus* in Lake Kanyaboli indicates that this population can be used as source to re-stock other genetically depauparate lakes of the Lake Victoria region (Loiselle, 1996). The high genetic diversity occurring in the *X. phytophagus* population can be used as a strong case to support this. There is therefore need to genetically characterize other cichlid populations within the Lake Victoria satellite lakes. Among the tilapiines, populations of *Oreochromis esculentus* (Graham, 1929) and *Oreochromis variabilis* (Boulenger, 1904) still thrive in Lake Kanyaboli and other Yala swamp lakes. Molecular genetic characterization of such populations can form the basis of 'fingerponds' environmentally sustainable aquaculture along the fringes of the lakes.

This study has shown that Lake Kanyaboli provides an opportunity for conservation of not only species but also of trophic and genetic diversity threatened by introductions of exotics and other anthropogenic impacts in Lake Victoria. Lake Kanyaboli is therefore an important refugium to Lake Victoria haplochromines. There is an urgent need to spearhead the conservation of this lake. Because of the critical socio – economic role Lake Kanyaboli plays in the lives of the local community (Abila, 2002, 2005) it is strongly recommended that any conservation initiatives be community based. Unfortunately, ongoing land use changes ostensibly to improve food security in this area have greatly altered the wetland and is major threat to the future survival of its biodiversity.

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References

Abila, R., 2005. Biodiversity and sustainable management of a tropical wetland lake ecosystem: A case study of Lake Kanyaboli, Kenya. *FWU – Water Resources Publications Vol.3 (2005).* ISSN 1613 – 1045.

Abila, R., Marta Barluenga, Johanness Engelken, Axel Meyer and Walter Salzburger, 2004. Population –structure and genetic diversity in a haplochromine cichlid of a satellite lake of Lake Victoria. *Molecular Ecology* (2004). 13, 2589 – 2602.

Abila, R., 2002. Utilization and economic valuation of the Yala swamp wetland, Kenya. In: Gawler, M. (Ed.). Best practices in participatory management. Proceedings of a workshop held at the 2nd International Conference on Wetlands and Development, Dakar, Senegal. *Wetlands International. PP96 – 104. IUCN – WWF Publications No 65 Wageningen, The Netherlands.* ISBN 90 -1882 – 0084.

Aloo, P.A., 2003. Biological diversity of the Yala Swamp lakes, with special emphasis on fish species composition, in relation to changes in the Lake Victoria Basin (Kenya): threats and conservation measures. *Biodiversity and Conservation 12: 905 – 920, 2003*.

Avise , J.C., 1994. *Molecular Markers, Natural History and Evolution*. Chapman and Hall. 1994.

Bruford, M.W., Hanotte O., Brookfield, J.F.Y., and Burke ,T., 1998. Multilocus and single locus DNA fingerprinting. In. A.R. Hoelzel, ed. M*olecular genetics analysis of populations: A practical approach*. Oxford University Press, Oxford. pp 287 – 336

Greenwood, P.H., 1981. *The haplochromine Fishes of the East African Lakes*. Kraus International Publications, Munchen.

Guo, S.W. and Thomson, E. A., 1992. Performing the exact test for Hardy – Weinberg proportions for multiple alleles. *Biometrics,* 48, 361 – 372.

Hyslop, E.J., 1980. Stomach contents analysis – a review of methods and their application. *Journal of fish Biology 17: 411 – 429.*

Kaufman, L.S. and Ochumba, P., 1993. Evolutionary and conservation biology of cichlid fishes as revealed by faunal remnants in Northern Lake Victoria. *Conservation Biology, Vol. 7 No. 3, Sept. 1993. 719-730.*

Lesica, P. and Allendorf, F.W., 1995. When are peripheral populations valuabable for conservation? *Conservation Biology*. 9, 753 – 760.

Loiselle, P.V., 1996. 'Fulu' of the Yala Swamp – *Cichlid News*, 1996.

Maithya, J., 1998. A survey of ichthyofauna of Lake Kanyaboli and other small water bodies in Kenya: *Alternative refugia for endangered fish species. Naga, The ICLARM Quarterly. Jan.,* March 1998. PP 54 – 56.

Mbabazi, D. R. Ogutu-Ohwayo, S.B. Wandera and Y. Kizito, 2004. Fish species and trophic diversity of haplochromine cichlids in the Kyoga satellite lakes (Uganda). *Afr. J. Ecol. 42: 59 – 68.*

Moritz, C. 1994. Application of mtDNA analysis in Coservation: A critical review. *Molecular Ecology* (1994) 3, 401 – 411.

Namulemo, G., 1998. *Distribution, Relative Abundance, Population Structure and Food of Surviving Haplochromine Cichlids in Littoral Areas of Napoleon Gulf (Lake Victoria).* MSc. Thesis, Makerere University, Kampala.

Ogutu-Ohwayo, R., 1990. The decline of the native fishes of Lakes Victoria and Kyoga (E. Africa) and the impact of introduced species, especially the Nile Perch, *Lates niloticus*,and the Nile tilapia, *Oreochromis niloticus*.*Env. Biol. Fish. 27: 81-96.*

Raymond, M. and Rousset, F., 1995. GENEPOP (Version 1.2): Population genetics software for exact tests and ecumenicism. *Journal of Heredity*, 86, 248 – 249.

Schneider, S. Rossli, D and Excoffier, L., 2000. *ARLEQUIN, Version 2.000: A software for population genetic data analysis*: Genetics and biometry laboratory, University of Geneva, Switzerland. (Available at:http://www.anthro.unige.ch/arlequin).

Seehausen, O., Alphen, J.J.M., and Witte, F., 1997. Cichlid fish diversity threatened by eutrophication that curbs sexual selectin*. Science Vol. 277, pp 1808 – 1811.*

Seehausen, O., 2000. Explosive speciation rates and unusual species richness in haplochromine cichlid fishes: effects of sexual selection. *Adv. Ecol. Res. 31: 237 - 274*.

Soule, M.E., 1987. *Viable populations for conservation*. Cambridge University Press, New York.

Sumaila, U.R. Chuenpagdee, R and Vasconcellos, M. eds. (2000). *Proceedings of the INCO – DC International Workshop on Markets, Global Fisheries and Local Development*. ACP – EU Fish. Res. Rep. (7) pp 86 – 87.

van Oijen, M. J. P., 1982. Ecological differentiation among the piscivorous haplochromine cichlids of Lake Victoria (East Africa). *Netherlands Journal of Zoology. 32(3): 336 – 363 (1963).*

Weir, B. S., and Cockerham, C. C. (1984). Estimating F – statistics for the analysis of population structure. *Evolution*, 38, 1358 – 1370.

Wright, S., 1978. *Evolution and the genetics of population Vol. 4: Variability within and Among Natural Populations*. University of Chicago Press, Chicago, 1978.

Reduction of eutrophication of Lake Sevan (Armenia) via fishery management

Bardukh Gabrielyan [1] and Khosrovyan Alla[2]

[1]Institute of Hydroecology & Ichthyology of National Academy of Sciences of Armenia, 24d, Bagramian str., room 907, Yerevan 375019 Armenia. Email-gabrielb@sci.am
[2]Institute of Hydroecology & Ichthyology of National Academy of Sciences of Armenia, 24d, Bagramian str., room 908,Yerevan 375019 Armenia.

Abstract

Eutrophication of freshwaters is a common problem throughout the world. Various attempts are made towards its solution. The aim of the present work is to optimise fish yield from Lake Sevan (Armenia) providing the remaining stock to utilise its food base as maximum as possible. This approach is based on the assumption that planktivorous fish can utilise no more than 50% of the total edible zooplankton biomass.

In Lake Sevan, the commercial fish stock is whitefish (Coregonus lavaretus), which is also the main consumer of the zooplankton as 80-100% of its food base is zooplankton. This fact hints there could be a strong relationship between the fishery biomass and edible zooplankton biomass.

By means of VPA the whitefish spawning stock biomass and recruiter numbers are calculated; short-term prediction of the spawning biomass is done for the year concerned. To quantify the relationship between the fishery biomass (in this case, spawning biomass) and edible zooplankton biomass, statistical analysis is performed that has resulted in the significant relationship between adult whitefish biomass and large crustacea species (Arctodiaptomus bacilifer) biomass. Based on this result it was determined the fishing of 20% of the fish total spawning biomass may lead to desirable 50% consumption of the crustacean by the remaining stock. For maintaining the reproductive ability of the whitefish population is it necessary to sustain its fishery biomass to at least 50% level; in our case, predicted catch amounted only to 20%. This is considered as optimal for two reasons: whitefish stock in Lake Sevan is almost depleted by overfishing and maximum possible organic matter is removed from the lake. Deterioration of conditions may lead to adverse effect on the whitefish population. Thus, to keep fishing pressure at 20% level is reasonable to ensure stable recruitment and recovery of the population.

Keywords: whitefish, zooplankton, eutrophication

Introduction

The Republic of Armenia is one of the countries of the South Caucasus region with extremely limited water resources, therefore rational use of water is of primary importance. There are more than 100 small lakes and reservoirs, with a total volume of 35.8 billion m^3 (Hunanian, 1990)**.** These reservoirs are generally shallow, with an average depth of less than 10 m. Lakes and reservoirs are mainly replenished by runoffs and snow melts. The largest reservoir of Armenia, indeed the whole South Caucasus, is Lake Sevan. It is the only large source of fresh water, which is extremely important for the economic development of the country.

Prior to reduction of water level Lake Sevan was located 1916.2 m above sea level, had large water volume (58.4 km^3) and hosted valuable endemic species. Trace concentration of mineral nitrogen and relative high mineral phosphorus concentration had accompanied the oligotrophic status of the lake. However, intensive use of water for development of the economy of Armenia since 1930 led to a fall in water level of more than 19m. This, in turn, stimulated changes in the physico-chemical and biological composition of the lake's ecosystem, resulting in eutrophication (Hovhanissian, 1994)

Consequently, the natural reproduction and food provision of some fish has been affected, in particular the spawning grounds of the trout were lost. In the 1990s, overfishing depressed the main commercial whitefish stock (more than 80% of the total catch) (Gabrielyan, 1998) which is the main consumer of zooplankton (Simonian, 1991). This promotes accumulation of organic matter in the lake increasing its trophic status.

Therefore, currently the problem of decreasing of the lake's trophic level has arisen, for solving which an integrated approach is required primarily an increase of the water level and control of the nutrient input from the catchment. However, many resources are needed for these measures to be implemented and technical solutions are very complex; meanwhile, other affordable measures could be applied to slow down eutrophication such as reduction of organic matter in the lake.

Thus, the aim of this study is to optimise fish yield from Lake Sevan (Armenia) so that let the remaining stock to utilise its food base as maximum as possible.

Proceeding from the aim the following tasks are set:

- estimate whitefish stock and forecast catch;
- find statistical relationship between whitefish stock and different zooplankton species as being the food factor for the fish;
- predict an optimal catch so that the remaining stock maximally utilises its food base
- The hypothesis in this work is to test if a strong correlation is found between zooplankton and whitefish biomass in different periods of eutrophication. If found, this relationship will be used to establish how a reduction of organic matter in form of unutilised by fish zooplankton

biomass can be achieved through regulation of whitefish biomass, i.e., fishery management.

The importance of the work is that:

a) it attempts to apply a biomanipulation technique to a large lake;

b) it is the first attempt to reduce organic matter (zooplankton biomass) in Lake Sevan via biomanipulation;

c) biomanipulation implies direct regulation of zooplankton biomass by fishery in the absence of piscivore species.

Eutrophication of Lake Sevan has brought about several changes in its ecosystem. Comparison of the current ecosystem concerned to its previous state - before reduction of the water level is required to reveal main factors and trends of eutrophication, changes in the fish population and fishery structure with respect to zooplankton species and biomass change are to be examined as well. This may suggest to look for the quantitative expression of the relationship between those two pursuing reduction of organic matter through regulation of the fish stock of the lake. Literature on the given problem, available in Armenia is studied comprising local and international cases. The theme of biomanipulation for Lake Sevan is fairly new, and biomanipulation techniques throughout the world are usually applied to small shallow lakes and not to big ones. Such technique supposes impact ing one trophic level by the other, however in the present work, the similar effect is proposed to be reached by direct fishery management (selective predation) that would affect fish food base - zooplankton. No explicit analogies were found while examining literature on the given issue.

Materials and methods

All raw data for this study are provided by the Institute of Hydroecology and Ichthyology (IHEI) of NAS Armenia. The data span is from 1979-1999 and includes annual average zooplankton biomass on Small Sevan pelagic, annual average zooplankton biomass by species, whitefish biological and population parameters and fishing data – weight and length of the fish by age, age structure, catch-at-age and natural mortality, fishing effort for different fleets (purse seine nets and gillnets) and landings.

Data are grouped for 3 periods: 1979-1985, 1986-1991 and 1992-1999. Such a grouping is explained by the difference of the trophic level of the lake: 1979-1985 after the peak of eutrophication, 1986-1991 - relative stabilised water level and less productive period of the lake, and 1992-1999 secondary eutrophication caused by intensive water abstractions (IHEI 1998).

It should be noted that, during the analysis, data from official sources have been used. Those data have been multiplied with a coefficient of uncontrolled fishing which was determined based on special experts' estimations together with inspectors of the National Park "Sevan", who are responsible for protection of the fish stock in the lake. The special experts' estimations are often used in cases when initial data do not conform or have discrepancies. This method is based on professional expertise and intuition, therefore it carries some element of subjectivity, but together with other methods and estimations it is reliable.

Only severe fluctuations in fish populations and climatic conditions essentially influence the growth rate of the fish and their natural mortality. Therefore, for this work, the coefficients of natural mortality for each age group are taken as constant for the whole period being studied (IHEI 1999).

Catch-at-age analysis is conducted to determine the dynamics of whitefish stock, as well as to forecast the catch by the help of the program Lowestoff VPA Suite, Version 3.1. (Darby and Flatman 1994) that has been incorporated in the International Council for the Exploitation of the Sea (ICES) Standard Assessment software package. For assessment of whitefish stock in Lake Sevan there are two methods, (Laurec-Shepherd tuning and Extended Survivors Analysis (XSA)) with different shrinkage, the most appropriate one is selected through retrospective analysis examining the bias in estimates derived from them.

Each method estimates fishing mortality and numbers at age in a stock using data on given catches and estimates (or assumed values) of natural mortality. Given appropriate weight and maturity-at-age data sets, the program will calculate stock and spawning biomass at the beginning of the year or at the spawning period. The minimum requirements for a run are the catch-at-age and natural mortality files.

The software also produces short-time prediction output with the estimated numbers of spawning stock for the predicted year.

Retrospective studies have established that patterns of consistent under or over-estimation bias in estimates of F (fishing mortality coefficient) and population numbers-at-age can be produced by the application of assessment methodologies to fish stocks data. Such biases may cause problems in the recommendations given to managers and therefore, need to be examined and, if possible, removed from the assessment and subsequent predictions. It was established that the degree of the bias could be reduced by introduction of shrinkage to the mean F in the assessment packages.

Retrospective series can be used to investigate the impact of particular assessment parameters (e.g. shrinkage to the mean F) on the accuracy and bias of the terminal year estimations. Retrospective runs are performed with a range of values for the selected parameters (all other parameter values are held constant), and the value producing the "best" retrospective pattern is chosen as the optimum value for assessment of a particular stock.

Thus, the program output includes recruitment numbers, total spawning biomass, and total biomass by year (from 1979 to 1999). Also, short-term prediction of the numbers of spawning stock-at-age (for 2000) is done. By multiplication of these numbers to the fish corresponding weights from the final year and summing them up one can get an estimation of the spawning biomass for the predicted year.

Most whitefish in Lake Sevan mature by age two, reaching commercial size by that time, so in this case spawning stock and fishery stock can be used interchangeably.

After obtaining spawning biomass estimate for year 2000, regression analysis is done by using MS Excel to establish significant relationship between zooplankton species biomass and total spawning biomass, as well as recruitment numbers to reveal feeding pattern of the fish. By extrapolation of the regression equation produced through the analysis one can estimate the zooplankton biomass appropriate for the predicted for stock.

Here, two assumptions are made:

1. maintaining reproductive ability of a whitefish population requires 50% of its total fishery stock to be sustained;
2. planktivorous fish can utilise no more than 50% of total edible zooplankton biomass.

Thus, in order to minimise the level of organic matter in the lake in form of zooplankton biomass and maximise whitefish yields, the following two equations must be satisfied:

- total allowed catch for the predicted year < = 50% of the total spawning biomass and
- edible zooplankton biomass must approach to 50% of the total edible zooplankton biomass to maximise utilisation by the fish.

The desirable zooplankton biomass (ideally, 50% of the total edible biomass) can be estimated by interpolation of the regression equation obtained which also evaluates the fish biomass corresponding to it. Fish catch for the predicted year will be estimated based on total spawning biomass for the predicted year and spawning biomass from the regression equation. This is the optimal catch estimate that takes into account fish food base factor and ensures that the remaining stock will be able to utilise its food base as maximum as possible.

Results

To calculate total spawning biomass, total biomass and recruitment numbers, as well as to estimate these parameters for the predicted year (2000) using Lowestoff VPA Suite software, the following files are required: landings, catch-at-age, catch weight-at-age, stock weight-at-age, natural mortality, maturity-at-age.

Table 1. Dynamics of main population parameters of Whitefish.

YEAR	RECRUITS NUMBERS (Thousands) (Age 1)	TOTAL BIOMASS (tones)	TOTAL SPAWNING BIOMASS (tones)
1979	10164	24091	20567
1980	11479	16091	11891
1981	13011	11624	7394
1982	16656	9824	5208
1983	20845	10955	5671
1984	16420	8272	5539
1985	30113	9679	4895
1986	33354	10281	6319
1987	29200	17937	10250
1988	67421	21876	11053
1989	45980	30564	15977
1990	36986	26705	16729
1991	20613	27399	18226
1992	10322	16904	10994
1993	10598	9414	6648
1994	61672	14542	4486
1995	43556	14782	5700
1996	41246	13860	5396
1997	28966	13230	5964
1998	11936	7424	3688
1999	38999	7538	2252

Table 2. Total annual zooplankton biomass by species by year.

	Arctodiaptomus bacilifer (mg/m^3)	*Cyclops strennus* (mg/m^3)	*Acanthodiaptomus denticornis* (mg/m^3)	*Daphnia longispina* (mg/m^3)
1979	508.7	75.2	61.1	204.4
1980	254.2	134.2	160.3	416.6
1981	119.2	145.4	100	238.8
1982	153.3	342.6	63.7	138.1
1983	0	175.2	48.3	158
1984	18.8	53.4	103.4	134.9
1985	18.3	90.5	49.6	183.5
1986	20	60	30	190
1987	20	110	30	390
1988	60	60	50	250
1989	50	50	10	100
1990	80	60	70	400
1991	170	190	60	330

Source: Simonian (1991)

To choose between two assessment methods - Laurec-Shepherd tuning and XSA retrospective analyses are used introducing shrinkage to mean F. Two levels of the shrinkage are suggested by the software for each of the methods: 0.2 and 0.5. It was found that shrinkage 0.2 improves the prediction accuracy of the final year mean fishing mortality coefficient values for XSA by reduction of the bias in the F estimate. Thus, retrospective runs with shrinkage 0.2 in XSA showed less variation in the estimates. Related to this, data from XSA run with shrinkage 0.2 are used in the work. The output of the program is given in Table 1.

For finding a relationship between total spawning biomass of whitefish (TSB) and total annual biomass of different zooplankton species, as well as between the latter and recruiter numbers (R) correlation-regression analysis on Microsoft Excel was run.

Four mass species of zooplankton are considered: from cyclopoids *Cyclops strennus*, from calanoids – *Acanthodiaptomus denticornis* and *Arctodiaptomus bacilifer*, from cladocerans *Daphnia longispina***.** Table 2 shows the total annual plankton biomass by species in various years. All species are the preferable food for the fish (Simonian 1991).

Correlation-regression analysis is performed using MS Excel simple linear regression model described by Pelosi and Sandifer (2000) and McClave *et al*. (1998). The model produces an output, which provides with regression equation and analysis of goodness of fit of the model.

As the model includes the error member (y=b+ax+e), we assume that error term has 0 mean value; distribution of errors is normal; for every value of x the standard deviation of error term is the same; the error terms for different observations are independent on each other. Such assumptions do not affect significance of the model.

As regression equation is somehow a “mechanical” result and might exist without underlying cause-effect sequence there is a need to check how much the relationship described by it is meaningful. Thus, the model also includes tools for such analysis:

Residual Output (differences between observed and predicted from equation values); they must be randomly distributed around the mean of 0 with no apparent pattern indicating that the linear model is appropriate,

Standard Error of the estimate (measure of how much the data vary around the regression line); the smaller the error, the smaller the variation of the estimate in different samples,

P-value for the estimated coefficient – precise measure of statistical significance of the given estimate, can be used instead of t-statistics when testing hypothesis about the significant linear relationship between two variable - Ho: slope is zero (if P-value is less than confidence level reject Ho); the lower it is for the estimate, the more confident is the estimate; P-value of .05 or less confirms that the coefficients is statistically significant at 5% level, **correlation coefficient (r)**, which tells that a linear trend may exists between two variables, obtained as **square root of R Square**, coefficient of determination **R-Square**, which tells what percent of the “sample variation in y can be explained by using x to predict y in the straight-line model” (McClave *et al*. 1998), indicate the overall fit of the regression line,

Adjusted R-square – measure of goodness of fit of the regression line; little difference between this and R-Square confirms that high R-Square value is not attributable to “excessive number of estimated coefficients relative to the sample size”, so it penalizes for “estimating numerous coefficients from relatively few observations” by decreasing its value (McClave *et al*. 1998),

Significance F - statistical significance of f-statistic from **ANOVA** output (also a measure of goodness of fit alternative to R-Square and adjusted R-square as it is difficult to say how much they should be to indicate a good fit, measures the total variation explained by the model relative to unexplained variation). Regression is significant if Significance F level is less than significance level adopted for the model. Generally, regressions with this statistics of .05 or less are considered statistically significant.

Intercept is the free term in the regression equation,

X Variable 1 is coefficient of the explanatory variable in the mentioned equation,

X Variable 1 Line Fit Plot tells how good is the fit, in the perfect case, predicted and observed (or fitted) values coincide with each other.

When non linearity is observed in the data plot, one can take logarithms of two variables before regressing and then obtain parameter estimates of the regression equation for the log-values. Then, mathematical transformations can be done if necessary in order to obtain y value.

In the present work:

a) significance level is 5%;

b) most whitefish species of Lake Sevan mature at age 2, reaching commercial size at the same time, so spawning and fishery stock can be used interchangeably;

c) to affect zooplankton biomass by regulation of commercial stock (in this case, TSB), the former is taken as dependable variable, and the latter as explanatory one.

For analysis between Recruiter Numbers (R) and zooplankton species, the former is used as explanatory variable, the latter as dependable one; such dependability is stipulated by the effect the numbers of recruitment exerts on zooplankton biomass.

The regression analysis conducted showed the following trends:

When looking for the relationship between certain species of zooplankton and fish biomass such as zooplankton-TSB and R-zooplankton, no significant correlation is found between *Daphnia longispina* and fish, as well as between *Acanthodiaptpmus denticornis* and fish in 1979-1985 and 1986-1995.

In contrast, analysis for *Arcodiaptomus bacilifer*-TSB for 1979-1985 reveals the P-value=0.00075 or the same Significance F value (less than 0.05) indicating that the model is highly significant at 95% confidence level, i. e. there is a relationship between them, and that relationship is linear (r=0.95). Also, R Square=0.9 indicates that 90% of variation in *Arcodiaptomus bacilifer* sample is explained by the regression, and Adjusted R Square =0.9 indicates that there is no effect of using only 7 observations for one independent variable.

Looking at the Residual Output one can see that the assumptions of the simple linear model are held: residuals are randomly distributed with no particular pattern. Line Fit Plot shows high amount of coincidences of predicted and observed y values.

Looking at the Intercept and X variable one can obtain the following equation:

y=-110.3+0.03x applied for the given period of time.

Thus, the dependence between TSB and *Arcodiaptomus bacilifer* is evident and significant.

For R and *Arcodiaptomus bacilifer* in 1979-1985 the regression shows a significant relationship between logarithm of R (R-log) - independent variable and the plankton – dependent one, as shown by P-value=0.037 which is less that 0.05. Correlation coefficient r=-0.78 shows 78% linearity in this relationship, R Square=0.61 indicates that 61% of variation in the plankton biomass is attributable to the recruitment numbers in this period, and Adjusted R Square=0.53 is only slightly different from R Square and might be explained by using only 6 observations in the model compared with 7 observation for the preceding period. Line Fit Plot shows good coincidence of predicted and fitted data and Residual Output shows adherence to simple regression assumptions.

Looking at the Intercept and X variable one can obtain the following equation:

y = -381.64Ln(x) + 3846.

So, there is also significant dependence between R and *Arcodiaptomus bacilifer* in 1979-1985.

In 1986-1991, the situation slightly changed:

Analysis has shown that the regression model for the indicated period is highly significant (P-value =0.027) for the logarithm of *Arcodiaptomus bacilifer* as a dependent variable vs TSB as independent. R Square=0.74 with Adjusted R Square=0.67, with correlation coefficient r=0.86. Line Fit Plot shows good coincidence of predicted and fitted data and Residual Output shows adherence to simple regression assumptions. Looking at the Intercept and X variable one can obtain the following equation: ln(y)=1.89+0.0002x, by transforming to the direct y value the following equation is obtained: $y=6.6208e^{0.0002x}$.

So, one can conclude that significant *Arcodiaptomus bacilifer* - TSB relationship continued to take place in 1986-1981.

As to recruitment stock, no significant correlation was found between R and *Arcodiaptomus bacilifer* (r=-0.33). This testifies to independence between those two populations.

The other regression model between *Cyclops strennus* and R for 1986-1995 (was applied to explore a possible relationship, but it showed no highly significant linear relationship, as P-value=0.058 indicating the model did not perform well in rejecting the null hypothesis of no relationship. Significance F=0.058 shows lower degree of the overall fit of the regression line. However, negative correlation coefficient r=-0.79 gives testimony to 79% of linearity in this relationship and may indicate a shift of food preference of recruiters from *Arcodiaptomus bacilifer* to this plankton. R Square=0.63 indicates that 63% of variation in the plankton could be explained by the changes of the recruitment numbers. Residual Output does show some pattern on the plot indicating violation of assumptions of regression model. Adjusted R Square=0.54 indicates there must be something that influences the relationship between those to variables. This may be caused by small sample size (few observations) or necessity to incorporate more variables, in other words, further investigation may be appropriate.

No significant correlation was shown between *Cyclops strennus* and TSB.

No data on zooplankton species are available after 1991, only average annual zooplankton data from 1979-1997 (Table 3).

Table 3. Average annual zooplankton biomass by years.

Year	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997
Average Annual Zooplankton Biomass (mg/m^3)	1060	1040	720	700	470	390	580	710	760	550	270	830	1100	1440	1600	2200	1370	600	1600

Source: Simonian (1991); IHEI (1999)

Therefore, an attempt was done to estimate the relationship between average annual zooplankton biomass (Z) and recruitment numbers and total spawning biomass. Table 4 shows the results of application of the Excel correlation function.

Table 4. Correlation coefficients for different periods.

	1979-1985	1986-1991	1992-1997
Z-R	-0.63 (-0.78)	0.11 (-0.79)	0.26
Z-TSB	0.83 (0.95)	-0.56 (0.86)	-0.34

(in parentheses, the highest correlation coefficients found for particular zooplankton species from the above mentioned analysis are given for comparison).

The only strong correlation is found between Z and TSB in 1979-1985, that is highly significant at 95%, i.e. based on P-values and Significance F value we reject null hypothesis of zero slope, i.e., there is indeed a significant relationship between these two.

Despite a significant regression line obtained for recruitment numbers and *Arcodiaptomus bacilifier* in 1979-1985 (with r=-0.78), the regression model for average annual values did not show any significant correlation.

The regression analysis between average annual zooplankton and recruitment in 1986-1991 also does not reveal a significant relationship.

No data on zooplankton species are available after 1991. For the purpose of establishing a numerical relationship we assume that the current trophic status of Lake Sevan is similar to the state of 1979-1985. For prediction of edible zooplankton biomass for 2000 the regression equation obtained for TSB – *Arcodiaptomus.bacilifer* in 1979-1985 is used. Edible zooplankton is considered to be *Arcodiaptomus bacilifer.*

To calculate spawning stock biomass for year 2000 numbers of the fish in that stock by age given in Appendix 1 must be multiplied by their corresponding weights and then totaled, resulting in 6385 tonnes.

By substituting this number into the independent variable (x) in the regression equation of the mentioned period y=-110,3+0,03x, edible zooplankton biomass amounts to y=81.25 mg/m^3 for year 2000.

The desirable zooplankton biomass to be maximally utilized by the fish - 50% of the total edible zooplankton biomass, in our case: 81.25/2=40.62 mg/m^3. From the regression equation, the corresponding fish biomass amounts to 5030 ton.

Thus, in order to provide maximum utilization of edible zooplankton, the spawning stock is estimated to be 5030 ton, so fish yields must constitute only 6385-5030=1355 tonnes in 2000.

Compared with the landings since 1979 (Table 5) this suggested number is less than in any of those and constitutes only 20% from the total spawning biomass.

Table 5. Whitefish landings by year.

Year	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
Landings (tones)	2530	3285	3402	2288	2587	4017	2776	2840	4881	5292	5688	5982	7192	5090	4025	5612	7684	6020	6800	4800	2800

Discussion

There are many works in the literature describing relationship between various trophic levels in freshwater ecosystems. Different biomanipualtion techniques are elaborated depending on the particular need of the reservoir. However, this project differs from the others by attempting to manipulate one trophic level by the other in terms of a relative big lake such as Lake Sevan.

The lake ecosystem encompasses relative simple trophic relationships and lacks piscivore fish species. However, it has undergone significant trophic changes in relatively short period of time. This allows one to consider those interrelationships for different trophic states of the lake.

The simplicity of the ichthyofauna of the lake with, a main commercially valuable fish species - whitefish, allowed the possibility of regulation of the zooplankton biomass via fishery management to be considered. Lehtonen (2000) showed that zooplankton populations in the lakes, with planktivorous fish being at the top of the trophic chain, are controlled by fish predation. Thus, the regression analysis is done to establish the numerical relationship of the link between fish - zooplankton for different periods of eutrophication to reveal strong dependability between those two. This may be the limitation some zooplankton species exert on the fish population as the main food base for them.

In 1978-1981, *A.bacilifer* was one of the main components of the zooplankton in the lake based upon the numbers present all year-round. The peak numbers occurred in 1979 after the peak of eutrophication. However, in 1983, it practically disappeared from the planktonic community of the lake. There were no regularity in seasonal dynamics of its biomass in 1937-1985 and that might have been caused by unstable age structure of the population (Simonian 1991).

As this is widely available large crustacean species in the lake, statistically significant correlation was found between *A.bacilifer* and whitefish of various ages in 1979-1985, when no correlation at all with other plankton species were established. This might be explained by consumption patterns of the fish, i.e., this zooplankter was the food component of the fish as adult, so recruiters.

Mass kills of the adult fish were observed in 1983-1984 because of worsening food conditions after the peak of eutrophication resulting from reduction of the food base for whitefish - starvation, empty guts as reported by Simonian (1987). That coincided with disappearance of *A.bacilifer* in 1983 and may testify to the limitation this species exerted on whitefish adult populations.

Disturbance of the fish population caused by the increased mortality and intensified fishery at that time have changed the situation towards its recovery as the remaining stock could satisfy its food requirements.

In 1986-1991, a strong significant correlation between adult whitefish and *A.bacilifer* again testifies to this zooplankter continuing to be the limiting food for them, as again no significant correlation with other plankton animals is found. However, recruiters changed their food preferences and possibly shifted to *C.strennus*. This might be explained by continuing instability of age structure of the *A.bacilifer* population resulted in switching of the recruiters to more accessible food starting 1985 until the end of the period studied.

Based on the only quantitative relationship found between spawning biomass and *A.bacilifer* as a preferred food component for the spawning stock the year 2000 catch was predicted. It was found to be 1355 tones which is only about 20% from the total spawning biomass. This prediction is found to be optimal:

- from the fishery standpoint: the spawning stock is depressed by overfishing; it is difficult to say what percent of the recruiters will reach commercial size and contribute to the fishery stock in predicted year, even if in 1999 there was sharp increase of the recruiter numbers, so it is risky to increase the estimated catch;
- from the ecosystem standpoint: (ensure maximum possible utilisation of edible zooplankton by the fish).

Extrapolation of obtained equation beyond the period investigated gives a possibility to forecast catch taking into consideration ichthyological requirements for the fishery as well as maximum utilisation of its food base. It is obvious that such a "mechanical" extrapolation assumes identical processes and conditions to happen in the lake. Unfortunately, lack of data on zooplankton species starting from 1991 did not allow such an analysis to be conducted for the end of the 1990s and therefore extrapolation is based on the assumption of unchanged conditions.

However, this work can be considered as a new methodical approach to finding solutions to eutrophication problems of freshwater ecosystems. If necessary data are available such approach can provide real results for the fishery management based on the requirements of the fish food base.

Thus, in the context of the estimations made by use of the statistical software the following conclusions are derived:

1. from 1979-1985, the preferred food for whitefish was *Arcodiaptomus bacilifer;* in 1986-1991, adult fish still depended on the plankton, recruiters might have changed their preferences and shifted to other species including *Cyclops strennus*;

2. starting 1990s, correlation-regression analysis does not reveal any relationship between the fish and average annual zooplankton biomass, thus indicating that there was no limitation of the food base for whitefish;

3. assuming that the current trophic status of Lake Sevan is approximately similar to that of the period studied and the fishery pattern does not change, the total spawning biomass of whitefish is estimated from the corresponding regression equation and amounts to 6385 tones in 2000;

4. taking into account that the predicted catch is only 20% of the total spawning biomass, it is considered to be optimal after years of depression of the whitefish stock both from a fishery standpoint (to avoid depletion of the fishery stock in any unfavourable condition for growth of the recruitment created in the predicted year) and from an ecosystem standpoint as a measure to sustain lower level of organic matter in the lake (in the form of organic matter regenerated by zooplankton) via maximum utilisation of it.

References

Darby, C. D., and Flatman, S., 1994. *Lowestoft VPA Suite. Version 3.1 User Guide*.

Gabrielyan B. K. 1998. *Dynamics of Population Parameters of Coregonus lavaretus Related to Eutrophication of Lake Sevan (Armenia)*. Paper read at 20th session and Symposium on Water for Sustainable Inland Fisheries and Aquaculture, 23 June - 1 July 1998, EIFAC, Algarve, Portugal.

Hovhanissian, R. H., 1994. *Ozero Sevan vchera, segodnia.* [Lake Sevan yesterday, today...]. Gitutun Yerevan.

Hovhanissian, R.H., Gabrielyan, B. K., Rubenian, A.R. & Rubenian, T.G. 1998. *Introdukcia cheshujchatogo karpa w ozero Sevan* [Introduction of Carp Cyprinus carpio in Lake Sevan]. *Paper read at Conference on Nature Protection Issues,* 20 October – 23 October 1998, Academy of Sciences, Yerevan.

Hovhanissian, R. H. and Gabrielyan, B. K., 2000. Ecological problems associated with the biological resource use of Lake Sevan, Armenia. *Ecological Engineering* 16(2000), 175-180.

Hunanian, N. A., 1990. *Ispolzowanie i okhrana vodnikh resursow Armenii* [Armenia Water Resourses Use and Protection]. Izd. Akad. Nauk Armenii, Yerevan.

Institute of Hydroecology and Ichthyology of NAS Armenia (IHEI), 1998. *Annual report.* Yerevan.

Institute of Hydroecology and Ichthyology of NAS Armenia (IHEI), 1999. *Annual report.* Yerevan.

Lehtonen H., 2000. Fish as components of the lake ecosystems. *In.* P. Heinonen, G. Ziglio & A. Beken, eds. *Hydrobiological and Limnological Aspects of Lake Monitoring*. 132-142. John Willey & Sons, Chichester.

McClave, J.T., Benson, P.G. & Sincich, T., 1998. *Statistics for business and economics*. 7th ed. Prentice Hall, New Jersey.

Simonian, A. A., 1987. Zooplankton pelagiali kak kormovaia baza sevanskogo siga (1972-1984) [Zooplankton as Food Base for Sevan Whitefish (1972-1984)]. *In.* V. A. Avagian. ed. *Nature, city, man*. 144-146. Haiastan, Yerevan.

Simonian, A. A., 1991. *Zooplankton ozera Sevan* [Zooplankton of Lake Sevan]. Izd.Akad.Nauk ArmSSR, Yerevan.

Distribution and association of Tilapine unit stocks in the Lake Victoria catchment (Kenya)

Jembe B.T.[1], Boera P. N.[1] & Okeyo Owuor J.B[2].

[1]Kenya Marine & Fisheries Research Institute, P.O. Box 1881, Kisumu. Email-tsuma_jembe@yahoo.com; pboera@yahoo.com, viredresearch@hotmail.com

[2]Moi University, School of Environmental Studies, Biological Sciences Division, P. O. Box 3900, Eldoret.

Abstract

The Lake Victoria Catchment (Kenya) harbours six species of tilapia. Two species *Oreochromis variabilis* (L) and *Oreochromis esculentus* (L) are endemic while four species *Oreochromis niloticus*, *Oreochromis leucostictus*, *Tilapia zillii*(Gervais) and *Tilapia rendalli*(Gervais) were variously introduced in 1950s and early 1960s. The existence of six tilapia species in the same geographical range has had significant ecological and economical implications.

The study investigated the ecological impacts by determining species diversity, distribution, association (allopatric, sympatric and parapatric) and habitata preferences for the six-tilapia species. Shannon-wiener index was used to determine diversity, while species distribution was evaluated on the basis of 39 habitats (dams and satellite lakes) within the catchment. Species associations were determined using Cole's Cab index. The Shannon Weiner function values of 2.44-2.7 indicate relatively high species diversity in 15% of 33 habitats where Tilapia were resident.

Cole's Cab indices showed significant relationships in *O. esculentus* X *O. variabilis, O.leucostictus* X *O. esculentus, O. niloticus* X *O. variabilis and O. niloticus* X *O. leucostictus* at P < 0.05. Habitat preference for the six species is provided.

Key words: Association, Distribution, Diversity

Introduction

The tilapiine inventory of the lake Victoria catchment , Kenya, consists of four *Oreochromis* and two *Tilapia* species. The existence of these tilapiine species in the Lake Victoria basin (Kenya) has significant ecological and economical implications. Two of the endemic tilapiines (*O. esculentus* and *O. variabilis*), which once dominated the fishery, are now relegated to the periphery. Their impact on the ecology and fishery economy is now insignificant. Dominance of two introduced tilapiines (*Oreochromis niloticus* and *O. leucostictus*, is now a reality, with *O. niloticus* contributing up-to 90% of the tilapiine catches (Fisheries statistics, 2003).

Establishment of viable populations by the six tilapiines within the Lake Victoria basin (K) is unknown, yet it is appreciated that *O. niloticus* is widespread and has swamped and dominated the other tilapiine species. Lake Victoria Environmental Management Project (LVEMP reports, 2000) surveys have established that extant populations of the endemic tilapiines still exist. These isolated extanct populations living in different habitats associate sympatrically, parapatrically or allopatrically with other tilapiine species. Interbreeding amongst tilapiines has been reported severally (Lowe, 1958; Jembe *et. al*., 1998) thus raising the possibility of a changing genetic status of the different populations. The introductions of alien species were intended to cover gaps in the ecological niches of Lake Victoria. *O. niloticus* and *O. leucostictus* are omnivorous (Greenwood, 1966; Witte and Van Densen, 1995) while *Tilapia zillii* is herbivorous (Lowe, 1958). They were all introduced during the 1950's to improve the fishery. The introduced tilapiines were once said to have similar ecological requirements to native tilapiines (Fryer, 1961). *O. niloticus* seems to have competitive advantage over all the other tilapiines and this seems to have contributed to its success. It grows to a larger size, has faster growth rate, is more fecund, has a longer life span and wider food spectrum, and is less habitat restricted (Lungayia, 1991). Further evidence on inter-specific competition is reported by Fryer (1961) between *O. variabilis* and *T. zillii*. With competition, the distribution and association of tilapiines in the various water bodies in Lake Victoria is expected to conform to niche characteristics and optimal species combination. In these combinations, optimal utilization of resources is achieved through resource partitioning. Trophic representation in many other species aim to achieve this objective. In tilapiines, stocks of herbivores, omnivores and phytoplanktivores achieve the partitioning of resources. While *O. niloticus* is an omnivore, *T. zillii* and *O. leucotictus* are herbivores (Greenwood, 1966, Witte and Van Densen, 1995). *O.esculentus* and *O.variabilis* are phytoplanktivores. The food of *T .rendalli* consists of macrophytes and detritus.

Resultant demographic changes in tilapia populations have great consequences on catches, genetics and sustainability. Between 2001 and 2003 tilapiine distribution was sampled on eight occasions and mapped. The study provided data on site characteristics and an assessment of species composition, spatial distribution, diversity and associations. Factors affecting distribution of the six tilapiines is also discussed.

Materials and methods

Study area

The study area showing sampled dams and lakes is given in figure 2. Their sizes, hydrological conditions and origin vary. Sizes range from 6,000 m^2, (Kamasawa) to 25,000 m^2 (Koketch) (LVEMP reports, 2000). While seasonal streams drain some dams, the water budget of others is solely dependent on surface run-offs. Some dams such as Kamasawa are replenished by springs. Most of the dams are man-made with an exception of a few which are natural entrapments of water. Some of the dams are temporary and were not included in the sampling regime. Dams short-listed for this study were 39 some of which are shown in figure 2 below.

Sampled areas in Lake Victoria to determine distribution of tilapiine species are also provided in figure 2. The selection covered littoral areas, (including river mouths), deep-water areas (>5 m), open water areas and sheltered bays. Other ecotones sampled included those with mud, gravel, rocky or sandy substratum. Areas with rocky outcrops were also included.

Four satellite lakes were sampled in this study; Lakes Kanyaboli, Sare, Simbi and Nyamboyo.

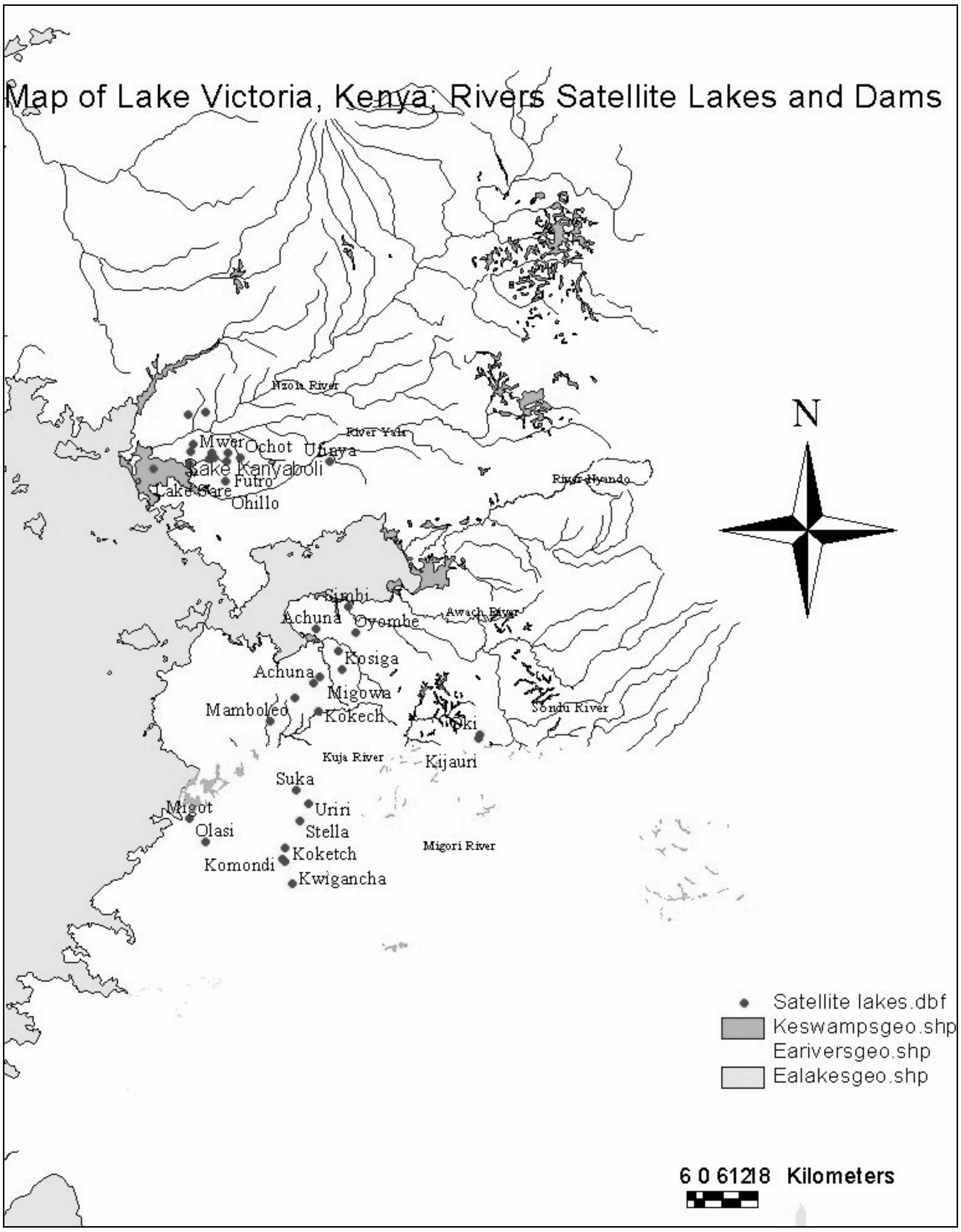

Figure 1. Map of Lake Victoria showing study area and sampled habitats.

Sampling

Each of the 39 dams and 3 satellite lakes was sampled twice during the study period. Sampling was done once during the wet and once during the dry season so as to accommodate physico-chemical attributes. Lake Victoria sites were sampled four times during the study period. All proposed ecotones were sampled.

Trawling was carried out in the deeper ecotones of L.Victoria while gillnetting was done in shallow areas and rocky outcrops. Catches from fishermen were also assayed to accommodate composition missed out during the experimental gillnetting and trawling exercises. While gillnetting was the main method of sampling dams and satellite lakes, seining was occasionally carried out whenever conditions permitted, as was the case with Lake Victoria. Identification tilapias was on a key by Greenwood (1966). Total length (TL) to the nearest 0.1 cm of each fish was recorded. A note was also made of aquatic and semi-aquatic plants for description and comparisons of tilapiine habitats.

Data analysis

The geographical location of each habitat sampled was identified using a (Geographic Positioning System (GPS) coordinates. Counts of individuals in each species were used to tabulate diversity while frequencies for each species were obtained from occurrence. Spatial distribution for allopatric, sympatric and parapatric populations was determined by plotting their positions on a map.

The occurrence of associated tilapiine species was analysed following the method of Mountford (1962) and expressed in the form of a three-dimensional diagram. Data for length frequency analysis was pooled from trawl and gillnet catches. Length frequency distribution was used to express community population structure in the various habitats. Habitat evaluation was based on substratum and macrophyte types.

Results

Lake Victoria had two tillapiine species, Oreochromis niloticus and Tilapia rendalii. Three satellite lakes (Kanyaboli, Nyamboyo and Sare) had three species, O. Niloticus, Orechromis esculentus and Tilapia zillii in varied combinations. Lake Simbi had no fish. 32 of the 39 dams sampled were inhabited by tilapiines. All the six species were variously represented.

Of the 39 dams sampled, 32 were inhabited by tilapiines, 5 were inhabited by non-tilapiines, while 3 had no fish. Three of the satellite lakes had tilapiines and other fish taxa, while one (Lake Simbi) had no fish. All six tilapiine species variedly inhabited the satellite lakes and dams. Lake Victoria had only two tilapiine species. The distribution patterns of other taxa per habitat category revealed a total of 15 families represented by 26 genera in the non-cichlid category. In the cichlid family category, the total genera were 19.

Spatial distribution of tilapiine species

Introduced species were represented by four populations of *T. zillii*, seven of *T. rendalli*, sixteen of *O. leucostictus* and eighteen of *O. niloticus*. Populations of endemics were nine for *O. esculentus* and sixteen of *O. variabilis*. The least commonly occurring tilapiine was *T. zillii* which occurred in Lake Sare, Kalenjuok (4), Ochot(32) and Kitaru (33). *T. rendalli* was more diverse found in Lake Victoria, Ochillo (1), Gesebei (35), Kokech (23), Kijauri (36) and Mamboleo (25). The percentage occurrence of each of these species is given in Figure 3

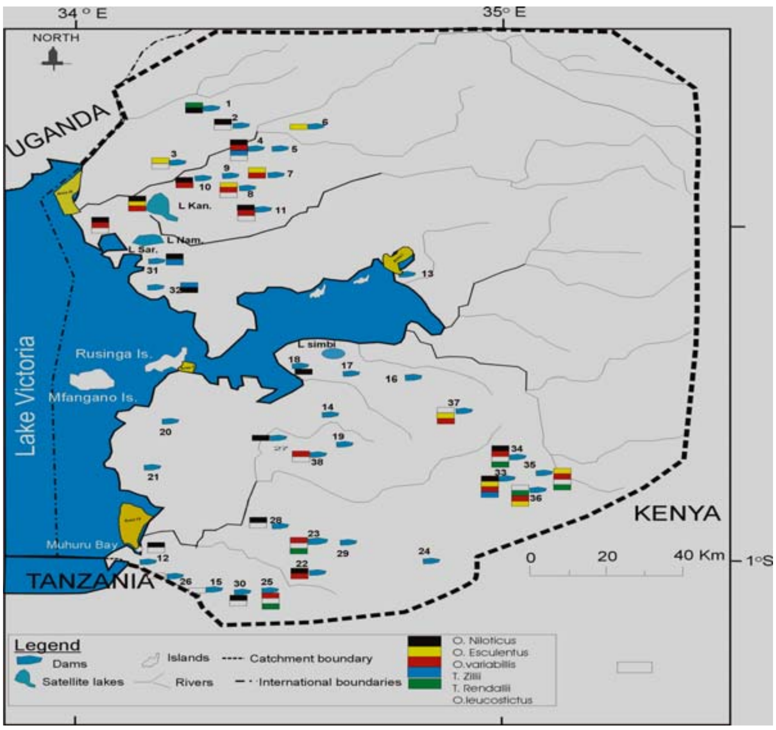

Figure 2. Spatial distribution of tilapiines in the Lake Victoria basin and catchment (Kenya).

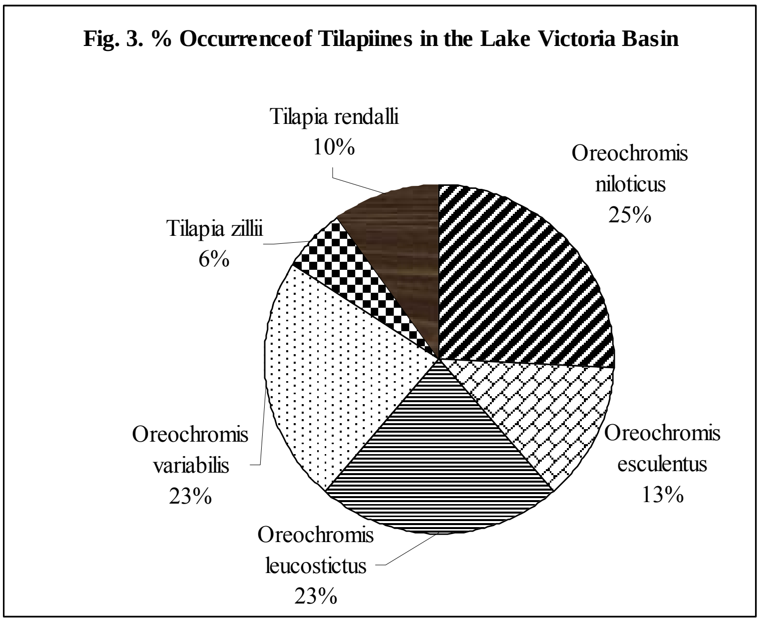

Fig. 3. % Occurrenceof Tilapiines in the Lake Victoria Basin

Species associations

The distribution of tilapiines showed most habitats accommodating more than a single species. Twenty two habitats had two or more species living sympatrically. *O. niloticus* lives sympatrically with *O. leucostictus* and *O. variabilis* in eight habitats; and with Tilapia zillii in only one habitat. Eleven populations of *O. variabilis* and *O.leucostictus* lived sympatrically while *O. leucostictus* and *O esculentus* occurred in five habitats. These associations were all significant at P<0.05 (Figure 4). The associations between O. variabilis and T. zillii, T. zillii and O. leucostictus, O. niloticus and O. esculentus were only significant at P<0.1. Other associations were not significant. Various associations included.

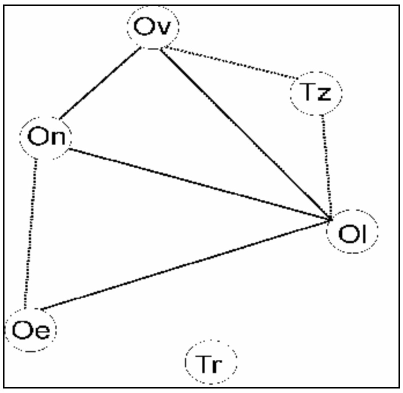

Figure 4. Species associations illustrated in the form of a three dimensional diagram. It was constructed using Cole's C_{AB} indices and the significance of association determined from values of the 2 x 2 contingency tables used to calculate C_{AB}.

Table 2. Gives a summary of the existing species combinations in Lake Victoria basin (K)

Species associations (sympatric/ parapatric)	Number of popopulations
O. niloticus / *O. leucostictus* / *T. rendalli* / *O .esculentus*	1
O. niloticus / *O. leucostictus / O. variabilis* / *T. zillii*	2
O. niloticus / *O. leucostictus / O. variabilis* / *T. rendalli*	1
O. niloticus / O. leucostictus / O. variabilis	4
O. esculentus / O. leucostictus / O. variabilis	2
O. vaiabilis / *O. leucostictus* / *O .esculentus*	2
O. niloticus / *O. leucostictus* / *O .variabilis*	2
O. niloticus / *O. variabilis/ O .esculentus*	1
O. niloticus / T. rendallii	1
O. niloticus/ O. leucostictus	4
O. esculentus / O. leucostictus	1
O. esculentus/ O. variabils	1
O. niloticus/ O. variabilis	2
O. variabilis /O. leucostictus	1
O. leucostictus	2
O. niloticus	2
O. esculentus	1

Discussion

Spatial distribution of tilapiine species in L.Victoria

It is evident that tilapiine species are well represented outside of Lake Victoria. The occurrence of *Tilapia rendalli* in close proximity to L. Victoria is only reflected amongst the tilapiine species. While distant established populations *T. rendalli* would mean determined efforts by man to benefit from the species, those close to L. Victoria may be accidental. Purposeful introductions were reflected in distribution of *O. leucostictus* and *O. niloticus* species where distant populations were to be found. However, since populations close to rivers or linked by perennial or seasonal streams were most widely distributed, such contributions cannot be ignored. This supposition conforms to the fact that indeed tilapiine species since their introduction have not remained where they are but have been redistributed through natural phenomena or man. Distribution can however be restricted within a geographical range as was the case of populations in Oki-37, Mamboleo-25, Gesebei-35, Kitaru-33 and Kijauri-36; and also Ugege-2, Kalenjuok-4, Ufinya-5, Ulanda-7, Futro-11, Mwer-10 and Lake Kanyaboli. Most of the dams were far detached from others and did not contain any tilapiine species; or had allopatric populations (Ochot-32, Achuna-18, Nyodegro-27, Olasi-26, Komondi-15, Uranga-6 / 16, Oyombe-17, Achuna-18, Kobondo-20, Kwabwai-21, Achuna Rangwe-24, Maranda-31). Therefore the occurrence of concentrations of multiple species in some areas reflected anthropogenic efforts in the dispersal of tilapiine species. Earlier, annual reports of the Kenyan Game and Fisheries Department outline the progress of stocking in various water bodies in the Lake Victoria catchment and Basin (Kenya). These records attest to the fact that most of the lakes and dams, which have been stocked with fish, had no fish until these species were introduced into Lake Victoria.

Although this study found tilapiines in the Lake Victoria basin and catchment existing in small, more or less isolated, sub-populations or races, unit stocks in Lake Victoria have adopted an aggregated type of spacing. Aggregations may have been due to influences of habitat characteristics such as abundance of food, depth, substratum type and vegetation. Distribution of *T. rendalli* in Lake Victoria was restricted to specific areas. *O. leucostictus* was found in the littoral areas, especially river mouths, while *O. niloticus* in the main lake was found to have a wide distribution, existing in all sampled habitats. Fryer and Iles (1972) attributed the dominance of Nile tilapia to its omnivorous food habits, extended spawning periods and fast growth rates. Getabu (1992) estimated growth of Nile tilapia in Lake Victoria at K=0.65 and Njiru (2003) at K=0.43 in the same environment indicating its resilience and capacity to flourish. While other tilapiines are unable to digest blue-greens, Moriarty and Moriarty (1973) demonstrated the capacity by *O. niloticus* to digest blue-greens albeit at a low pH of 1.4 in it's stomach. Njiru (2003) has attributed the low densities of *O. niloticus* in the open waters to low algal densities.This may explain the predominance of blue-greens in the dams that may favour the dominance of *O. niloticus* while being restrictive to other tilapiine species. The occurrence of tilapiines may also be instigated by breeding behaviour.

Tilapiines differ in their ability to establish themselves in the small water bodies as well as in feeding and breeding habits. This data indicated the distribution of *O. niloticus* and *O. leucostictus* in the lake basin to be widespread. Since both species were introduced, initially into Lake Victoria, their existence in other isolated water bodies infers transfers, introduction and possible re-introductions. *O. niloticus* populations living in Lake Victoria are however not separated by any geographical barrier. However, distance and differences in microhabitats characteristics may play a role in limiting migration. *O. niloticus* inhabited all the varied habitats of L.Victoria. It was found in exposed shores, muddy bays and lagoons to the more open sandy shores. Juvenile *O. niloticus* were found in shallow sheltered nurseries that were usually well oxygenated. Such niches are the ideal breeding sites for *O.niloticus* and are usually 40 to 300 cm of water in L.Victoria.

Lowe (1957) noted that *O. leucostictus* is generally a lagoon fish. It's preferred habitat was in the lagoons and near papyrus fringes in the shallow muddy bays and inlets of the lake. Current niches it occupies in the dams are much similar to this description i.e. muddy bottomed, shallow, and deoxygenated swamp. Populations of *O. leucostictus* also exist in rivers Sio, Awach and Sondu-Miriu especially in areas of lateral flooding (LVEMP reports).

The satellite lakes are connected by streams and / or rivers, therefore reducing possibility of isolation mechanisms and increasing possibilities of commonness of species. Dams were similarly linked through overflow streams that serve to transfer both nutrient loads and biological elements. With the advent of the water hyacinth over the last decade comes a resurgence of species, which were until then known to be under great extinction pressure. These changes in species richness have been attributed to reduced fishing pressure, safe habitat and less predation from Nile perch (*L. niloticus*).

Local riparian communities have for long used traditional fishing methods in obtaining fish from all water bodies. Most of these methods have been abandoned in favour of advanced techniques in fishing resulting in over-fishing and sometimes destructive fishing. However, through sensitization campaigns and legislation efforts, the fisheries of this region seem to be under less pressure and are recovering. One such example was noted for Lake Kanyaboli where community based enforcement and conservation units are in place. Here, species such as *O. esculentus*, *Lipochromis maxillaris* and *phytophagus* which are not found anywhere else find a safe haven. In Lake Nyamboyo, conservation is effected through strong traditional beliefs whereby the lake and its immediate surroundings are used for rituals, thus limiting establishment of commercial fisheries. The last effort is by the Government and other quasi government agencies to establish sanctuaries and other form of protected areas for endangered species.

Tilapiine associations

Habitats occupied by *O. niloticus* and *O. leucostictus* populations; show various types of associations that include *O. esculentus* and *O. variabilis* in the L.Victoria basin. Allopatric populations are only *O. niloticus*, probably due to over-dominance. Sympatric populations include the following combinations; *O. niloticus* / *O. variabilis, O. niloticus / O. esculentus, O. niloticus / O. leucostictus*, and *O. niloticus / O. esculentus*. A possible consequence of such associations is the occurrence of hybrids and subsequent changes in the genetic structure of both or either of the species. Such changes are achieved through natural selection pressures. The observed changes are reflected as genetic changes in the populations.

Ecological preferences by the tilapiine species

Each tlapiine has a habitat preference to maximize efficiency in feeding, reproduction and growth. Accordingly, the distribution of tilapiines is subject to prevailing ecological conditions and may result in endemism or migration.

O. niloticus is primarily a phytoplankton feeder and is especially dominant in areas of dense algal stocks such as Nyanza Gulf. It also feeds by grazing from muddy surfaces, rocks and macrophytes. Phytoplankton densities in the main lake are insufficient and cannot support *O. niloticus* directly, causing it to move long distances to feed. Breeding *O. niloticus* have been found over all types of surface bottom, but principally over sand in shallow water between 10 and 30 feet deep.

O. esculentus in Lake Victoria occurs in sheltered bays, where the bottom consists of algal mud (Lowe, 1955). Though a phytoplanktivore, it occasionally feeds on zooplankton and insect larvae. According to Greenwood (1953), diatoms are the most important food elements of the phytoplankton, particularly the filamentous *Melosira*. Consequently, *O. esculentus* prefers habitats with living layers of diatoms in bottom deposits. Both *O. niloticus* and *O. esculentus* are reported to have a black and red male breeding dress, unlike that of any other known tilapia. The breeding dress is almost identical in the two species. This supports the view that these geographical replacement species are closely related as well as ecological counterparts in their respective

T. zillii had the most clearly defined habitat preferences. It prefers the rocky outcrops and rocky shores where it scrapes on epilithic algae. It also ingested gravel and sand particles, bearing algal or bacterial floras. Small populations were recorded in vicinity of extensive *Potamogeton* beds where epiphytic growths provided a suitable food source.

Conclusion

Most of the species live in sympatric associations. O. niloticus and *O. leucostictus* rae the most dominant of all the tilapiine species and seem to tolerate most of the habitats where environmental conditions may be unfavourable for the other tilapiine species. Feeding habits may have played a major role in the survival of these two species which have a wide food spectrum. The dispersal of *O. niloticus* has been infuenced by anthropogenic activities and overflows where such connections exist while that of *O. leucostictus* may be through the efferent outflows since it is not a targeted commercial fish. O. variabilis is alsowidely distributed may be due to its increased resilience to harsh environmental conditions. The low distribution of the other species can be attributed to their inability to tolerate harsh environmental conditions such as turbidity, pH, low dissolved oxygen and high total alkalinity.

The partitioning of resources seems to lend credence to the species associations. This is reflected in the partitioning of the various resources in each habitat by accommodating omnivores, phytoplanktivores and herbivores. The associations between *O. niloticus, O. variabilis, T. zillii* and *O. leucostictus, O. niloticus* / *O. leucostictus* / *T. rendalli* / *O .esculentus*, *O. niloticus* / *O. leucostictus / O. variabilis* / *T. zillii*, *O. niloticus* / *O. leucostictus / O. variabilis* / *T. rendalli* lends credence to this hypothesis.

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References

Fryer, G. 1961. Observations on the biology of the Cichlid fish *Tilapia variabilis* Boulenger in the northern waters of Lake Victoria (East Africa). *Rev. Zool. Bot. Afr*., LXIV, 1-2.

Fryer, G. and Iles, T.D. 1972. *The Cichlid fishes of the Great lakes of Africa.* Oliver and Boyd, Edinburgh. Pp. 641.

Getabu, A. 1992. Growth parameters and total mortality in *Oreochromis niloticus* (L) from Nyanza Gulf, Lake Victoria. *Hydrobiologia* (Netherlands), Vol. 232 (1) p. 91-97.

Greenwood, P.H. 1966. The Fishes of Uganda. The Uganda Society, Kampala. 1966. 132pp.

Jembe, T. 1998. Report of the Fish Biology and Biodiversity Conservation Sub-Component of LVEMP. 98pp.

Lowe, R. H., 1955. The fecundity of tilapia species. *East Africa Agriculture Journal* 11: 45-52.

Lowe, R. H., 1957. Observations on the dagnosis and biology of *Tilapia leucosticta* Trewavas in East Africa. (Pisces:Cichlidae). *Rev.Zool. Bot. Afr.*, 55: 353-373.
Lowe, R.H., 1958. Observations of the biology of *Tilapia nilotica* LINNE in East African waters (Pisces: Cichlidae). *Rev. Zool. Bot. Afr*., 57: 129-170.
Lungayia, H.B., 1991. Some aspects of the reproductive biology of the Nile tilapia Oreochromis niloticus (L) in the Nyanza Gulf, Lake Victoria. In. Recent Trends of Research on Lake Victoria Fisheries. Proceedings of the Second EEC / KMFRI Regional Seminar on Lake Victoria Fisheries, 25th -27th September, 1991, Kisumu, Kenya.
LVEMP Annual Report, 2000. Yearly report submitted to LVEMP National Secretariat, Kisumu. 104pp.
Moriarty, C.M. and Moriarty, D.J.W. 1973. Quatitative estimation of the daily ingestion of phytoplankton by Tilapia nilotica and Haplochromis nigrioinnis in Lake George, Uganda. *J. Zool.* 171: 15-24.
Moriarty, D. J. W., 1973. The physiology of blue-green algae in the cichlid fish Tilapia nilotica. *J. Zool.* 171: 25-40.
Njiru M., 2003. *Ecology and population characteristics of Nile Tilapia, Oreochromis niloticus (L.) in Lake Victoria, Kenya*. PhD Thesis, Moi University, Eldoret, Kenya. 199pp.
Witte, F. and Densen, W.L.T. Fish stocks and fisheries of Lake Victoria: A handbook for field observations. Samara publishing limited, Samara House, Cardigan, Dyfed SA43 2JG, Great Britain, pp.399.

Zoogeographical distribution and population structure of *Taeniolethrinops praeorbitalis* (Regan 1922) exploited by artisanal fishermen in the inshores of Lake Malawi

Wisdom Changadeya, Aggrey Ambali & Lawrence Malekano

Molecular Biology and Ecology Research Unit, Ambaka Building, Chancellor College, P.O. Box 403, Zomba, Malawi, Fax: 265 1 525695/524545

E-mail-wchanga@chanco.unima.mw

Abstract

Lethrinops species flock is among the major commercially important fish species exploited in Lake Malawi. It contributes a large portion of inshore and deep-water catches yet little is known about their population genetics, which could contribute information for making management decisions for sustainable utilization and conservation. This study was carried out to determine the zoogeographical distribution, genetic diversity and population structure of Taeniolethrinops praeorbitalis populations in traditional fisheries of Southeast and Southwest arms of the lake, Mangochi District and Nkhota-kota lakeshore area respectively. A total of 10 populations of 40 individuals each were analyzed at 6 polymorphic microsatellite DNA loci. The populations were not in Hardy-Weinberg equilibrium possibly due to, sampling error caused by Wahlund effect. This is supported by inter-deme migration of more than seven individuals per generation, as determined by a multilocus estimate of number of migrants using private alleles (Slatkin's Method). However the high rate of migration has not reduced population differentiation substantially in T. praeorbitalis, mean FST of 0.1517. Allelic diversity in Mangochi populations is not significantly different from that of the Nkhota-kota populations ($p \leq 0.05$) despite higher fishing pressure in Mangochi. The genetic relationships among the populations appear to be less influenced by geographical distance. The populations seem to be evolving towards homogeneity given the high rate of migration but presently conservation and management measures should involve managing these populations as distinct populations.

Key words: Zoogeographical distribution, Microsatellite DNA loci, Genetic diversity.

Introduction

Fishing is an important economic activity in Malawi where fish provides about 70% of animal protein in the diet. Anon (1998) and Palsson *et al.,* (1999) reported that a total of 37,000 people operate traditional fishing crafts and approximately 200,000 people are directly employed in the traditional fisheries sector being the main economic activity supporting the shoreline population which had risen by about 211% in 1998 from the 1977 census (National Statistics Office, 1998). The lake steadily produces about 50-60,000 tonnes of fish per year (Fisheries Department, unpublished data) although recently fish production has declined to about 45 000 tons per annum due to overexploitation and other environmental perturbations that directly affect recruitment and mortality.

Lake Malawi harbours 500-1,000 endemic species of cichlids, all presumably derived by adaptive radiation from a single founding population within the past two million years (Greenwood, 1990; Konings, 1990; Klein *et al*., 1993; Turner, 1995). These cichlid fishes comprise the largest and most diverse known assemblage of vertebrates (Ribbink, 1999). They provide a spectacular model system with which to investigate zoogeographical distribution and population structure related questions. In this study, such questions were investigated on *Taeniolethrinops praeorbitalis,* one of *Lethrinops* species*,* found in the lake.

In many parts of the world, artisanal fisheries are of great socio-economic importance. They play a vital role as a major source of animal protein and employment (Emmerson, 1980; Pizzali, 1988). In Malawi, artisanal fishermen normally concentrate their fishing activities in the inshore waters due to lack appropriate gear for offshore fishing. Common gears include dugout canoes, plank boats without engines, gillnets (*matchela*), hooks, traps, open-water seine nets (*chirimila*), seine nets and mosquito nets. Although the inshore region has greatest species diversity, it is the most difficult to manage because it is essentially an open-access fishery, that is, the lakeshore people have virtually unlimited access to the fishery and no quotas imposed (Ribbink, 1999). Consequently they are known to exert the greatest fishing pressure, accounting for 85-90% of the total annual catch (Ribbink, 1999). Banda and Tomasson (1996; 1997) attributed most of the catches in all shallow areas of the lake to traditional artisanal fisheries. An annual landing of about 33,000 tonnes of fish is realised from inshore shallow areas of Lake Malawi (Kachinjika, 1997), thus making traditional fisheries generally important along the coast of the lake.

Lethrinops spp, locally known as *chisawasawa*, are among many commercially important species exploited in the inshores of the lake by artisanal fishermen. These fish are distinguished from other cichlids by their particular feeding habit, sifting of sand for edibles (Sprenait, 1995). *Lethrinops* is a large genus consisting exclusively of sand dwelling fish (Lewis *et al*., 1986). Species of this genus are among the most successful cichlids of the lake. They occupy large stretches of sandy and muddy bottoms in the lake, which is about 95% of the total available living space for bottom dwellers (Konings, 1995). Initially the genus encompassed all the fishes that are now under genera *Taeniolethrinops*, *Tramitichromis* and *Lethrinops* (Turner, 1996). These genera are distinguished from other Haplochromine cichlids by their dentition. In the three genera, the outer row of teeth on the lower jaw curves round to the end just posterior to the inner rows, while in other haplochromine genera it extends posteriorly as a long single row (Turner, 1996). *Lethrinops* with a solid melanin diagonal line on their body were later placed into their own genus *Taeniolethrinops* (Konings, 1995) while those with a pharyngeal bone directed downward were placed in the genus *Tramitichromis* (Ngatunga and Snoeks, 1999). The rest were retained in the genus *Lethrinops* characterized as deep bodied, vertically barred and with small ventrally placed mouths (Turner, 1996). This study was carried out on *T. preaorbitalis.*

Different depth distribution ranges for *T. praeorbitalis* have been reported. Turner (1996) reported that *T. praeorbitalis* was abundant 40m off Monkey Bay, 9-37m off Nkhota-kota, 55m off Mbenji Island and 18-25m in Southeast Arm. Banda and Tomasson (1996) in their 1994 Domira-Nkhata-bay trawl survey encountered *T. praeorbitalis* in the depth ranges of 4-25m and 36-46m and in 1995, when they bottom trawled Domira bay area, they found the species in the depth ranges of 9-15m and 22-25m. Its abundance was high in Monkey Bay, Nkhotakota, Mbenji Island, South East Arm of Lake Malawi, north west coast (Chilumba), east coast (Makanjila/ Fort Maguire), Cape Maclear and Upper Shire but it is not common in Lake Malombe (Figure 1). Little is known of population genetics of the species in Lake Malawi. The objective of this study was to determine zoographical distribution, the genetic diversity and population structure of *T. praeorbitalis* in Mangochi and Nkhota-kota districts based on artisanal fishermen catches.

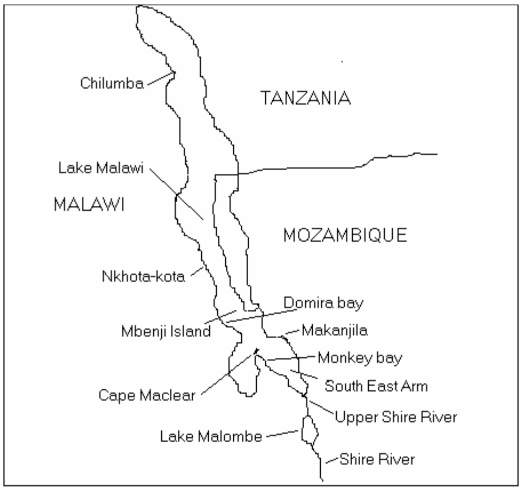

Figure 1. Distribution of *T. praeorbitalis* in Lake Malawi (Spreinat, 1995 and Turner, 1996).

Materials and methods

Fish samples were collected from various sites of Lake Malawi from November 1999 to April 2000. Major areas sampled are listed in Table 1 and Figure 2. Tissue of about 5-10 mm^2 was extracted from each fish and preserved in 95% ethanol. The samples were brought to the Molecular Biology and Ecology Research Unit DNA Laboratory for analysis.

Table 1. Fishing sites and sample size (n) along Lake Malawi

Site	Identity number	N	Location
Mangochi District			
Bindula	1	50	Western side of Eastern arm of Lake Malawi
Malembo	2	47	Eastern side of Western arm of Lake Malawi
Nkope	3	50	Western side of Eastern arm of Lake Malawi
Liganga	4	50	Western side of Eastern arm of Lake Malawi
Bakili	5	47	Eastern side of Western arm of Lake Malawi
Namiasi	6	50	Western side of Eastern arm of Lake Malawi
Nkhota-kota District			
Liwaladzi	7	50	Central area of Lake Malawi
Chia	8	33	Central area of Lake Malawi
Bana	9	44	Central area of Lake Malawi
Sungu Spit	10	36	Central area of Lake Malawi

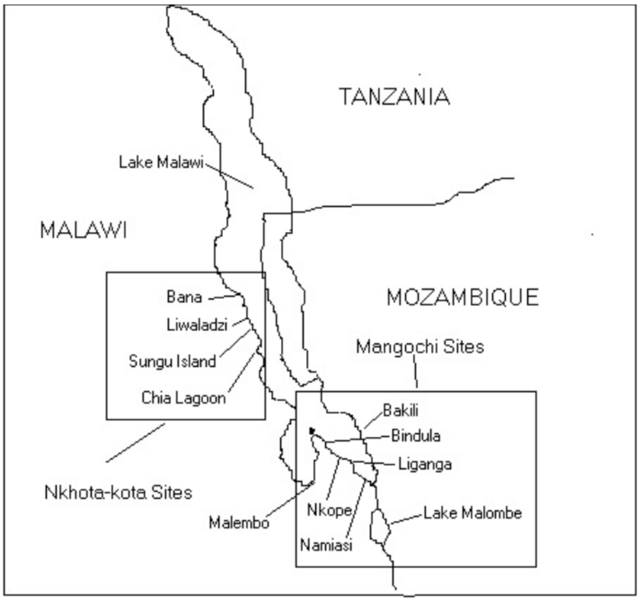

Figure 2: Map of Lake Malawi showing fish landing sites sampled for *T. praeorbitalis*

DNA extraction

DNA was extracted using a protocol as described in Cook *et al.* (1996) with a few modifications as outlined in the following procedure; Muscle tissue of 3mm^2 size was macerated and placed in a 2.0ml microtube and washed in 1.0ml high TE (100mM Tris-Cl, 40mM EDTA). The mixture was vortexed for 30s and left to stand on the bench for at least 10m before the liquid was aspirated off. 250µl Marine Gene Probe Laboratory (MPGL) buffer (10mM Tris-Cl pH=8.3, 1mM EDTA, 200mM LiCl, 0.8% SDS) and 2.5µl proteinase K were added to the microtube. MPGL buffer lyses cellwalls while proteinase K catalyzes the process. The sample mixture was then incubated at 50°C in an automated Advantec water bath for at least three hours with intermittent mixing until the tissue was completely digested. This was followed by vortexing the mixture before spinning in a Tomy High Speed Refrigerated Micro Centrifuge at 150 000 rpm for 5m. The aqueous phase was transferred into new 2.0ml microtubes. 500µl TE (10mM Tris-Cl, 1mM EDTA) was then added and the mixture vortexed. A further addition of 35µl of 3M NaCl and 750µl cold isopropanol followed. The mixture was vortexed and incubated at -40°C for at least an hour to precipitate DNA. The mixture was spun in a microfuge for 10 m and the supernatant was discarded. The DNA pellets were washed with 500µl of 70-95% cold ethanol followed by 5 m of centrifuging after mixing. The ethanol was then decanted and the DNA pellets air-dried for 10-15 m before dissolution in 100µl TE. The DNA was diluted to a concentration of 25ng/µl for pcr.

Six microsatellites loci, were scored (Table 2). OS 64 primer was designed for *Oreochromis shiranus* and the UNH primers were designed for *Oreochromis niloticus*; therefore they were cross-primed with *T. praeorbitalis* DNA.

Table 2: Primer sequences and their annealing temperatures (T_{ann})

Locus			T_{ann} (ºC)
UNH 130:	Forward:	5´-AGGAAGAATAGCATGTAGCAAGTA-3´	54
	Reverse:	5´GTGTGATAAATAAAGAGGCAGAAA-3´	
UNH 132:	Forward:	5´-ATATAAGAAACTGAGTCGGTGAG-3´	54
	Reverse:	5´-TGGAAATAGAGGGTGGGTGAG-3´	
UNH 154:	Forward:	5´-ACGGAAACAGAAGTTACTT-3´	54
	Reverse:	5´-TTCCTACTTGTCCACCT-3´	
UNH 146:	Forward:	5´-CCACTCTGCCTGCCCTCTTAT-3´	54
	Reverse:	5´-AGCTGCGTCAAACTCTCAAAAG-3´	
UNH 201:	Forward:	5´-CTGCTAGACTGCGAAAC-3´	54
	Reverse:	5´-ACAGTGCAACACCAGAC-3´	
OS 64:	Forward:	5´-CAGTGTCTTCAGTTCCTTGC-3´	54
	Reverse:	5´-CAGAAGCATCTTATTGATGAC-3´	

Polymerase Chain Reaction (PCR)

Polymerase Chain Reaction (PCR) cocktail included the following: 5.7µl PCR grade water, 1µl of 10mM DNTP mix, 1.3µl of 10 x PCR buffer, 1µl of 25mM $MgCl_2$, 15pmol of both forward and reverse primers, 0.06µl of 5U AmpliTaq-Gold and 2µl of 25ng template DNA.

PCR was performed in a Perkin Elmer GeneAmp PCR system 9600-thermocycler version 2.01. The PCR conditions were as follows: Amplitaq-Gold activation step at 95ºC for 12m followed by 10 cycles of amplification consisting of denaturing step at 94ºC for 30s, annealing step at 54ºC for 15s and extension step at 72ºC for 30s. This was followed by 25 cycles of amplification consisting of denaturing step at 89ºC for 30s, annealing step at 54ºC for 15s and extension step at 72ºC for 30s. Amplification was completed with a final extension at 65ºC for 20m.

Electrophoresis of PCR products on ABI Model 310 Prism Genetic Analyzer

PCR products were analyzed on a capillary-based ABI 310 Prism Genetic Analyzer. 12µl of deionized formamide combined with 0.5µl GeneScan Rox-350 Size Standard and 3.0µl PCR product was placed into a Genetic Analyzer 0.2ml sample tube. The sample tubes were closed with Genetic Analyzer septa strips. After gentle mixing, the samples were denatured in Perkin Elmer GeneAmp PCR System 9600 thermocycler at 95°C for 3m and briefly chilled on ice. The samples were then loaded on the ABI 310 Genetic Analyzer and the run started in accordance with the supplier's protocol (PE Applied Biosystems ABI PRISM 310 Genetic Analyzer User's Manual, 1995). The Size Standard GS Rox-350 was used to estimate the size of PCR products in the GeneScan runs. GeneScan Analysis Software (ABI Prism™ Genescan Analysis 2.1 User's Manual, 1996) provides highly repeatable estimates of fragment size.

Data analysis

GENEPOP (version 3.3; Raymond and Rousset. 1995) was used to conduct the following analyses: test for conformity to Hardy-Weinberg Equilibrium (Haldane.1954; Weir, 1990; Guo and Thompson, 1992), test for genotypic linkage equilibrium, test for genic differentiation, generation of Rho-ST (Rousset, 1996) Matrix and estimation of effective number of migrants (Slatkin's 1985 private allele method).

POPGENE Version 1.31 software (Yeh, *et al.,* 1999) was used to compute a number of measures of genetic variation within and between sample populations. The following variables were computed to determine allelic diversity: observed number of alleles, effective number of alleles (Kimura and Crow, 1964), observed heterozygosity, expected heterozygosity (Nei, 1973) and Shannon's information index (Lewontin, 1974). Significance of various analyses was determined by the formula (Mean ± 2SE) confidence interval. The among-population component of genetic variance F_{ST} was calculated to measure the proportion of total variation that could be ascribed to differences between population allele frequencies. F_{ST} values were computed between population pairs. F_{IS} values were also calculated to determine heterozygote deficiency and excess within populations.

Mantel's test was carried out to determine the correlation between geographical distance and Rho-ST (Rousset, 1996) values between populations. The test was based on the null hypothesis that there is no correlation between genetic distance and geographical distance between locations where the population samples were collected. The MXCOMP program of NTSYS-pc was used to compute a product-moment correlation coefficient (i.e. normalized Mantel's statistic Z) for the two distance matrices (Rohlf, 1992). To determine if the correlations were significant, actual coefficient was compared to the values produced by randomly permuting the matrix pair 1000 times.

Genetic relationship among the populations was analysed by a multidimensional scaling (MDS) of the Nei's (1978) Unbiased Genetic Distances. MDS analysis depicts a complex set of relationships among sample populations represented by distance or similarity matrix, in space of a few dimensions without any significant loss of resolution. MDS does not assume linearity and therefore is appropriate analysis for most natural populations where a certain degree of geneflow occurs (Kamonrat, 1996).

Table 3 Total number of alleles (A) and allele size range (SR) in base pair in *T. praeorbitalis* populations at six loci

Population	UNH201		OS 64		UNH146		UNH132		UNH130		UNH154	
	A	SR	A	SR	A	SR	A	SR	A	SR	A	SR
Bindula	11	150-266	3	126-136	4	109-119	5	105-219	22	181-235	13	99-225
Malembo	12	144-294	5	124-140	2	119-123	7	105-125	20	177-235	12	99-225
Nkope	17	150-280	2	124-126	2	119-123	4	111-119	17	173-235	11	99-143
Liganga	15	144-294	2	126-132	3	119-127	8	105-219	22	173-235	12	115-225
Bakili	13	154-294	2	124-126	9	107-127	5	111-125	13	177-235	14	99-225
Namiasi	12	156-280	2	124-126	5	107-123	7	105-219	21	173-235	6	147-225
Liwaladzi	15	154-280	2	124-126	3	119-127	1	111	19	181-235	11	99-135
Chia	9	156-280	2	124-126	3	119-127	6	111-125	21	173-235	9	99-145
Bana	15	144-266	2	126-132	8	107-127	6	109-125	13	173-235	13	99-225
Sungu Spit	7	144-256	2	124-126	6	109-121	4	111-119	18	173-235	14	99-225

Results

Allelic variation

Data on number of alleles scored per population and allele size range are presented in Table 3. Loci UNH 201, UNH 130 and UNH 154 showed high variability compared to the other three. A general trend observed was that the loci with low variability had alleles of smaller size than those of high variability.

Genetic variability

Data on genetic diversity are presented in Table 4 and Figure 3 and 4. Genetic diversity varied considerably among the populations. There was significant difference in mean number of alleles between Malembo and Nkope populations (p<0.05; Table 4). All the other populations were not significantly different (p>0.05). Although Malembo had significantly lower number of alleles than Nkope, no significant difference was observed between the two populations in terms of mean effective number of alleles. Mean heterozygosity in the populations was more than 0.5. Liganga and Chia populations have significantly higher heterozygosity than Sungu Island population. In overall no significant difference in allelic diversity and heterozygosity was observed between the Mangochi District and Nkhota-kota District populations(Figure 3). Mean observed number of alleles at six loci were higher than mean effective number of alleles since the latter takes into account the relative frequencies of alleles, to which rare alleles contribute negligibly to the estimates (Kamonrat, 1996).

Table 4 Mean genetic variability for *T. preaorbitalis* populations at six loci.

Population	na*± S.E	ne*± S.E	Obs Het** ± S.E	Exp Het**± S.E	I*± S.E
Bindula	7.67± 0.85	3.90± 0.50	0.68±0.05	0.64± 0.04	1.39± 0.12
Malembo	6.67± 0.87	3.84± 0.41	0.63±0.05	0.62± 0.04	1.28± 0.12
Nkope	10.17±0.75	5.65± 0.45	0.68±0.03	0.71± 0.03	1.68± 0.09
Liganga	8.67± 0.77	5.66± 0.55	0.79±0.03	0.69± 0.02	1.54± 0.10
Bakili	10.17±0.85	5.87± 0.51	0.63±0.03	0.74± 0.02	1.71± 0.09
Namiasi	9.67± 0.59	4.85± 0.33	0.68±0.03	0.70± 0.03	1.62± 0.09
Liwaladzi	8.83± 0.76	5.41± 0.39	0.64±0.02	0.75± 0.02	1.65± 0.08
Chia	8.50± 0.85	6.08± 0.60	0.73±0.04	0.63± 0.04	1.48± 0.13
Bana	8.50± 0.82	3.34± 0.27	0.67±0.03	0.62± 0.02	1.33± 0.09
Sungu Is	9.83± 0.60	4.83± 0.33	0.57±0.03	0.70± 0.02	1.63± 0.08

** = Expected heterozygosity and observed heterozygsity na*= Observed number of alleles ne*= Effective number of alleles I*= Shannon's Information index (Lewontin (1972)

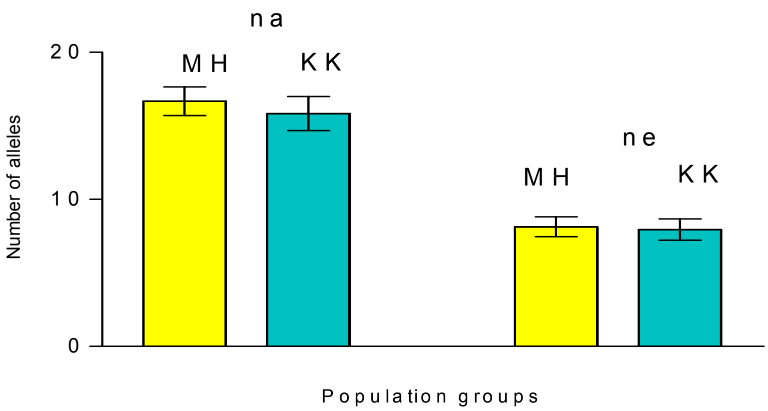

Figure 3 Mean ± SE observed number of alleles (na) and effective number of alleles (ne) at all six loci for pooled Mangochi (MH) and Nkhota-kota (KK) *Taeniolethrinops praeorbitalis* populations.

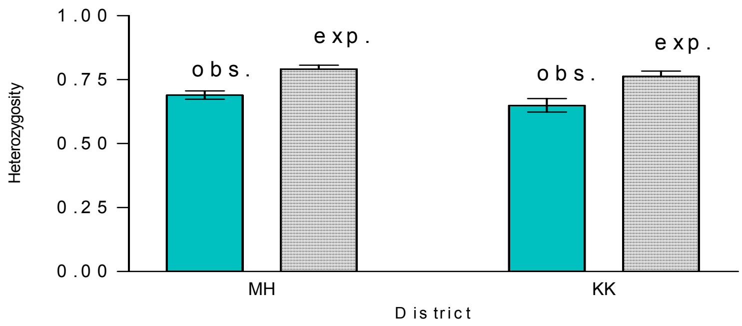

Figure 4 Mean ± SE observed and expected heterozygosity at all six loci for pooled Mangochi (MH) and Nkhota-kota (KK) *Taeniolethrinops praeorbitalis* populations.

Table 5 Summary of F-Statistics for all loci for *Taeniolethrinops praeorbitalis* populations

Locus	F_{ST}	F_{IS}
UNH201	0.0929	0.2809
UNH130	0.0536	0.2460
UNH154	0.1317	0.1234
UNH132	0.3547	0.2049
UNH146	0.1537	-0.3524
OS 64	0.1975	-0.8901
Mean	0.1517	0.0022

Population structure

Table 5 presents intra- and interpopulation variation at the six loci. F_{IS} values indicated that there was heterozygosity deficiency at all loci except at OS 64 and UNH146 and mean values showed a considerably low heterozygosity deficiency. All F_{ST} values indicated a considerable amount of differentiation with locus UNH132 exhibiting the highest F_{ST} value of 35.5%. The number of migrants per generation was 7.57 based on adjusted mean samples size of 39 (data not shown).

The relationship between genetic distance and geographical distance

Nei's genetic distance values (data not shown) and multidimensional scaling of the populations based on genetic distance is presented in Figure 5. No obvious pattern of clustering could be identified. This is supported by weak relationship between genetic distance and geographical distance (Table 6).

Table 6. Mantel's Statistics.

Normalised Mantel's statistic Z	Z= 0.06
Mantel's t-test	t= 0.3742
Probability (random Z < observed Z)	*P*= 0.6459

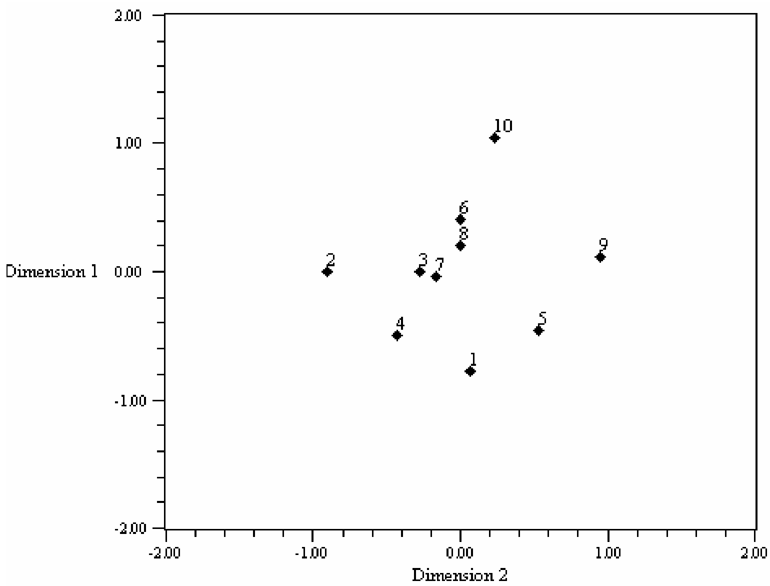

Figure 5 Genetic relationships between populations of *Taeniolethrinops praeorbitalis*. The plot is against dimension 1and 2 of the configuration produced by multidimensional scaling (MDS) analysis of Nei's Genetic distances. 1-6 (1Bindula, 2 Malembo 3 Nkope, 4 Liganga 5 Bakili, 6 Namiasi) Mangochi populations and 7-10 (7 Liwaladzi, 8 Chia 9 Bana 10 Sungu Spit) Nkhota-kota populations. Location refer to Figure 2.

Discussion

Zoogeographical distribution of *T. praeorbitalis*

Distribution of *T. praeorbitalis* populations seem to cover the whole lake (Spreinat, 1995; Konings, 1990 as cited by Turner, 1996) though some areas between Nkhota-kota and Chirumba show no existence of these populations. These populations are more widely distributed in the southern and central parts of the lake, areas that are shallower than the northern part, and their inshore lake bottom is predominantly sandy.

Conformity to Hardy-Weinberg Equilibrium and test for linkage disequilibrium

At the intraspecific level detection of mixing of stocks is possible if there are different frequencies of the same allele at a locus. Two tests commonly used to determine if a fish stock is a mixture of fish from more than one population are a test of Hardy-Weinberg expectations and a test for linkage disequilibrium. Both tests work on the population genetics principle that a mixture of genepools will exhibit a Wahlund effect, i.e., heterozygote excess relative to binomial distribution (Kamonrat, 1996). Populations of *T. praeorbitalis* were not in HWE (data not shown) probably due to sampling error caused by Wahlund Effect (Hartl and Clark, 1989). This observation could be supported by 60% homozygosity excess that was observed among the populations (Data not shown). Departure from HWE by these populations can also be explained by the high rate of inter-deme migration of more than seven individuals per generation.

Genetic diversity

Among the Mangochi and Nkhota-kota populations, there is still considerable amount of genetic diversity as indicated by a good proportion of rare alleles represented by margins between observed number of alleles (na) plot and effective number of alleles (ne) plot (Figure 4). Large margin between the two estimates indicates the existence of several low frequency alleles (rare alleles) at the six loci (Figure 4) This is typical of microsatellites, in which each locus displays a large number of alleles, of which many are individually rare (Kamonrat, 1996). Shannon's information index (Table 4) also indicated that the allelic diversity is high among the populations with a range of 1.28-1.71. Although the Mangochi populations generally experience higher exploitation pressure due to intensive fishing that goes on in the areas, their allelic diversity is not significantly different from that of Nkhot-kota populations. This supports the fact that there are population admixtures for the species in the lake.

The relationship between genetic distance and geographical distance

The low relationship between genetic distance and geographical distance implies that these populations do not fit well with isolation by distance model. Under this model, rate of geneflow is the highest between close populations and it is expected that close populations should have similar genetic composition but such is not the case in these populations. This is supported by the high rate of migration that occurs among the populations in the lake. Our findings concur with earlier studies of *Lenthrinops* species flock which have reported that flock behave like a single population with high migration rate (Duponchelle, et al, 1999)

Conclusion

The *T. praeorbitalis* populations are distributed almost over the whole lake with the southern and central parts of lake having higher distribution than the northern part. They are not in Hardy-Weinberg Equilibrium possibly due to sampling error caused by Wahlund effect. This is supported by inter-deme migration of more than seven individuals per generation. The populations still harbour considerable amount of genetic diversity and there is no significant difference in genetic diversity and heterozygosity between Nkhota-kota and Mangochi populations despite differences in exploitation pressure. The populations are still heterogeneous with a differentiation of 15%. This differentiation is not enough and the populations seem to be evolving towards homogeneity given the high migration rate of eight individuals per generation. But presently, conservation and management procedures should involve conserving and managing these populations as distinct populations since they still harbour distinct gene pools The genetic relationships among the populations appear to have low dependence on geographical distance.

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References

Anon, 1998. Frame Survey of Malawian Fisheries 1998. Fisheries Research Unit, Fisheries Department, (unpubl. manuscript)

Banda M. C. & Tomasson, T., 1996. Government of Malawi, Fisheries Department. Survey of trawling grounds and demersal fish stocks in central Lake Malawi, from Domira bay to Nkhatabay in 1994 and 1995. 1-2, 6-7.

Banda M.C. & Tomasson, T., 1997. *Government of Malawi, Fisheries Department; Demersal fish stocks in southern Lake Malawi: Stock assessment and exploitation.* 21.

Cook D. I., Kamonrat W., Paquet D & Pitman E. R., 1996.A single tube rapid extraction of DNA from fish blood. *J. Fish. Biol*. (submitted).

Duponchelle F. Snoeks J., Agnese J. F., Hanssens H., Mafuca J., Mandere D. & Msukwa A. (1999) Comparison of taxonomic, genetic and life history characteristics of two species (*Mylochromis anaphyrmus* and *Lenthrinops gossei*) in various parts of Lake Malawi/Nyasa. *In:* A.J. Ribbink and A.C. Ribbink eds. *Abstracts of the SADC/GEF Lake Malawi/Nyasa Biodiversity Conservation Project Final Workshop*, 4-3/03/1999. Senga Bay, Malawi: 34-35.

Emmerson D. K., 1980. Rethinking artisanal fisheries development: Western concepts, Asian experiences. *World bank Staff working paper No 423*, Washington, D.C.

Greenwood P.H., 1990. Speciation, pp 86-102. *In.* M. Keenleyside ed. *Cichlid fish: Behaviour, Ecology and Evolution*. Chapman and Hall London.

Guo S. W. & Thompson E. A., 1992. Performing the exact test of Hardy-Weinberg proportion for multiple alleles. *Biometrics* , 48, 361-372.

Haldane J. B. S., 1954. An exact test for randomness of mating. *J Genet* 52: 631-635.

Hartl D. L. & Clark A. G. (1989) Principles of Population Genetics. Second Edition. Sinauer Associates, Inc. Publishers. Sunderland.

Kachinjika O.M., 1997. Biodiversity conservation and sustainable utilisation of pelagic cichlid fishes of Lake Malawi/Niassa. Proceedings of the first workshop of the ODA Ncheni Project. Senga Bay. Malawi. Volume1: 7 –8.

Kamonrat W. 1996 *Spartial genetic structure of Thai Silver Barb Puntius gonionotus (Bleeker) populations in Thailand*. PhD. Thesis, Dalhousie University.

Kimura M. & Crow J. K., 1964. The number of alleles that can be maintained in a finite population. *Genetics*, 49, 725-738.

Klein D., Ono H., O'hUigin C., Vincek V., Goldschmidt T & Klein, J., 1993. Extensive MHC variability in cichlid fishes of Lake Malawi. *Nature.* Volume 364. 330 –334.

Konings A., 1990. *Cichlids and all other fishes of Lake Malawi*. THF Publication, Nepture, New Jersey.

Konings A., 1995. *Malawi cichlids in their natural habitats*. Second edition. 282-283, 296-298.

Lewis D., Trendall J.& Reinthal, P., 1986. *A guide to the fishes of Lake Malawi National Park*. WWF, Gland, Switzerland. 41.

Lewontin R. C., 1972. The apportionment of human diversity. *Evolutionary Biology*, 6, 381-398.

Lewontin R. C., 1974. *The genetic basis of evolutionary change*. Columbia University Press, New York.

National Statistical Office Malawi Government, 1998. Population and Housing Census: *Report of the Preliminary Results*. 16, 29-37.

Nei M., 1973. Analysis of gene diversity in subdivided populations. *Proc. Natl. Acad. Sci.* USA 70: 3321-3323.

Nei M., 1978. Estimation of average heterozygosity and genetic distance from a small number of individuals. *Genetics* 89: 583-590.

Ngatunga, B. & Snoeks ed., J., 1999. Preliminary key to the shallow water (<20m) *Lethrinops* sensu lato, pp 2. *In.* Snoeks ed. *Report on systematics and taxonomy. SADC/GEF Lake Malawi/Nyasa Biodiversity Conservation Project*..

Palsson O.K. Bulirani A. & Banda M., 1999. A review of biology, fisheries and population dynamics of chambo (*Oreochromis spp*., CICHLIDAE) in Lakes Malawi and Malombe. Fisheries Department. *Fisheries Bulletin* No. 38. 2-3.

Pizzali A.F., 1988. Small-scale fish landing and marketing facilities. FAO Fisheries Technical Paper 291, Rome.

Raymond M & Rousset F., 1995. GENEPOP (v.1.2). A population genetics software for exact test and ecumenicism. *Journal of Heredity* 89: 248-249.

Regan C.T., 1922. The cichlid fishes of Lake Nyassa. *Proc. Zool. Soc. Lond*. (PZSL). 27 Jan. 1921. (pt 4) (no.36) 675 – 727.

Ribbink A. J., 1999. Aspects of conservation and sustainable levels of use of Lake Malawi fishes. 203-215.

Rohfl F. J., 1992. *NTSYS-pc. Numerical Taxonomy and Multivariate Analysis System.*Exeter Software, Setauket, NY.

Rousset F., 1996. Equilibrium values of measure of population subdivision for stepwise mutation processes. *Genetics* 142: 1357-1362.

Slatkin M., 1985. Rare alleles as indicators of geneflow. *Evolution.* 39(1), 53-65.

Spreinat A., 1995. Lake Malawi cichlids from Tanzania. 87-93.

Turner G. F., 1995. The Impact of Species Changes in African Lakes. 365.

Turner G. F., 1996. Offshore cichlids of lake Malawi. 76-77, 97-99.

Weir B. S., 1990. *Genetic Data Analysis*. Sinauer Associates, Sunderland, MA.

Yeh F.C., Yang R. & Boyle T., 1999. POPGENE VERSION 1.31, Microsoft Window-based Freeware for Population Genetic Analysis. Quick User Guide University of Alberta and Centre for International Forestry Research.

Biodiversity of the largest floodplain lake ecosystems in Latvia

Davis Gruberts[1], Ivars Druvietis[2], Lelde Enele[3], Jana Paidere[4], Elga Parele[5], Jenis Priedtis[6], Arkdijs Poppels[7], Artarsakute[8]

[1]: Department of Chemistry and Geography, Daugavpils University, Pardes Street 1, Daugavpils, LV 5400, Latvia. Email-davis@dau.lv; phone: +3715422302; fax: +3715422890
[2]: Institute of Biology, University of Latvia, Miera Street 3, Salaspils, LV-2169, Latvia
[3]: Nature Fund of Latvia, Kronvalda bulv. 4, Riga, LV 1842, Latvia
[4]: Department of Chemistry and Geography, Daugavpils University, Pardes Street 1, Daugavpils, LV 5400.
[5]: Institute of Biology, University of Latvia, Miera Street 3, Salaspils, LV-2169, Latvia
[6]: Faculty of Natural Sciences and Mathematics, Daugavpils University, Pardes Street 1, Daugavpils, LV 5400, Latvia
[7]: Latvian Agency of Fish Resources, DaugavGrivas Street 8, Riga, LV 1048, Latvia
[8]: Department of Biology, Daugavpils University, Vienbas Street 13, Daugavpils, LV 5400, Latvia

Abstract

Differences in biological diversity of the four largest floodplain lakes in Latvia located along the Daugava River have been explored in 2004. Lake Skuku and Lake Dvietes are considered as Internationally Important Bird Areas and potential Natura-2000 sites. Because of large and prolonged spring floods, significant impact of the Daugava River flood pulse on phytoplankton, zooplankton and macrozoobenthos communities as well as on macrophyta vegetation of these floodplain lakes was expected. The aim of this study was to test this hypothesis. Seasonal fluctuation of the water level for each lake was determined in the field by levelling the highest position of the spring flood debris to actual water level in July 2004. Samples of phytoplankton, zooplankton and macrozoobenthos were collected in central part and littoral zone of each lake. Composition of aquatic vegetation was observed in the field. Collected samples were analysed for species composition by standard methods. For community comparison, several similarity and diversity indices were calculated. Typical taxons (species) for these ecosystems were stated. Possible correlation between species diversity and flooding hydrology was also explored. In general, floodplain lakes of the Daugava River can be characterised by high species diversity and significant differences in community composition between the sampling sites. The highest diversity was found in the shallow eutrophic lake Ljubasta, dominated by rich aquatic vegetation and characterised by clear water stage. Seasonal fluctuation of water level has minor impact on community composition of these aquatic ecosystems in summer. Other local environmental factors, like water temperature, nutrient (food) availability or diversity of biotopes, were considered to be more important. An impact of annual flooding was probably found in a well-seen dominance of rare species in a composition of phytoplankton, macrozoobenthos and macrophyta communities.

Key words: biodiversity, floodplain lakes

Introduction

As stated from limnological studies by Junk, Bayley and Sparks (1989), floodplain lake ecosystems function as important refugee and reproduction sites for many riverine and wetland species. Community structure and composition in these ecosystems are strongly influenced by such external factor as annual flooding, which allows an exchange of water, chemicals, organic matter, sediments and biota between the river and lake ecosystems. According to Huszar and Reynolds (1997), periodic flooding simplifies ecosystem's structure, rejuvenates resources and stimulates productivity. However, many riverine and wetland species have disappeared from river floodplains due dramatic changes in river morphology and water quality. Because of inputs of agricultural fertilizers and sewage, many large European rivers are highly eutrophic. Frequent influxes of water from polluted rivers impoverish aquatic communities of floodplain lakes, therefore minimising species diversity. For example, in floodplain lakes along the Lower Rhine and Mouse in Netherlands, species composition and diversity of plankton communities showed clear negative correlations with the long-term average annual flood duration and frequency. According to Van den Brink, Van Katwijk and Van der Velde (1994), the presence or absence of characteristic species in the study sites was related to the availability of nutrients and substrates and the chemistry of water and sediment.

The Daugava River is one of the largest in Eastern Europe. It flows through Russia, Byelorussia and Latvia and drains to the Baltic Sea. For centuries Vikings used it as trade route from Baltic to the Black Sea. Until the middle of 19th century, it served as important waterway for merchant boats and wood floats. Today, it is used mainly for hydroelectric power generation. As stated in National Environmental Policy Plan for Latvia (1995), aquatic ecosystems and habitats have been significantly harmed by construction of the Daugava Hydroelectric Power Station Cascade, which destroyed natural biotopes in huge areas both in the Daugava River itself and also in its ancient valley. Since the second part of the 20th century, dams of three huge reservoirs (Rīgas, Ķeguma, Pļaviņu) have completely blocked migratory paths for salmon, eel and lamprey. Large numbers of baby fish perish when sucked through filters and into the turbines of power stations. Aquatic plants, benthic organisms and fish spawn are also often killed by water level fluctuations in reservoirs. Construction of the Cascade has also created other serious environmental problems, such as erosion of the shores of reservoirs, fluctuation of groundwater level in adjacent territories, formation of ice jams during the spring floods and so on.

During the next 10-12 years, construction of 4 new hydroelectric power stations on the Upper Daugava in Byelorussia is planned (Gruberts, 2003). Despite their small scale, they can adversely affect floodplain lakes along the Daugava River in Latvia. Meanwhile, biological diversity of these lakes is largely unexplored. Under such circumstances, comprehensive investigations in hydrology and ecology of these floodplain lakes are urgently needed.

The four largest floodplain lakes in Latvia are located in Daugavpils region, Southeast Latvia, on both sides of the Daugava River (see Figure 1). They are medium sized, shallow lakes of glacial origin characterised by similar morphological and hydrological parameters.

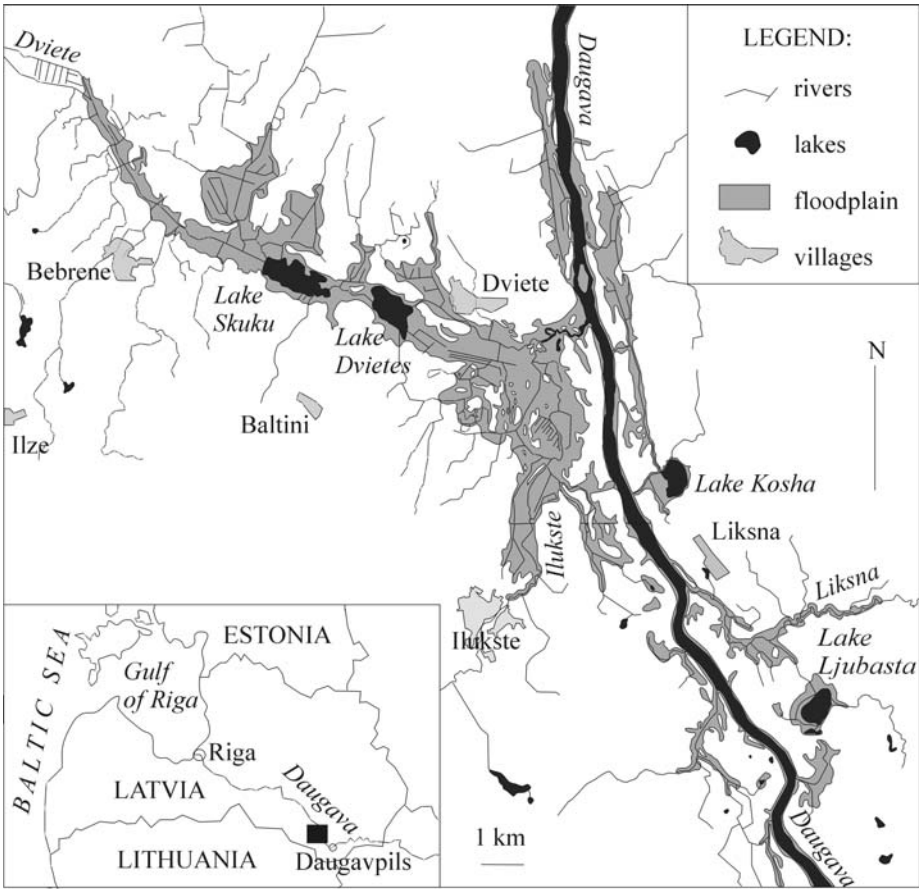

Figure 1. Location of the four largest floodplain lakes of the Daugava River in Latvia.

Seasonal water level fluctuation in these lakes is influenced by such regional factors as the amount of snow accumulated in the Daugava River drainage area during the winter, the air temperature increase rate in spring or formation of ice jams in the Daugava valley during the spring floods. According to hydrological grouping presented by Gruberts, Paidere, Priedītis *et al*. (2005), these lakes are flooding repeatedly, not only during average spring floods in March, April and May but also at the highest summer (autumn) water level in the Daugava River caused by heavy rainstorms. Almost all of them can be considered as shallow eutrophic lakes with minor or moderate anthropogenic influence, mainly by water level regulation during the 20th century.

These four lakes have, also, a high nature protection value: three of them are considered as potential Nature-2000 sites. In addition, Lakes Skuku and Dvietes are situated in the Dvietes Floodplain Nature Park considered as an Important Bird Area of European Union importance since 2000. According to Rainskis (2004), this site is used by endangered waterbirds like *Anser fabalis* and *Cygnus cygnus* as resting place during the annual migration. In summer, vast floodplain meadows of this site are significant nesting biotopes for such rare bird species as *Crex crex* and *Gallinago media* included in European Union Birds Directive. During the annual floods, these lakes serve, also, as spawning places for the Daugava River fish populations.

Annual flooding significantly influences seasonal water level fluctuation, aquatic chemistry and biotic communities of these lakes. In 1999, the flood pulse impact on phytoplankton composition and diversity in the Skuku (Grivas) Lake has been already observed by Gruberts and Druvietis (2001). Compositional and structural shift between the planktonic and epiphytic (benthic) algae groups controlled by flooding hydrology was stated during this study. Because of large and prolonged annual flooding, significant impact of the Daugava River flood pulse on zooplankton and macrozoobenthos communities and aquatic vegetation of its floodplain lakes has been also expected. The aim of this study was to test this hypothesis by exploring possible consequences of annual flooding on summer planktonic and benthic communities of these four floodplain lakes, which are not only the largest for Latvia but also for the whole Daugava River.

Materials and methods

Floodplain lakes Skuku, Dvietes, Kosha and Ljubasta were explored in July, 2004. Seasonal fluctuation of water level for each lake was determined in the field by levelling the highest position of spring flood debris to actual water level in the lake on the day of sampling. Samples of phytoplankton were collected in central part and littoral zone of the lakes from 0,5 m depth by Ruttner type water sampler and fixed in the field by Lugol solution. The inverted microscope was used for identification of phytoplankton species according to standard methods. Samples of zooplankton (volume 100 l) were collected in central part of the lakes from epilimnion by 65 μm plankton mesh and fixed in the field by formaldehyde to 4 % solution. The light microscope was used for zooplankton species identification. Macrozobenthos samples were collected in central part and littoral zone of the lakes by Ekman type grab with a 1/40 m^2 surface. A sieve with a mesh size 0,5 mm and 4% formaldehyde solution was used for zoobenthos sample concentration and fixation. Composition and distribution of aquatic vegetation was observed in the field.

To determine if the communities of these lakes can be classified together or need to be separated, two similarity measures (qualitative and quantitative) were obtained. At first, Sorensen's similarity coefficients were calculated for each systematic group based only on the species presence/absence data. In addition, calculations of Renkonen's index (percentage similarity) were also performed for each systematic group after the data standardization in terms of percentages. At least, species diversity was calculated for the samples collected in central parts of the lakes applying Shannon's index (Shannon & Weaner 1963).

Results

According to water level fluctuation measurements in July 2004, annual flooding of the Daugava River influenced all four lakes. The highest amplitude of seasonal water level fluctuation (3,6 m year $^{-1}$) was observed in the lakes Dvietes and Skuku, the lowest – in the Lake Kosha (Table 1). During this study, more than 58 taxa of phytoplankton, 32 taxa of zooplankton, 42 taxa of macrozoobenthos and 42 taxa of macrophyta were stated (Gruberts *et al*., 2005). Lake Ljubasta was characterised by the highest number of phytoplankton and zooplankton taxa, Lake Dvietes – of macrozoobenthos, Lake Skuku – of macrophyta (Table 1). In total, 18 species (taxa) were considered as common, found in each lake (Table 2).

Table 1. Annual flooding amplitude and biological diversity of the largest floodplain lakes in Latvia, July 2004.

Lake	Water level change, m year $^{-1}$	Total number of species (taxa) / Shannon's index†			
		Phytoplankton	Zooplankton	Macrozoo-benthos	Macrophyta
Skuku	3,6	17 / 1,76	9 / 2,94	21 / 2,25	31 / -
Dvietes	3,6	4 / 1,57	6 / 0,31	27 / 2,65	27 / -
Ljubasta	3,3	48 / 0,93	19 / 2,21	6 / 0,98	24 / -
Kosha	1,7	5 / 0,03	13 / 1,22	4 / 0,41	28 / -

† - Shannon's index calculated for the samples collected in central parts of the lakes.

Table 2. Common species (taxa) of the largest floodplain lakes in Latvia, July 2004.

Phytoplankton	*Cryptomonas* sp.
Zooplankton	*Chydorus sphaericus*
Macrozoobenhos	Chironomidae, Oligochaeta
Macrophyta	*Alisma plantago-aquatica*, *Ceratophyllum demersum, Chlorophyta, Equisetum fluviatile, Glyceria maxima, Hydrocharis morsus-ranae, Lemna minor, Lemna trisulca,* Potamogeton sp., *Sagittaria sagittifolia, Scirpus lacustris, Sparganium erecta, Spirodela polyrhiza, Stratiotes aloides*

Table 3. Dominant species (taxa) of the largest floodplain lakes in Latvia, July 2004.

Lake	Phytoplankton	Zooplankton	Macrozoobenthos	Macrophyta
Skuku	*Cryptomonas* sp.	*Graptoleberis testudinaria*, *Trichocerca myersi*	*Cloeon dipterum*, *Asellus aquaticus*, *Sphaerium corneum*, *Anisus contortus*	*Phragmites australis*, *Carex* sp.
Dvietes	*Cryptomonas* sp.	*Ceriodaphnia affinis*	Chironomidae, Oligochaeta, *Asellus aquaticus*	*Butomus umbellatus*, *Equisetum fluviatile*
Ljubasta	*Cryptomonas* sp.	*Synchaeta* sp., *Bosmina longirostris*	Chironomidae	*Stratiotes aloides*
Kosha	*Oscillatoria* sp.	*Pompholux complanata*	Chironomidae	Potamogeton spp., *Nuphar lutea*

The most widely distributed were the macrophytes. In the Lake Kosha, aquatic vegetation covered 15 % of total surface area, in the Lake Skuku – 70 %, Lake Ljubasta – 80 %, Lake Dvietes – more than 95 %. Meanwhile, rare species formed the highest abundance (biomass) within these communities rather than common ones. For example, Cladocera species *Ceriodaphnia affinis* formed more than 96 % of total number of zooplankton organisms in the Lake Dvietes, whereas in other lakes it was not found at all. Analogous, Cyanophyta *Oscillatoria* sp. dominated in the Lake Kosha but actually was not presented in other lakes (Table 3). Therefore, the observed communities could be characterised by low similarity between the sampling sites. It was validated also by calculations of Sorensen's and Renkonen's similarity coefficients based on qualitative and quantitative data.

Table 4. Values of similarity measures based on the community data of the largest floodplain lake ecosystems in Latvia, July 2004.

	Skuku	Dvietes	Kosha	Ljubasta
		Phytoplankton		
Skuku		48,19††	0,01††	42,33††
Dvietes	**0,36**†		0,01††	**50,33**††
Kosha	0,17†	0,22†		0,13††
Ljubasta	0,16†	0,18†	0,35†	
		Zooplankton		
Skuku		1,44††	3,45††	**13,39**††
Dvietes	0,27†		0,85††	3,00††
Kosha	0,09†	0,11†		2,08††
Ljubasta	**0,43**†	0,33†	0,31†	
		Macrozoobenthos		
Skuku		53,57††	6,41††	13.79††
Dvietes	0,73†		32,42††	40,64††
Kosha	0,50†	0,44†		**88,33**††
Ljubasta	0,60†	0,44†	**0,75**†	
		Macrophyta		
Skuku		-	-	-
Dvietes	**0,86**†		-	-
Kosha	0,72†	0,69†		-
Ljubasta	0,69†	0,76†	0,63†	

† - Sorensen's coefficients; †† - Renkonen's coefficients (percentage similarity)

During this study, different values of similarity were obtained depending on the systematic group analysed and the method (similarity measure) used for comparisons (Table 4). For example, according to both tests, the highest similarity based on zooplankton community data was found between the lakes Skuku and Ljubasta (Table 4). On the other hand, these lakes had very different macrozoobenthos communities. Therefore, observed lake ecosystems can't be classified together despite their common hydrology and morphometry. Based on these results, at least two different types of floodplain lake ecosystems can be distinguished:

1. Very shallow, frequently flooding eutrophic lakes dominated by Cryptophyta algae and characterised by rich macrophyta vegetation;
2. Relatively deep, rarely flooding eutrophic lakes dominated by Cyanophyta algae and characterised by scare macrophyta vegetation.

Discussion

The largest floodplain lake ecosystems in Latvia can be characterised by high biological diversity and very different phytoplankton, zooplankton, macrozoobenhos and macrophyta communities during the summer isolation phase.

Two types of floodplain lakes stated during this study can be related to two stages (clear and turbid) of shallow eutrophic lakes distinguished by Scheffer, Hosper, Meijer, Moss and Jeppesen (1993). In a clear water stage, seasonal growth of phytoplankton (especially Cyanophyta) is oppressed by intensive development of macrophytes. During the vegetation period, they accumulate dissolved nutrients from the water column therefore minimising its availability to the algae (Wetzel, 2001). In a turbid stage, growth of macrophytes is limited to narrow littoral zone and phytoplankton dominates the lake's water solumn. Similar situation was observed also during this study. In general, lakes Skuku, Dvietes and Ljubasta could be characterised by a clear water stage, whereas Lake Kosha – by a turbid.

The observed sharp contrast in species abundance and their distribution among these aquatic ecosystems has been supported by similarity and diversity analysis. A well-seen dominance of rare species in the community composition of these lakes could be related to an impact of annual flooding. According to Huszar and Reynolds (1997), the frequency of such periodic disturbances impacts the strength of species selection, the importance of interspecific competition and the diversity of species representation: annual flooding resists the maturation of successions, so competitive interactions are weak and diversity of species high.

Against all expectations, any correlation between species diversity and flooding conditions (seasonal water level fluctuation) has not been found. On the contrary, although the observed lakes have actually the same flooding amplitude, frequency and duration as well as very similar morhometric parameters (except for the Lake Kosha), differences in their ecosystems' structure and composition are essential. Other environmental factors, such as the lakes' morphometry, local runoff, water temperature, nutrient (food) availability, diversity of biotopes and so on, are probably more important than seasonal flooding, when comparing floodplain lake ecosystems at isolation phase.

Hydrological regime of the Daugava River in Latvia can be significantly changed by the construction and operation of new hydroelectric power stations on the Upper Daugava, in Byelorussia. According to the worlds' experience summarised by Avakjan, Saltankin and Sharapov (1987), the most adverse effect of river reservoir operation is upon the amplitude and duration of the flooding: the runoff of the river is diminished downstream of the dams, therefore, the amplitude of seasonal water level fluctuation is lowered and flooding duration shortened. In addition, because of an increased anthropogenic eutrophication, frequent Cyanophyta blooms as well as negative alternations in water quality of the Daugava River in summer can be predicted, based on previous observations in the Daugava's reservoirs by Druvietis (1998). These changes could decrease biological diversity within these lakes, affect seasonal migration and nesting of endangered bird species in their basins and so on. Therefore, floodplain lake ecosystems along the Daugava River could be affected in various ways. In this case, an international collaboration between Latvian and Byelorussian governments and Environmental Impact Assessment for the whole project is needed.

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References

Avakjan A. B., Saltankin V. P. & Sharapov V. A. (1987). *Reservoirs*. Mislj, Moscow.

Druvietis I. (1998). Observations on cyanobacteria blooms in Latvia's inland waters. In: *Harmful algae* (eds B. Reguera, I. Blanco, M. L. Fernández & T. Wyatt) pp. 35-36. Intergovernmental Oceanographic Commission of UNESCO, Xunta de Galicia.

Gruberts D. (2003). Ecological consequences of creation of the Daugava's reservoirs: Latvian experience. In: *International Environmental Experience: Applications of Belarus*. Collected papers (ed U. K. Slabin) pp. 52-55. Vitebsk.

Gruberts D. & Druvietis I., 2001. Phytoplankton periodicity in floodplain lake Grivas, Latvia – Baltic State. *In. Partnerships for sustainable life in lake ecosystems*. Proceedings of 9th International Conference on the Conservation and Management of Lakes, pp. 54-57. BIWAKO, Shiga.

Gruberts D., Paidere J., Priedītis J., Škute A., Druvietis I., Popels A., Parele E. & Enģele L., 2005. Biodiversity of the Daugava's floodplain lakes in Latvia. In: *Research and Conservation of Biological Diversity in Baltic Region*. 3rd International Conference. Book of Abstracts, p. 60. Baltic Society of Coleopterology, Daugavpils.

Huszar V. L. D. & Reynolds C. S., 1997. Phytoplankton periodicity and sequences of dominance in an Amazonian floodplain lake (Lago-Batata, Para, Brazil) – responses for gradual environmental change. *Hydrobiologia*, 346, 169-181.

Junk W. J., Bayley P. B. & Sparks R. E., 1989. The flood pulse concept in river-floodplain systems. In: Proceedings of the International Large River Symposium (ed D. P. Dodge). *Can. Spec. Publ. Fish. Aquat. Sci.*, 106, 110-127.

National Environmental Policy Plan for Latvia, 1995. Gandrs, Riga.

Račinskis E., 2004. Important Bird Areas of European Union importance in Latvia. Latvian Ornithological Society, Riga.

Scheffer M., Hosper S. H., Meijer M.-L., Moss B. & Jeppesen E., 1993. Alternative Equilibria in Shallow lakes. *TREE*, 8, 275-279.

Shannon C. E. & Weaner W., 1963. *The mathematical theory of communication*. University Press, Urbana.

Van den Brink F. W. B., Van Katwijk M. M. & Van der Velde G., 1994. Impact of hydrology on phyto- and zooplankton community composition in floodplain lakes along the lower Rhine and Meuse. *Journal of Plankton Research*, 16, 351-373.

Wetzel R. G., 2001. *Limnology. Lake and river ecosystems*. 3-rd ed. Academic Press, San Diego.

Systematics and distribution of zooplankton in Lake Victoria basin, Kenya

Masai D. Mutune, R. Omondi and M. Owili
Kenya Marine and Fisheries Research Institute, P.O. Box 1881, Kisumu, Kenya
E-mail: mmmutune@yahoo.co.uk

Abstract

Zooplankton samples were collected, using a 60μm nansen net, from Lake Victoria and adjacent water bodies with emphasis being placed on the different habitats within the ecosystems. A total of 116 species were identified, 63 rotifers, 24 cladocerans and 29 copepods. A number of these were new records for the zooplankton fauna for the country.

Introduction

Zooplankton play a vital role in energy transfer by grazing on phytoplankton and epithilic communities thereby converting plant material into animal protein which is transferable to higher trophic levels in the aquatic food chain. In addition to the transfer of energy in ecosystem zooplankters, especially Rotifera, can be used as indicators of water quality (Sladecek, 1983).

Gaps in the taxonomy of the lake's zooplankton fauna still exist, as certain habitats, particularly littoral, have not been studied in details. There is scanty data on the systematics of zooplankton of satellite lakes and other small water bodies in the region.

In the following study, an attempt was made to identify zooplankton species in Lake Victoria basin. The results hopefully will provide basic taxonomic information, which can be used for future specialized investigation of individual species and studies of secondary productivity.

Study area

The study was carried out in Lake Victoria Kenya, satellite lakes; Kanyaboli, Sare, Simbi and small water bodies (Figure 1) were sampled between 1995 and 2003.

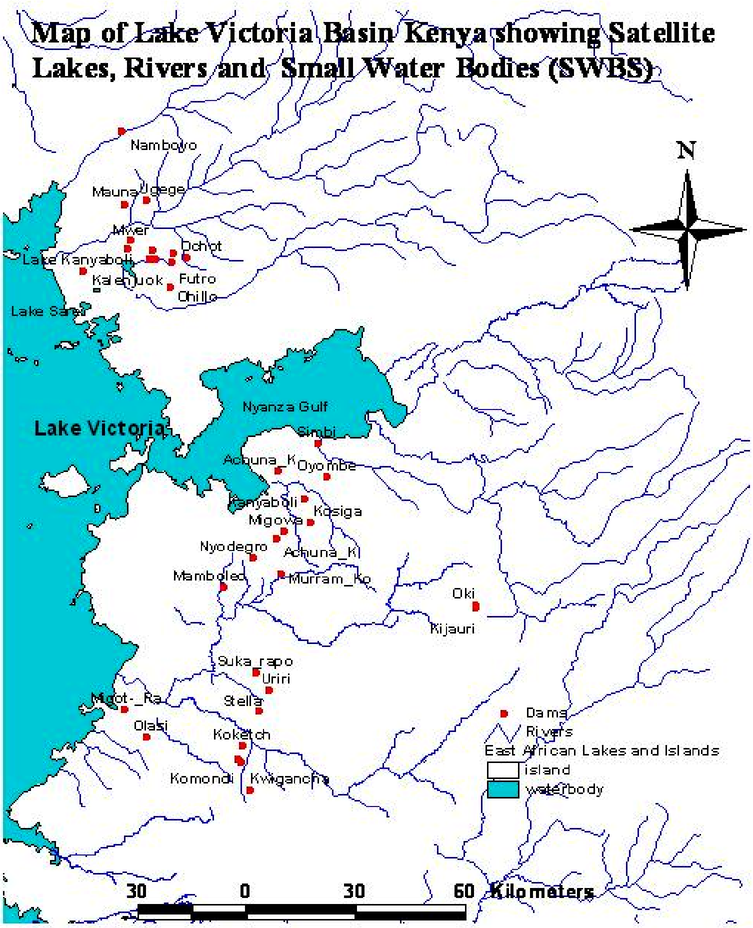

Figure 1. Map of the sampled sites.

Materials and methods

Zooplankton samples were collected with Nansen type plankton net of 76µm mesh size and 30cm mouth opening diameter 1.0m long. Vertical hauls were obtained as outlined by Tonolli (1971). Within the macrophytes, in station 1, a small hand net with a screen at the mouth was used. Samples were preserved in 5% formaldehyde solution. In the laboratory successive aliquots of every sample were examined under a Wild M3 dissection microscope. Rotifers were were sorted by ultra fine capillary tubes while Copepoda and Cladocera were sorted with the aid of fine needles. Copepoda and Cladocera were dissected and all specimens identified using morphological features under high magnification (X1000 oil immersion) of a Kyowa Medilux M12 compound microscope. Trophi of illoricated rotifers were obtained after dissolving the organisms in sodium hypochloride. They were then used to identify organisms upto species level . External morphological characteristics of the lorica were used to identify loricated Rotifers. The following taxonomic keys were used for specimen identification: Lindberg (1955, 1957), Van de Velde (1984), Herbst (1986), Smirnov (1996), Korovchinsky(1992), Segers (1995), Koste (1978) and Koste and Shiel (1987, 1990).

Results and discussions

Physico-chemical parameters

Lake Simbi had the highest mean temperature readings at 31°C followed by Lake Victoria at 26°C while the lowest was recorded in the SWBs at 25°C (Table 1).

Table 1. Physico-chemical parameters of the sampled water bodies.

	Victoria	**Kanyaboli**	Sare	**Simbi**	**SWBs**
Turbidity (NTU)	25.16 ± 7.24	43.00	5.15	25.60	36.60 ± 14.2
Conductivity (µScm$^{-1)}$	145.29± 11.10	287.00	103.00	670.00	249.00± 1.9
pH	7.54 ± 0.20	7.20	7.80	-	7.58 ± 0.08
Temperature (°C)	26.77 ± 0.50	27.00	27.00	31.00	25.64 ± 0.37
Dissolved oxygen (mgl^{-1})	7.86 ± 0.65	7.40	6.90	9.20	4.95 ± 0.51

The high dissolved oxygen content in L. Simbi could be attributed to the large population of *Spirulina* sp while the low values in the SWBs could be due to small water volumes and decomposing organic materials from both in and out of the ecosystems. Conductivity levels were highest in L. Simbi (670µScm^{-1}) and lowest in L. Sare (103µScm^{-1}). Turbidity levels were highest in the small water bodies. This could be so since agricultural areas surround most of the SWBs and also they are the watering point sources for livestock. They are also exposed to high levels of siltation

A total of 116 species of zooplankton were recorded: 29 copepods, 24 cladocerans and 63 rotifers (Table 2). Copepoda had only two families while Cyclopoida dominated with 22 species. Six families of Cladocera were identified of which the family Daphnidae dominated with nine species while Bosminidae and Sididae had only one species each. Rotifera had the highest number of families (16) and species (62). The family Lecanidae with 17 species followed by Brachionidae with 14 species dominated the group. The families Floscularidae, Philodinidae and Scaridiidae had only one species each.

Table 2: Zooplankton recorded from Lakes Victoria, Kanyaboli, Sare, Simbi and other small water bodies (SWBs)

TAXA	Victoria	Kanyaboli	Sare	Simbi	Swbs
COPEPODA					
Calanoida					
Diaptomidae					
Diaptominae					
Tropodiaptomus stulmanni Mrazek, 1895	+		+		+
Thermodiaptomus galeboides (Sars, 1901)	+		+		
Cyclopoida					
Cyclopinidae					
Cyclopinae					
Cryptocyclops linjancticus (Kiefer, 1928)	+				
C. gemellus (Gurney, 1928)					+
Mesocyclops ogunnus Onabamiro, 1957	+	+			+
M. kieferi Van de Velde, 1984	+				+
M. aspericornis (Daday, 1906)	+				
M. major Sars, 1927	+				+
M. tenuisaccus (Sars, 1927)					+
M. equatorialis equatorialis (Kiefer, 1927)	+				

Microcyclops jenkinae Lowndes, 1933					+
Microcyclops varicans (Sars, 1927)	+				
M. elgonensis Dussart, 1977					+
Thermocyclops decipiens (Kiefer, 1929)	+				
T. emini Mrazek, 1895	+				+
T. incisus Kiefer, 1932	+				
T. inopinus (Kiefer, 1926)			+		+
T. neglectus (Sars, 1909)	+				
T. oblongatus (Sars, 1927)	+				+
T. oithonoides (Sars, 1863)	+				
Eucyclopinae					
Ectocyclops rubescens Brady, 1940	+				
Eucyclops serrulatus Fischer, 1851	+				
Macrocyclops albidus Jurine, 1820	+				
Tropocyclops confinis (Kiefer, 1929)	+				+
CLADOCERA					
Ctenopoda					
Bosminidae					
Bosmina longirostris Muller, 1885	+		+		+
Chydoridae					
Alona davidi Richard, 1895	+				
A. guttata Sars, 1862	+				
A. karua King, 1853					+
A. pulchella King, 1853					+
Chydorus parvus (Daday, 1898)	+				
C. eurynotus Sars, 1901					+
C. parvus (Daday, 1898)					+
Dunhuvedia colombiensis Stingen, 1913	+				
Daphnidae					
Ceriodaphnia cornuta Sars, 1885	+	+	+		+
C. quadrangula (Muller, 1776)	+	+	+	+	+
Daphnia barbata Weltner, 1898	+				+
D. lumholtzi Sars, 1885	+				+
D. laevis Birge 1878	+				+
D. longispina Muller, 1785	+				
D. magna Straus, 1820	+				
Simocephalus expinosus (Koch, 1841)					+
S. vetulus (Müller, 1776)					+
Macrothricidae					
Macrothrix hirsuticornis Norman, 1867					+
M. spinosa King, 1853	+				+
M. triserialis Brandy, 1836					+
Moinidae					
Moina macrocopa (Strauss, 1820)	+		+		
M. micrura Kurz,1874	+	+	+	+	+
Sididae					
Diaphanosoma excisum Sars 1885	+	+	+	+	+
ROTIFERA					
Asplanchnidae					
Asplanchna brightwelli (Gosse, 1850)	+				+
A. sieboldi (Leydig, 1854)	+	+			+
Brachionidae					
Brachionus angularis (Gosse, 1851)	+	+			+
B. bidentatus Anderson, 1889	+				+
B. calyciflorus Pallas, 1776	+	+	+		+
B. caudatus Barrois & Daday, 1894	+				+
B. dimidiatus Bryce, 1931				+	
B. diversicornis (Daday, 1883)					+
B. falcatus Zacharias, 1898	+				+
B. quadridentatus (Herman, 1773)	+				+
B. rubens Ehrenberg, 1838	+				+
B. urceolaris (Müller, 1773)					+
Keratella cochlearis (Gosse, 1851)	+				
K. tropica (Apstein, 1907)	+	+	+		+
Plationus patulus (Müller, 1786)	+				+
Platyias quadricornis (Ehrenberg, 1832)	+				+
Colurellidae					

Lepadella patella (Müller, 1786)					+
Heterolepadella heterodactyla (Bartros, 1955)					+
Epiphanidae					
Epiphanes clavulata (Ehrenberg, 1832)	+	+			+
E. macroura (Barrois & Daday, 1894)	+				+
Euchlanidae					
Dipleuchlanis propatula (Gosse, 1886)	+				+
Euchlanis dilatata Ehrenberg, 1832					+
E. calpida (Myers, 1930)					+
E. triquetra Ehrenberg, 1838	+				+
Filinidae					
Filina longiseta (Ehrenberg, 1898)	+	+			
F. opoliensis (Zacharias, 1898)	+				+
F. terminalis (Plate, 1886)	+	+			+
Floscularidae					
Sinantherina socialis Linnaeus, 1758	+				
Hexarthridae					
Hexarthra mira (Hudson, 1831)	+				+
H. jenkinae (Beauchamp, 1932)				+	
Lecanidae					
Lecane bulla Gosse 1851	+	+			+
L. ludwigii (Eckstain, 1833)					+
L. clostercerca (Schmarda, 1859)					+
L. thienemanni (Hauer, 1938)					+
L. lunaris (Ehrenberg, 1832)					+
L. leontina (Turner, 1892)		+			+
L. ungulata (Gosse, 1887)		+			+
L. curvicornis (Murray, 1913a)		+			+
L, furcata *(Murray, 1939a)*					+
L. braumi (Koste, 1988)					+
L. papuana *(Murray, 1913c)*		+			+
L. hastata *(Murray, 1913a)*					+
L. hornemani (Ehrenberg, 1834)		+			+
L. monostyla (Daday, 1897)					+
L. tenuiseta (Harring, 1914)					+
L. hamata (Stoks, 1896)					+
L. ungulata (Gosse, 1837)	+				
Mytilinidae					
Lopocharis salpina (Ehrenberg, 1834)					+
Mytilina ventralis (Ehrenberg, 1832)	+				+
Philodinidae					
Rotaria neptunia (Ehrenberg, 1832)	+				+
Scaridiidae					
Scaridium longicaudum (Muller, 1786)					+
Synchaetidae					
Polyathra vulgaris Carlin, 1943	+				+
P. longiremis Carlin, 1943					+
Testudinellidae					
Testudinella emerginula (Sternoos, 1898)	+				+
T. patina (Herman, 1783)					+
Trichocercidae					
Trichorcerca braziliensis (Murray, 1913)	+				+
T. elongata (Gosse, 1886)	+	+			+
T longiseta (Shrank, 1802)	+				+
T. rattus (Muller, 1776)	+				+
Trichotriidae					
Trichotria buchneri Koste, 1988					+
T. subquadratus (Perty, 1850)					+
T. tetractis (Koste, 1984)					+

The small water bodies had the highest number of species (85) while the lowest number was recorded in Lake Simbi (5). The high number of species recorded under SWBs could have been due to the high number of ecosystems sampled with varying habitats. These are generally shallow water bodies with heavy macrophytic growth thus the prominence of groups that prefer such dwellings e.g. Lecanidae. Many species in the SWBs could have been introduced from Lake Victoria during fish translocations. The low number of species in Lake Simbi, on the other hand, could be attributed to the harsh conditions. This is a small saline lake with a conductivity of 670 $\mu s\ cm^{-1}$. Many of the organisms are new records for the Kenyan zooplankton fauna.

Conclusions

The results of the study reveals that there many more zooplankton species yet to be recorded in Kenya and indeed many in the check list are new records for the country. It is also clear that for systematic samples there is need to venture into all habitats within an ecosystem as was realized species in with Lake Victoria which had earlier not been reported despite a number of studies.

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References

Herbst, H.-V., 1986. Beschreibung des *Thermocylops hastatus antillensis* n. sp. Miteinem Bestimmung ischlussel fur die Gattung *Thermocyclops* Kiefer, 1927. Bijdr. Dierkunde 56 (1): 165-180. (Translation)

Korovchinsky, N. M., 1992. *Sididae and Holopidae (Crustacea: Daphniformes). Guides to the identification of the Microinvertebrates of the continental waters of the world*. 3, SPB. The Hague, 82pp.

Koste, W., 1978. *Rotaria Die Reidertiere Mitteleuropas Bestimmungswerk begrundet Von Max Volgt*. 2 Vols. Borntraeger, Struttgart. 1229pp.

Koste, W and R. J. Shiel.,1987. Rotifera from Australian Inland waters, II Epiphanidae and Brachionidae (Rotifera: Monogononta). *Aust. J. Mar. Freshwat. Res*. 37: 765-792.

Lindberg, K. 1955 *Cyclopoids (Crustaces: Copepodes)*. Recoltes Au Perou Par Le De Hernad De Macedo Anuaro de la Estacion Altoandia de Biologia Itacidinda Checayani azangaro puno- Peru. Vol 1. 18pp.

Lindberg, K., 1957. Cyclopedes (Crustaces: Copepodes) de la Cote d' Ivoire. Bulletin de I.N.F.A.N.T. XIX ser. No 1 134-179.

Segers, H., 1995. *The Lecanidae (Monogononta). Guides to the identification of the microinvertebrates of the continental waters of the world vol. 2*. SPB Academic Publishing bv, the Hague 226pp.

Sladecek, V., 1983. Rotifers as indicators of water quality. *Hydrobiologia* 100: 169-201.

Smirnov, N. N., 1996. *Cladocera: Chyrodinae and Saycinnae (Chydoridae) of the world. Guides to the identification of the microinvertebrates of the continental waters of the world*. SPB Academic Publishing bv, The Netherlands, 197pp.

Tonolli, V., 1971. Zooplankton: Methods of collection. *In.* Edmondson, W.T and G.G. Winberg (eds). *A Manual on Methods for the Assessment of Secondary Productivity in Fresh Waters*. International Biological Programme. Blackwell Scientific Publications, Oxford and Edinburgh, U.K.

Van de Velde, I., 1984. Revision of the African species of genus *Mesocyclops* Sars, 1914 (Copepoda: Cyclopoidae). *Hydrobiologia* 109: 3-66.

Disruption in the distribution of fish fauna simultaneous with the increase of introduced Bluegill and Largemouth Bass in a lagoon of Lake Biwa, Japan

Masahiro Nakagawa

Department of Fisheries, Faculty of Agriculture, Kinki University, Nakamachi 3327-204, Nara 631-8505, Japan
Email-gf998002@yahoo.co.jp

Abstract

To understand the mechanism of decrease in biodiversity around Lake Biwa, seasonal and annual changes in fish fauna were surveyed in Katada-naiko Lake, which is located at the western part of Lake Biwa. Specimens were collected by casting net sampling for 20 min each night from August 2001 to May 2005. Initially, to examine seasonal changes in fish fauna, specimens were collected on 112 nights in the first year. To examine annual changes in fish fauna, specimens were collected on 22, 15, and 12 nights in the second, third, and fourth year, respectively. Samples included 2,314 specimens, representing 15 fishes and 3 crustaceans. In the first year, 3 cyprinid species crucian carps, Pseudorasbora parva, and Rhodeus ocellatus ocellatus were the most abundant, constituting 40.4%, 19.8%, and 11.4%, respectively, of the total specimens, followed by Ischikauia steenackeri (4.2%), bluegill (2.4%), and Abbottina rivularis (2.3%). Crucian carps were frequently collected from March to July, especially in May. P. parva, and R. o. ocellatus were frequency collected from January to March and from January to June, respectively. During the four study periods, the frequency of the three most abundant cyprinid fishes dramatically changed to 71.6%, 80.5%, 64.5%, and 16.6%, whereas that of two invasive species bluegills and largemouth bass changed to 3.4%, 4.2%, 15.2%, and 68.9%. In the fourth year, P. parva, R. o. ocellatus, and Rhinogobius sp. disappeared, and 2 crustaceans, except for red swamp crayfish significantly decreased. This suggests that the decrease in dominant species was simultaneous with the increase in bluegill and largemouth bass; switching of dominant species occurred suddenly after the frequency of bluegill and largemouth bass surpassed 15% of the total fish population. Therefore, the population size of the two invasive alien species introduced into the waters should be strictly controlled by selective elimination of them.

Key words: dominant species, invasive alien species, Katada naiko

Introduction

“Katada naiko” of Lake Biwa in Shiga prefecture of central Japan is an attached lake, which is geologically categorized as lagoon and ecologically defined as ecotone between terrestrial and aquatic ecosystems. The lake has escaped vast reclamation by public land reforms and produced some cyprinid species as the dominant species, small populations of native medaka *Oryzias latipes*, and introduced predatory species such as largemouth bass *Micropterus salmoides* and bluegill *Lepomis macrochirus* (Minobe and Kuwamura, 2002). It is remarkable that the currently dominant species in the lake are not the two introduced species from North America but the cyprinid species in the 1990s.

In general, “Naiko” has several indirect economic values and plays an important role in purifying water, providing biotopes for aquatic organisms, and supplying water for irrigation. Previously, certain naiko had been completely and partly reclaimed for agricultural lands; however, few naiko have been restored for improving their functions. Thus, the ecosystem of naiko has been receiving increasing attention in both the limnology and biology literatures.

Therefore, examining the fish fauna in Katada naiko is a valuable model in understanding the fish community before the invasion by the two species and the process of disruption in the distribution of fish fauna. The study of quantitative examination focused on the monthly and annual changes in fish fauna and the transition of dominant species in Katada naiko. In this paper, several novel findings and suggestive data pertaining to the above-mentioned aspects have been reported.

Materials and methods

The sampling site is a part of Katada naiko (E135°55'30", N35°07'10"), which is located at the western part of Lake Biwa. The lake consists of 4 areas linked by channels. The samplings were performed at the eastern parts of the region with the third largest surface area. Mud forms a large part of the bottom of the area around the site. Specimens were collected with a casting net (mesh size 10 mm, net length 3 m) for 20 min (7 casts in one sampling) from 2100 to 2400 in order to maintain a constant accuracy throughout the process, with the only exceptions being the gaps between sample collection and the daylight conditions. The methods of sampling prepared in advance were sufficiently constant for the quantitative investigation.

The samplings were performed 161 times during a span of 4 years. In the first year, to examine the seasonal changes in fish fauna, specimens were collected on 112 nights. To examine the annual changes in fish fauna and to prove the transition of dominant species by using the index of catch per unit of effort (CPUE), specimens were continuously collected on 22, 15, and 12 nights in the second (September 2002 to July 2003), third (November 2003 to July 2004), and fourth year (November 2004 to May 2005), respectively. The CPUE was calculated by dividing the total number of specimens of each species by the number of samplings, and the ranks of dominant species were separately determined for the fish and crustacean species.

Specimens including the crustaceans were mainly fixed with 10% formalin solution. Although the specimens were generally identified according to the method used by Nakabo (2000), three types of crucian carps, including the round crucian carp *Carassius buergeri grandoculis*, the deep bodied crucian carp *C. cuvieri*, and the silver crucian carp *C.* sp. were simply categorized as "crucian carp" or "*Carassius* sp." In crustaceans, the river shrimp *Palaemon paucidens* and the freshwater prawn *Macrobrachium nipponense* were categorized as "shrimps" and *Procambarus clarkii* was categorized as "crayfish".

Results

Monthly changes in fish fauna

Table 1 shows the monthly changes in the species composition of fishes and crustaceans collected in the first year. Three cyprinid species—crucian carps, stone moroko *Pseudorasbora parva*, and rosy bitterling *Rhodeus ocellatus ocellatus*—were the most abundant, constituting 40.4% (695 fish), 19.8% (341 fish), and 11.4% (197 fish), respectively, of the total 1721 specimens. These were followed by lake weed chub *Ischikauia steenackeri* (4.2%; 73 fish), bluegill (2.4%; 42 fish), Chinese false gudgeon *Abbottina rivularis* (2.3%; 39 fish), Biwa gudgeon *Gnathopogon caerulescens* (2.2%; 38 fish), carp *Cyprinus carpio* (0.9%; 16 fish), the common freshwater goby *Rhinogobius* sp. (0.9%; 16 fish), largemouth bass (0.9%; 16 fish), dark chub *Zacco sieboldii* (0.2%; 4 fish), flat bitterling *Achilognathus rhombeus*, the far eastern catfish *Silurus asotus* (0.06%; 1 fish), and the northern snakehead *Channa argus* (0.06%; 1 fish). Shrimps were 14.0% (241 individuals), whereas no crayfish were collected.

The cyprinid fishes occurred seasonally in this study. The crucian carps were frequently collected from March to July, particularly in the breeding season, i.e., in May. Stone moroko and rosy bitterling were frequently collected from January to March and January to June, respectively. The lake weed chub were collected throughout the study, except from November to January. The short-lived Chinese false gudgeon and Biwa gudgeon were frequently collected between autumn and spring. The freshwater common goby also occurred seasonally and were mainly collected from December to March. Shrimps were frequently collected from February to March. These show that the fishes have shared the shore of Katada naiko seasonally. However, sunfishes were sparsely collected all year around, except on 10 June 2002 when 10 bluegills were collected (data not shown).

Annual changes in fish fauna

Figure 1 shows the percentage of fishes and their individual numbers in Katadanaiko. In the second year, the species composition was generally similar to that in the first year. The 3 cyprinid species-crucian carps, stone moroko, and rosy bitterling were still the most abundant, constituting 60.9% (173 fish), 8.8% (25 fish), and 10.2% (29 fish), respectively, of the total 284 specimens. They were followed by Chinese false gudgeon (4.9%; 14 fish), bluegill (3.5%; 10 fish), lake weed chub (1.4%; 4 fish), Biwa gudgeon (1.4%; 4 fish), carp (1.4%; 4 fish), common freshwater goby (1.1%; 3 fish), largemouth bass (0.7%; 2 fish), and dark chub (0.4%; 1 fish). Shrimps constituted 4.6% (13 individuals), whereas crayfish constituted 0.4% (1 individual) of the total specimens collected.

In the third year, the number of rosy bitterling—one of the dominant species—decreased and that of bluegill increased. Crucian carps, stone moroko, and bluegill were the most abundant, constituting 48.7% (77 fish), 12.0% (19 fish), and 10.8% (17 fish), respectively, of the total 158 specimens. They were followed by Chinese false gudgeon (5.1%; 8 fish), largemouth bass (4.4%; 7 fish), rosy bitterling (3.8%; 6 fish), lake weed chub (3.2; 5 fish), Biwa gudgeon (3.2; 5 fish), carp (1.9%; 3 fish), and common freshwater goby (0.6%; 1 fish). Shrimps constituted 1.3% (2 individuals), whereas crayfish constituted 5.1% (8 individuals) of the total specimens collected.

In the fourth year, the species composition dramatically differed from that in the first year and was similar to that of the other attached lakes around Lake Biwa. Bluegill, crucian carps, and largemouth bass were the most abundant species, constituting 60.3% (91 fish), 16.6% (25 fish), and 8.6% (13 fish), respectively, of the total 151 specimens. They were followed by lake weed chub (4.6%; 7 fish), pale chub *Z. platypus* (3.3%; 5 fish), Chinese false gudgeon (2.6%; 4 fish), carp (1.3%; 2 fish), and Biwa gudgeon (0.7%; 1 fish). Shrimps constituted 0.7% (1 individual), whereas crayfish constituted 1.3% (2 individuals) of the total specimens collected. Stone moroko and rosy bitterling, which had been the dominant fishes, as well as common freshwater goby in the site, were not collected in the term.

Table 1. Monthly changes in the species composition in fishes and crustaceans collected at Katada naiko in Shiga prefecture from August 2001 to August 2002.

Species	Month	AUG	SEP	OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	Total
	Air temperature (°C)	N.D.	21.3 ±2.9	14.4 ±2.3	6.9 ±1.4	4.1 ±1.7	5.0 ±2.6	2.3 ±1.7	5.2 ±3.2	2.4 ±3.9	16.8 ±1.9	24.6 ±2.5	30.4 ±2.7	29.6 ±2.1	
	Water temperature (°C)	N.D.	21.8 ±2.9	16.3 ±1.1	9.5 ±1.9	6.6 ±1.8	5.7 ±1.6	5.3 ±1.3	9.6 ±2.0	14.7 ±3.2	18.7 ±1.6	24.0 ±1.6	29.3 ±1.9	29.4 ±2.0	
	Sampling frequency (days)	1	7	8	8	10	11	8	14	8	8	9	12	8	112
Fishes															
Carassius sp.		7	52	42	41	36	51	16	72	72	111	71	72	52	695
Pseudorasbora parva		0	6	20	15	43	50	46	121	27	3	5	5	0	341
Rhodeus ocellatus ocellatus		1	8	7	2	9	24	19	60	14	10	31	6	6	197
Ischikauia steenackeri		3	10	4	0	0	0	1	13	2	4	15	16	5	73
Lepomis macrochirus		2	0	1	2	9	1	0	7	4	1	13	2	0	42
Abbottina rivularis		1	0	1	9	11	2	4	6	5	0	0	0	0	39
Gnathopogon caerulescens		0	0	0	5	12	3	5	13	0	0	0	0	0	38
Rhinogobius sp.		0	0	0	0	3	4	4	3	0	0	2	0	0	16
Micropterus salmoides		0	2	1	2	1	3	0	1	1	2	0	2	1	16
Cyprinus carpio		0	2	2	2	0	1	1	0	0	1	1	2	4	16
Zacco sieboldii		0	0	0	0	0	0	0	0	0	1	1	1	1	4
Acheilognathus rhombeus		0	1	0	0	0	0	0	0	0	0	0	0	0	1
Silurus asotus		0	0	0	0	0	0	1	0	0	0	0	0	0	1
Channa argus		0	0	0	0	0	0	0	0	1	0	0	0	0	1
Crustaceans															
Shrimps		0	1	24	9	10	23	39	88	19	4	13	8	3	241
Total		14	82	102	87	134	162	136	384	145	137	152	114	72	1721

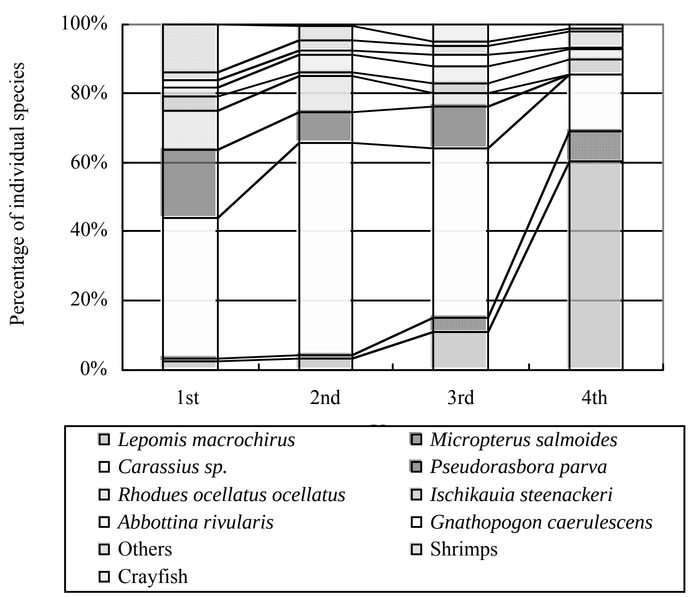

Figure 1. Annual changes in fish fauna in Katada naiko. Specimens were collected on 112, 22, 15, and 12 nights in the first (August 2001 to August 2002), second (September 2002 to July 2003), third (November 2003 to July 2004), and fourth year (November 2004 to May 2005), respectively. In crustaceans, *Palaemon paucidens* and *Macrobrachium nipponense* were categorized as "Shrimps" and *Procambarus clarkii* was categorized as "Crayfish".

Annual changes in the CPUE

Table 2 shows the annual changes in CPUE in Katada naiko. The CPUEs of 3 cyprinid dominants in the first and second years were more than 9.4; however, the CPUE of the three fishes decreased in the third year (6.40) and fourth year (2.08 of only crucian carps). The CPUE of shrimps also sharply decreased in the third and fourth years. In contrast, the CPUE of bluegill sharply increased with a gradual increase in the CPUE of largemouth bass. Lake weed chub and Chinese false gudgeon showed no significant changes in the CPUE, which ranged from 0.3 to 0.7. The CPUE of crayfish (0.53) in the third year, which was similar to that of the shrimp in the fourth year, decreased in the fourth year (0.17). The CPUE of the total specimens was mostly contributed by the 3 cyprinid fish and shrimps in the first year, whereas by crucian carps and the introduced sunfishes in the forth year.

Table 2. Annual changes in CPUE and dominant species in Katada naiko †

Species	Average of individual numbers in one sampling trial (rank)			
	1st year	2nd year	3rd year	4th year
Fishes				
Carassius sp.	6.21 (1)	7.21 (1)	5.13 (1)	2.08 (2)
Pseudorasbora parva	3.04 (2)	1.04 (2)	1.27 (2)	0.00 (–)
Rhodeus ocellatus ocellatus	1.76 (3)	1.21 (3)	0.40 (6)	0.00 (–)
Ischikauia steenackeri	0.65 (4)	0.17 (6)	0.33 (7)	0.58 (4)
Lepomis macrochirus	0.38 (5)	0.42 (5)	1.13 (3)	7.58 (1)
Abbottina rivularis	0.35 (6)	0.58 (4)	0.53 (4)	0.33 (6)
Gnathopogon caerulescens	0.34 (7)	0.17 (6)	0.33 (7)	0.08 (8)
Rhinogobius sp.	0.14 (8)	0.17 (6)	0.20 (9)	0.17 (7)
Cyprinus carpio	0.14 (8)	0.13 (9)	0.07 (10)	0.00 (–)
Micropterus salmoides	0.14 (8)	0.08 (10)	0.47 (5)	1.08 (3)
Zacco sieboldii	0.04 (11)	0.04 (11)	0.00 (–)	0.00 (–)
Acheilognathus rhombeus	0.01 (12)	0.00 (–)	0.00 (–)	0.00 (–)
Silurus asotus	0.01 (12)	0.00 (–)	0.00 (–)	0.00 (–)
Channa argus	0.01 (12)	0.00 (–)	0.00 (–)	0.00 (–)
Zacco platypus	0.00 (–)	0.00 (–)	0.00 (–)	0.42 (5)
Crustaceans				
Shrimps	2.14 (1)	0.50 (1)	0.13 (2)	0.08 (2)
Crayfish	0.00 (2)	0.04 (2)	0.53 (1)	0.17 (1)
Total	15.36	11.76	10.52	12.57

†Specimens were collected on 112, 22, 15, and 12 nights in the first (August 2001 to August 2002), second (September 2002 to July 2003), third (November 2003 to July 2004), and fourth year (November 2004 to May 2005), respectively. In crustaceans, *Palaemon paucidens* and *Macrobrachium nipponense* were categorized as "Shrimps" and *Procambarus clarkii* was categorized as "Crayfish".

Discussion

Disruption in the fish fauna distribution after establishment of the population of the introduced bluegill and largemouth bass around Lake Biwa has been receiving increasing attention, particularly since the two species were frequently found around the coastal areas of the lake in the early 1980s (Maehata, 1993). In order to comprehend the monthly and annual changes in the fish fauna and the impacts caused by the two introduced species, the fish fauna in Katada naiko was examined from August 2001 to May 2005 in this study by using a casting net. The important findings of this study are as follows: (1) 3 cyprinid fishes were the most abundant during the first two years from August 2001 to July 2003, (2) remarkable and swift changes occurred in the fish fauna since the beginning of the third year, i.e., from November 2003, and (3) disruption in the distribution of fish fauna was simultaneous with the increase in the population of the introduced bluegill and largemouth bass. These findings have been discussed below.

1. The three dominant fishes in the first year were crucian carps, stone moroko, and rosy bitterling. This result is inconsistent with that of a previous study in which a fixed shore net

was used in 1994 and 1995 reported only stone moroko and rosy bitterling as the dominant fishes (Minobe and Kuwamura, 2001). This difference could be attributed to the difference in the sampling points and sampling methods. However, these data are fairly consistent with each other in identifying bluegill and largemouth bass as the minority population. This indicates that Katada naiko had functioned as a biotope of cyprinid fishes until around 2002. The occupation by cyprinid fishes was unusual in the simplified ecosystems of the other attached lakes around Lake Biwa and was worth investigating.

2. The remarkable shift in the type of dominant species did not occur gradually but occurred rapidly in just a few years. The phenomenon of decrease in the number of stone moroko and rosy bitterling with an increase in the number of largemouth bass is in good accordance with the report by Izu-numa in Miyagi Prefecture, Tohoku region (Takahashi, 2002). Because stone moroko is one of the fishes with the strongest resistance against water pollution (Uchiyama, 2001) and the water quality of Katada naiko has been improved recently (Nakagawa, 2005), the decrease in the number of individuals of this species cannot be attributed to pollution. There were sufficient bivalves, at least during the study period, to facilitate the deposition of eggs by rosy bitterling during the breeding season. On the other hand, a recent study in an aquarium demonstrated that adult bluegills frequently prey on stone moroko (Katano *et al*., 2003), and largemouth bass in this site frequently preyed on several fishes, which was confirmed from the analysis of the stomach contents (Nakagawa, unpublished data). Moreover, there are many reports suggesting that the invasion by largemouth bass had negative effects on the cyprinid fish population in Japanese waters (Environmental Agency, 2004). These results indicate that the remarkable decreases in the stone moroko and rosy bitterling populations in this study were certainly due to predation and/or competition by increased bluegill and largemouth bass populations. The studies conducted in the future should examine the population dynamics of lake weed chub and Chinese false gudgeon and investigate the transition of body size of the prey before and after an increase in the bluegill and largemouth bass populations.

3. In this study, the switching of dominant species occurred suddenly after the sampling frequency of bluegill and largemouth bass surpassed 15% of the total fish population. It must be emphasized that synergism involving several invasive species might accelerate disruption in the distribution of an ecosystem and reduce resistance to the non-native species. Water lettuce *Pistia stratiotes* from tropical Africa, which is categorized as an invasive alien plant, had occupied the surface of the sampling site in the summer of 2004, and transparency in the lake has risen (Nakagawa, 2005). Increase in the visibility will be advantageous for predation, resulting in the propagation of the two species, and may be also advantageous for selective collection and extermination of these species by using a spear in the small and shallow Katada naiko. Because selective collection of invasive alien species can result in an increase in the population of another introduced species (Maezono and Miyashita, 2004), the population of the two species should be selectively controlled by collecting the other introduced species to prevent any secondary problem.

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References

Environmental Agency, 2004. *Impact on native biotic community and ecosystem by black bass and bluegill and the provision*. Japan Wildlife Research Center, Tokyo (in Japanese).

Katano O., Nakamura T. & Yamamoto S., 2003. Predation of Japanese minnow *Pseudorasbora parva* by bluegill *Lepomis macrochirus* in experimental aquaria. *Nippon Suisan Gakkaishi*. 69, 733–737 (in Japanese with English abstract).

Maehata M., 1993. Dynamics of fishes around Biwako Bunkakan (South Lake). *The Annual Reports of the Biwako Bunkakan*. 11, 43–49 (in Japanese).

Maezono Y. & Miyashita T., 2004. Impact of exotic fish removal on native communities in farm ponds. *Ecol Res*. 19, 263–267.

Minobe H. & Kuwamura K., 2001. Analysis of the change in fish fauna and ecological environment in “naiko” around Lake Biwa. *Ecol. Civil Eng*. 4, 27–38 (in Japanese with English abstract).

Nakabo T., 2000. *Fishes of Japan with pictorial keys to the species, 2nd eds*. Tokai University, Tokyo (in Japanese).

Nakagawa M., 2005. Present state of *Pistia stratiotes* growing at Katada Naiko, Shiga Prefecture in 2004. *Bulletin of Kansai Organization for Nature Conservation*. 27, 53–56 (in Japanese).

Nakajima T., Fujioka Y., Maehata M., *et al*., 2001. Relationship between fish distribution pattern and geomorphology in the Konan area surrounding the southern part of Lake Biwa. *Jpn. J. Limnol*. 62, 261–270 (in Japanese with English abstract). Takahashi K., 2002. Impact by largemouth bass on fish community– the case of Izu-numa and Uchi-numa.

In: *Invader of River and Lakes and Marshes, Black bass- its Biology and Impacts on Ecosystem*. (ed the Japanese Society of Japan) pp. 47–59. Kouseisha-kouseikaku, Tokyo (in Japanese).

Uchiyama R., 2001 *Pseudorasbora parva*. *In.* Kawanabe H., Minuno N. & Hosoya K. eds. *Freshwater fishes of Japan*. pp. 302–305. Yamatokeikokusha, Tokyo (in Japanese).

Feeding habits of introduced Largemouth Bass around Lake Biwa and in ponds without Mesopredators in Japan

Masahiro Nakagawa

Department of Fisheries, Faculty of Agriculture, Kinki University., Nakamachi 3327-204, Nara 631-8505, Japan
Email : gf998002@yahoo.co.jp

Abstract

To understand the feeding habits of largemouth bass *Micropterus salmoides* in Japanese inland waters, the stomach contents of fish from three rivers, one satellite lake around Lake Biwa, and two closed water systems without mesopredators that differed in catchment landscape, were examined. Specimens from lotic areas were collected from Iso River in spring 2001; from Etsura River, spring 2002 and 2003; and from Kawamichi River, summer 2001. Specimens from lentic areas were collected from Katada-naiko Lake from October 2001 to May 2005; Pond A, summer 2001; and Pond B, summer 2002. To estimate the frequency of occurrence of prey, the stomach contents were categorized into fishes, shrimps, aquatic and terrestrial insects, daphnia and midges, and others. The main prey in Iso River was fishes, including *Gymnogobius isaza*, which is an endemic goby of Lake Biwa and is categorized as “near threatened” in the Japanese Red Data Book. The main prey in Etsura River remarkably changed from fishes (86.0%) in 2002 to shrimps (87.1%) in 2003. The main prey of fry in Kawamichi River was fishes and shrimps. The main prey in the lentic areas were fishes in Katada-naiko Lake, aquatic and terrestrial insects in Pond A, and daphnia and midges in Pond B. These findings indicate that depending on their established habitats, largemouth bass mainly prey on fishes and shrimps around Lake Biwa; they dramatically change their main feeding habits within a short time in the same habitat, and they normally prey on fishes and/or shrimps irrespective of the habitat area, i.e., rivers or lakes. These findings also demonstrated that the largemouth bass residing in closed water systems without mesopredators frequently prey on aquatic and terrestrial insects or daphnia and midges, and the top-down effects on account of direct predation will lead to confusion in the trophic cascade.

Key words: endangered species, *Gymnogobius isaza*, invasive alien species

Introduction

Largemouth bass *Micropterus salmoides* is a typical invasive alien fish species that belongs to the family Centrarchidae of the order Perciformes. The largemouth bass, which is naturally distributed in the northern part of America, was primarily introduced into Lake Ashinoko in central Japan in 1925 and was then artificially dispersed in Japanese inland waters. The species is now widely distributed over 74% of the waters out of the examined 227 water bodies, including 94 dam lakes (Environmental Agency, 2004), because it possesses the ability to adapt and thereby establish itself in various environmental conditions due to its predominant feeding, growth, and reproductive habits (Yodo, 2002).

Largemouth bass has already established in Lake Biwa, which is the largest lake in Japan (surface area, 670.5 km^2; maximum depth, 104 m), a hotspot of biodiversity (more than 50 fish species) with some endemic species. The fish was first collected in 1974 from the northern part of the lake, and since the early 1980s, mass propagation was recorded throughout the coast of the lake (Maehata, 1993). It is common knowledge that the yields from fisheries specific for other fish species as well as the biodiversity in the lake has shown a decline with the increase in the population of largemouth bass in the lake.

On one hand, the species must have occupied many ponds out of more than 200,000 ponds in Japan, although very little information on this subject is currently available in the published literature. Some ponds have recently been considered to be important because they have functioned as refuges for endangered fishes in terms of *in situ* conservation (*e.g.*, Kawamura & Hosoya, 1997). In some ponds with limited capacity, mesopredators are often exhausted by the indiscriminate introduction of fish. Therefore, the feeding habits of the largemouth bass might be unusual.

In order to accumulate case studies and obtain new findings regarding the feeding habits of the largemouth bass and for the assessment of its negative effects on the ecosystem, this study of the stomach contents of the species focused on prey selectivity, attack on propagation of an endemic goby, and the top-down effect on low trophic levels in closed water systems. In this study, the author has reported several novel findings and suggestive data pertaining to the above-mentioned aspects.

Material and methods

Sampling sites

This study was performed in Lake Biwa water system in Kinki region, Japan. The sampling sites around Lake Biwa in Shiga Prefecture were as follows: the Iso River (E136°15'50'', N35°18'15''; width of the river, 5 m) in Sakata county, the Etsura River (E136°12'55'', N35°15'15''; width, 20 m) in Hikone city, the Kawamichi River (E136°14'30'', N35°23'30''; width, 6 m) in Higashi-azai county, and a part of the Katada-naiko lake (E135°55'30'', N35°07'10''; surface area, 2340 m^2) in Otsu city. The population size of the largemouth bass in the Katada-naiko Lake had been constant at low levels until 2003, but has increased recently (Nakagawa, unpublished data). The sampling sites of the ponds were as follows: Pond A (E135°44'45", N34°46'45"; 150 m^2) in Tanabe county of Kyoto Prefecture and Pond B (E135°53'40", N34°52'40"; 200 m^2) in Otsu city of Shiga Prefecture. In both the ponds, no fishes, except the largemouth bass and a few big carps (only Pond A), and no shrimps, except, crayfish were observed.

Sampling methods

All the specimens from the 3 rivers and the Katada-naiko Lake were collected using a casting net. Specimens from Pond A and Pond B were collected by fishing, except for 3 specimens that were colleted using a hand net and 7 specimens that were collected using a casting net, respectively. In order to prevent largemouth bass from swallowing fishing baits, which could result in mistaken assumption of the bait as part of the stomach contents, sausage and angleworm were used as baits. The specimens were collected during the daytime, with the exception of some specimens from the Etsura River in 2003 and the Katada-naiko Lake. Following their capture, the specimens were immediately fixed with 10% formalin solution to prevent expulsion and digestion of the stomach contents.

Specimens

In 2001, 166 fish specimens were obtained from the Iso River (Standard body length ± SD, 147.5 ± 22.6 mm; Min–Max, 115–256 mm); 130 fish on April 14, 15 fish on May 13, and 21 fish on June 21. In 2002, 296 fish specimens from the Etsura River were caught (142.1 ± 32.3 mm, 68–256 mm; N = 260, except 36 specimens that were not measured); 27 fish on May 26, 67 fish on May 29, 63 fish on June 2, 19 fish on June 9, 7 fish on July 21, 27 fish on September 22, 64 fish on September 26, and 22 fish on September 29. In 2003, 105 fish specimens were obtained from the Etsura River (146.0 ± 36.2 mm, 95–298 mm); 16 fish, 61 fish, and 28 fish on June 15, June 19, and June 20, respectively. In 2001, 99 fish specimens were obtained from the Kawamichi River (86.6 ± 14.9 mm, 60–124 mm; N = 76, except 23 specimens that were not measured); 14 fish and 43 fish on August 22 and August 16, respectively, 29 fish on September 16, and 13 fish on October 13. From September 2001 to May 2005, 30 fish specimens were obtained from the Katada Lake (135.4 ± 69.9 mm, 59–360 mm). A total of 13 fish specimens were obtained from Pond A (116.6 ± 15.6 mm, 98–150mm); 10 fish and 3 fish on July 31 and August 3, 2002, respectively, and 26 fish specimens were obtained from Pond B (12.7 ± 4.1 mm, 72–254mm); 4, 2, 10, and 2 fish on 8, 10, 11, and 31 of August, respectively and 8 fish on September 4, 2003.

Analysis of stomach contents

To understand prey selectivity in largemouth bass, the stomach contents were examined using the specimens from the above-mentioned sampling sites. Frequencies of the occurrence of the prey were examined after measuring the standard body length. The preys in the stomach were categorized as follows: fishes, shrimps, insects (not including midges), daphnia and midges (DM), and/or others. Fishes were categorized into the following categories: osmerid genus, cyprinid genus, and gobiid genus. Shrimps were categorized into the following categories: river shrimp and/or freshwater prawn (river shrimps) and crayfish.

Egg count in Biwa goby

To clarify the negative effect of the feeding on Biwa goby (indigenous goby fish population of Lake Biwa), 7 specimens that preyed on Biwa goby without digestion from the Iso River were used in this study. Biwa goby was identified by inspecting the maxillary length, interorbital width, and caudal peduncle, and they were measured in terms of their body weight. Each ovary of the preyed goby was picked from the stomach of the largemouth bass and then counted the eggs in the ovary one by one.

DM count

In order to analyze the effect of the feeding on small organisms, the stomach contents of all the 24 specimens that preyed on the DM from Pond B were investigated in detail. The weight of the stomach contents was measured after removing the moisture by using an absorbent paper towel. In order to determine the correlation coefficient between the weight and the number of DM, the prey from one randomly selected specimen of the 24 specimens were counted one by one on graph paper. The numbers of DM in the remainder of the specimens were determined using the correlation coefficient. Daphnia and midges were equated in terms of weight due to the ease of calculations. In the case that the stomach contents included any insects, the weight measurements were repeated to ascertain the DM weight.

Results

Frequency of prey occurrence at each sampling site

Figure 1 shows the frequencies of all specimens, except those with an empty stomach, at each sampling site: Iso River, Etsura River in 2002 and in 2003, Kawamichi River, Katada-naiko Lake, Pond A, and Pond B. The frequencies and prey categories differed among the sampling sites and sampling years. The diet composition also differed significantly.

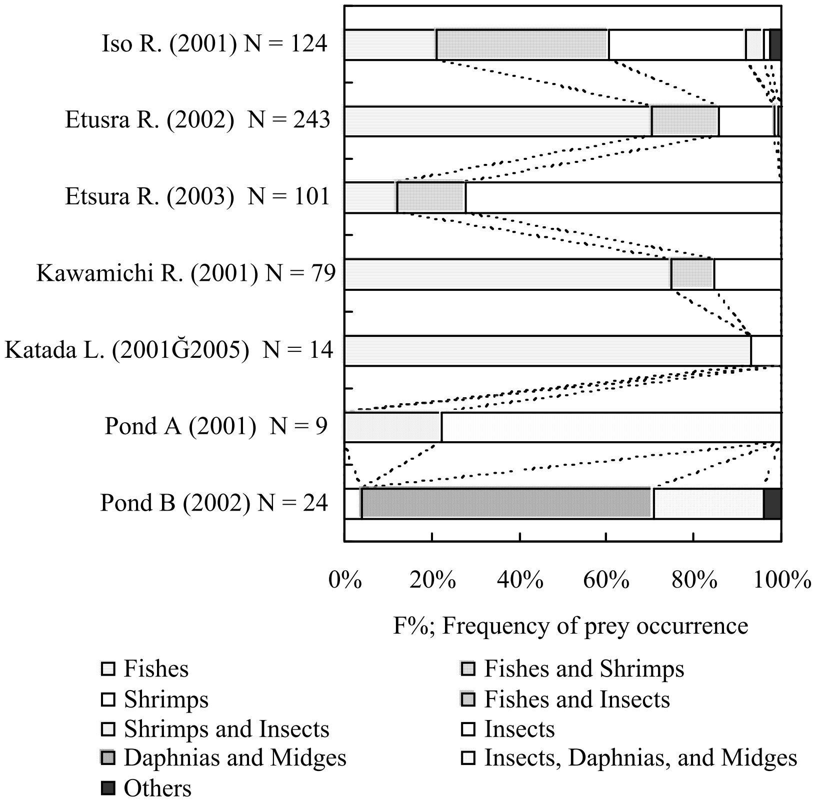

Figure 1. Frequency of prey occurrence of largemouth bass *Micropterus salmoides* in Iso River, Etsura River, Kawamichi River, Katada-naiko lake, and two ponds without mesopredators.

In this study, the main prey components of the largemouth bass were fishes and shrimps, except in the ponds. In the Iso River, stomach contents were obtained from 124 specimens (74.7%). Fishes, shrimps, and insects were obtained from 75 specimens (60.5%), 88 specimens (71.0%), and 7 specimens (5.6%), respectively. In the fishes category, the osmerid genus (Ayu *Plecoglossus altivelis altivelis* and Japanese smelt *Hypomesus nipponensis*), cyprinid genus (stone moroko *Pseudorasbora parva*), and gobiid genus (common freshwater goby *Rhinogobius* sp. and Biwa goby *Gymnogobius isaza*) were obtained in 3, 3, and 38 specimens, respectively. In the shrimp category, river shrimps and crayfish were obtained from 89 and 6 specimens, respectively.

In the Etsura River, stomach contents were obtained from 243 specimens (82.1%) in 2002 and 101 specimens (96.2%) in 2003. The stomach contents showed a significant change between the two years. Fishes were obtained from 209 specimens (86.0%) in 2002 and from 28 specimens (27.7%) in 2003. Shrimps were obtained from 68 specimens (28.0%) in 2002 and from 88 specimens (87.1%) in 2003. Insects were obtained from 3 specimens (1.2%) only in 2002. In the fish category, in 2002, fish of the osmerid genus (Ayu) and gobiid genus (common freshwater goby) were obtained from 53 and 23 specimens, respectively. On the other hand, in 2003, fish of the osmerid genus (Ayu) and gobiid genus (common freshwater goby) were obtained from 6 and 4 specimens, respectively. In the shrimp category, in 2002, river shrimps were obtained from 52 specimens, whereas crayfish were obtained from only 1 specimen. On the other hand, in 2003, river shrimps were obtained from 87 specimens, whereas crayfish were obtained from 3 specimens.

In the Kawamichi River, stomach contents were obtained from 79 specimens (79.8%). Fishes and shrimps were obtained from 67 (84.8%) and 20 specimens (25.3%), respectively. In the fish category, fish of the osmerid genus (Ayu), cyprinid fish (field gudgeon *Gnathopogon elongatus elongatus* and dark chub *Zacco temminckii*), and gobiid genus (common freshwater goby) were obtained from 1, 2, and 26 specimens, respectively. In the shrimp category, river shrimps were obtained from 22 specimens. Although all the specimens were not adults, none of the specimens preyed on insects or DM.

In the Katada-naiko Lake, stomach contents were obtained from 14 specimens (46.7%). Fishes and shrimps were obtained from 13 specimens (92.9%) and 1 specimen (7.1%), respectively. In the fish category, gobiid genus (common freshwater goby) was obtained from 2 specimens, whereas in the shrimp category, 3 crayfish were obtained from 1 specimen. None of the specimens preyed on insects or DM.

In Pond A, stomach contents were obtained from 9 specimens (69.2%). Shrimps and insects were obtained from 2 specimens (22.2%) and 9 specimens (100%), respectively. The shrimps were identified as crayfish. Furthermore, 1 frog was obtained from 1 specimen (11.1%). In Pond B, stomach contents were obtained from 24 specimens (92.3%). Crayfish, insects, and DM were obtained from 1 specimen (4.2%), 7 specimens (29.2%), and 22 specimens (91.7%), respectively. None of the specimens preyed on the fry of largemouth bass in both ponds.

Number of eggs in the abdomen of goby from the stomach of largemouth bass

Table 2 shows the number of eggs in the abdomen of Biwa goby from the stomach of specimens. The standard body length of 7 specimens that were used in the study was 192.7 ± 44.2 mm (min–max: 145–253 mm). The weight of the stomach contents of the specimens was 5.9 ± 2.9 g (min–max: 2.7–10.2 g). The stomach contents consisted of Biwa goby, Japanese smelt, stone moroko, an unidentified fish, and river shrimp (data not shown). All the specimens directly preyed on 1–3 mature females of Biwa goby and indirectly preyed on 377–4339 eggs of Biwa goby.

Table 2. Stomach contents and the number of eggs of Biwa goby in the stomach of largemouth bass *Micropterus salmoides* in the Iso River

No.	Body length (mm)	Weight of stomach contents (g)	Category	No. of Female/s	No. of eggs of the goby
1	157	3.1	Fishes/Shrimps	1	1252
2	159	4.23	Fishes/Shrimps	2	1781
3	176	2.66	Fishes/Shrimps	1	377
4	145	4.66	Fishes	1	1662
5	244	7.92	Fishes/Shrimps	2	2406
6	253	8.42	Fishes/Shrimps	3	2168
7	215	10.24	Fishes/Shrimps	3	4339

Table 3. Stomach contents of largemouth bass *Micropterus salmoides* in Pond B.

No.	Body length (mm)	Weight of stomach contents(g)	Category (Numbers)	%DM by weight to stomach contents
1	98	0.08	Insects (1)/ DM (240)	95.4
2	109	0.22	Insects (10)/DM (510)	79.0
3	100	0.17	Insects (4)/DM (440)	89.4
4	127	0.55	DM (1600)	100
5	88	0.35	DM (1000)	100
6	106	0.61	DM (1770)	100
7	143	0.41	Insects (1)/DM (780)	66.0
8	254	4.60	Shrimps (1)/Insects (3)	0
9	146	0.11	DM (330)	100
10	113	0.60	DM (1750)	100
11	145	0.20	DM (570)	100
12	110	0.24	DM (700)	100
13	86	0.16	DM (470)	100
14	128	0.42	DM (1210)	100
15	92	0.19	DM (550)	100
16	135	0.64	UI (1)/DM (1110)	59.8
17	190	0.52	DM (1500)	100
18	139	0.35	Insect (1)/DM (620)	60.7
19	196	1.49	Others	0
20	95	0.07	DM (190)	100
21	72	0.04	DM (100)	100
22	100	0.13	DM (380)	100
23	112	0.07	DM (190)	100
24	135	0.14	DM (400)	100

UI: Unidentified. Other: rubber goods used in fishing (lure)

Number of DM in the stomach of largemouth bass

Table 3 shows the number and weight of DM from the stomachs of the specimens. DM was dominant in the diets of largemouth bass of Pond B in terms of quantity. The specimen that is selected as a monitor for determining the correlation coefficient between the weight and numbers of DM preyed on 137 individuals of DM per 0.047 g. In other words, the correlation coefficient is 29 individuals per 0.01 g. The standard body length of the 26 specimens used in the study was 126.9 ± 40.9 mm (min–max: 72–254 mm). The weight of the stomach contents of the specimens was 0.5 ± 0.9 g (min–max: 0.01–4.6 g). The stomach contents consisted of DM, strider, backswimmer, crayfish, and rubber goods. Each specimen preyed on approximately 100 to 1770 individuals of DM. Based on the remarkably high value (59.8%–100%) of the frequencies of DM by weight; it is supposed that the specimens mainly preyed on DM.

Discussion

The relationship between predation by largemouth bass and disruption in the distribution of fish fauna that is simultaneous with the increase in largemouth bass population in Japanese inland waters has been receiving increasing attention since the species was introduced into the Japanese waters. In this study, the stomach contents of largemouth bass from 4 points around Lake Biwa and 2 ponds without mesopredators, were examined from 2001 to 2005 in order to understand the feeding habits, their effects on native ecosystem, and the top-down effects. The important findings from this study are as follows: (1) remarkable changes in stomach contents among sampling sites, (2) mass predation of endemic goby in the Iso River, and (3) direct impact on low trophic levels. These points have been discussed below, in the same order:

1. In this study, the largemouth bass mainly preyed on fish and/or shrimps around Lake Biwa. On the other hand, the largemouth bass mainly preyed on insects or DM in the two ponds. This result is consistent with other studies that suggest that largemouth bass that have been introduced are flexible in terms of feeding habits and can change the target of predation in each environment (Yodo & Kimura, 1998). However, the frequency of prey occurrences differed among the sampling sites of Lake Biwa and also differed between sampling years at the same sampling site. In Izu-numa of Miyagi Prefecture, the largemouth bass mainly prey on daphnia up to 20 mm body length (BL) and gradually prey on fishes during further growth (Takahashi, 2002). This report is consistent with the data in the Kawamichi River, whereas it is completely inconsistent with the data from the two ponds.
2. In the Iso River, Biwa goby *Gymnogobius isaza*, which is an endemic goby of Lake Biwa and is categorized as “near threatened” in the Japanese Red Data Book, was largely preyed on by largemouth bass. The mass predation of Biwa goby is the first record, although freshwater common goby was one of the main diets, as reported by the previous studies (Maehata *et al*., 1987; Yodo & Kimura, 1998). Biwa goby is an easily available vulnerable prey for the largemouth bass in the breeding season; this is similar to that observed in the case of other migratory gobiid fishes (Azuma & Motomura, 1998). Furthermore, the specimens preyed on the mature females of the Biwa goby and indirectly preyed on the eggs before they were laid. These results indicate that predation by largemouth bass virtually equals that of the predation of fertilized eggs by egg feeders such as introduced bluegill and leads a great loss in the propagation of Biwa goby.
3. Various field studies have reported the direct and indirect effects of bass occurrence in reducing the diversity of small-bodied fish species; thereby creating more homogenous fish communities and alternating planktonic and benthic communities (Jackson, 2002; Maezono & Miyashita, 2003). In this study, largemouth bass intensively preyed on DM irrespective of the body size in the closed water system without mesopredators. It must be emphasized that in this study, of the 26 collected specimens, 22 specimens (BL greater than 75 mm) preyed on DM, and the frequency of DM in the stomach contents was very high. This result indicates that the predation by largemouth bass primarily affects the low trophic levels and secondarily causes remarkable changes in the ecosystem and water quality. Further studies are necessary for obtaining a complete report on the top-down effect due to the predation of largemouth bass in the study areas. The above-mentioned results suggest that largemouth bass is a euryphagous species; therefore, extermination of this species from nonnative habitats is of utmost importance.

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References

Azuma M. & Motomura Y., 1998. Feeding habits of largemouth bass in a non-native environment: the case of a small lake with bluegill in Japan. *Environ. Biol. Fish*. 52, 379–389.

Environmental Agency, 2004. *Impact on native biotic community and ecosystem by black bass and bluegill and the provision*. Japan Wildlife Research Center, Tokyo (in Japanese).

Jackson D. A., 2002. Ecological effects of *Micropterus* introductions: The dark side of black bass. *American Fisheries Society Symposium*. 31, 221–232.

Kawamura K. & Hosoya K., 1997. Discovery of an endangered cyprinid, *Pseudorasbora pumila* subsp. sensu Nakamura (1969), from the Miya River System, Mie. *Jpn. J. Ichthyol*. 44, 57–60 (in Japanese with English abstract).

Maezono Y. & Miyashita T., 2003. Community-level impacts induced by introduced largemouth bass and bluegill in farm ponds in Japan. *Biol. Conserv*. 109, 111–121.

Maehata M., Kuwahara M., Matsuda M. & Akiyama H., 1987. Feeding habits of largemouth bass *Micropterus salmoides* (Lacepede) in Lake Biwa (South Lake). *The Annual Reports of the Biwako Bunkakan*. 5, 1–14 (in Japanese).

Maehata M., 1993. Dynamics of fishes around Biwako Bunkakan (South Lake). *The Annual Reports of the Biwako Bunkakan*. 11, 43–49 (in Japanese).

Takahashi K., 2002. Impact by largemouth bass on fish community- the case of Izu-numa and Uchi-numa. In: *Invader of rivers and lakes and marshes, black bass-its biology and impacts on ecosystem* (eds the Japanese Society of Japan) pp. 47–59. Kouseisha-kouseikaku, Tokyo (in Japanese).

Yodo T., 2002. *Micropterus salmoides*. In: *Handbook of alien species in Japan* (eds the Ecological Society of Japan) pp. 117. Chijinshokan, Tokyo.

Yodo T. & Kimura S., 1998. Feeding habits of largemouth bass *Micropterus salmoides* in Lakes Shorenji and Nishinoko, Central Japan. *Nippon Suisan Gakkaishi*. 64, 26–38 (in Japanese with English abstract).

The phytoplankton community of an inland lake, Zimbabwe

Ndebele Mzime Regina and Magadza Christopher H. D.

Department of Biological Sciences, University of Zimbabwe. PO MP167, Mount Pleasant, Harare, Zimbabwe
Email: ndebele@science.uz.ac.zw

Abstract

Some aspects of the phytoplankton community namely, species composition, biomass and diversity within Lake Chivero were investigated during the period of March 2003 to April 2003. The highest concentration of phytoplankton was within 0 to 3m with values ranging from 0.8 (2 mg m^{-3} dry weight) to 7.6 mg/l (86 mg m^{-3} dry weight) in fresh weight (FW). The average phytoplankton biomass was 4.9 mg/l FW (31.1 mg m^{-3} DW). There were 25 phytoplankton species that were identified from the phytoplankton samples, which mainly consisted of cyanobacteria, chlorophytes and diatoms. The most abundant phytoplankton species was *Microcystis aeruginosa* (64.3%) followed by *Melosira* sp. (19.3%). The phytoplankton species diversity was 0.92 according to the Shannon-Weiner diversity index. The composition of the phytoplankton community was used as an indication of the trophic status and general health of the lake's ecosystem.

Key words: phytoplankton, diversity, Lake Chivero,

Introduction

The phytoplankton community of Lake Chivero is typical of eutrophic waters (Falconer, 1973; Thornton 1982; Moyo and Worster 1997) with *Microcystis aeruginosa* being the dominant species in the lake (Moyo and Mtetwa 2002). The distribution of phytoplankton is influenced by a number of factors as the organisms accumulate where conditions are most favourable for growth. In Lake Chivero, the phytoplankton density varies with time and basically the phytoplankton community is more evenly distributed at midnight than at midday (Elenbaas 1994) and the organisms tend to accumulate near the dam wall where the currents blow them. The community in the lake has been a good indicator of the health of the ecosystem as indicated by the loss of biodiversity through the years (Robarts 1979, Elenbaas 1994, Mhlanga n.d). Algal blooms have been repeatedly reported in the lake and these have posed problems in terms of the water treatment by clogging the filters during the water purification process. The blooms also pose a serious health threat to both animals and humans when they release toxins into the water (Johansson and Olasson, 1998). They have drastically reduced the aesthetic value of the lake and continue to adversely affect other aquatic organisms especially fish (Moyo 1997). With the exception of a few studies such as that by Magadza (1978, 1980, 1994), who has used plankton in multivariate studies to ascertain the eutrophic status of aquatic environments, only the usual chemical analysis have been used to study aquatic systems and in particular Lake Chivero's health state since the 1980s.

The present study is aimed at make comparative phytoplankton studies using the phytoplankton as indicator organisms in a biological assay of the water quality and trophic status of the lake. Magadza (1994) noted that plankton communities can be useful and finer indicators of the trophic status and also the quality of water as compared to physical and chemical analysis as the organisms are directly affected by the changes in the environment (water) and respond to them in particular, known ways (Magadza, 1978; 1980; 1994).

Materials and methods

Integrated phytoplankton samples were taken on a fortnight basis at five-meter depth using a 5 m hose from three sites (Figure 1). The phytoplankton specimen were preserved in Lugol's solution. These were analysed for species composition and the species were identified using an Utermöhl microscope and phytoplankton guides after Cronberg (1982, 1997), Lund and Lund (1998) and Elenbaas (1994). A three-way analysis of variance (MANOVA) was used to test the variation of the phytoplankton species composition among the sites, the depth profiles and the time of sampling. Physical and chemical parameters namely temperature, conductivity, dissolved oxygen (DO), pH and chloride were measured on site using a Horiba U23 multiprobe meter. Water samples for the determination of total P and total N were collected at each site every two weeks and the nutrient levels were determined in the laboratory by using a HACH ER/ 04-nutrients chemical analysis kit.

A total of 22 gut contents samples of two phytophagous fish species: that is the *Oreochromis niloticus* and *Oreochromis macrochir*, were collected over the six weeks sampling period. The fish were randomly selected from a caught sample of adult (matured) fish from gill nets that were left overnight near site 1 in the lake. The fish were caught once every fortnight during the two months of sampling. A pH meter was used to measure the pH of the fish guts whilst the contents were analysed under the inverted microscope to determine the diet constituents and especially the phytoplankton species ingested by the fish. A CANONICAL correlation method namely, Correspondent Component Analysis (CCA) was used to analyze the data to determine the influence of the different variables measured as well as to produce cluster analysis or associations for all the species and environmental variables.

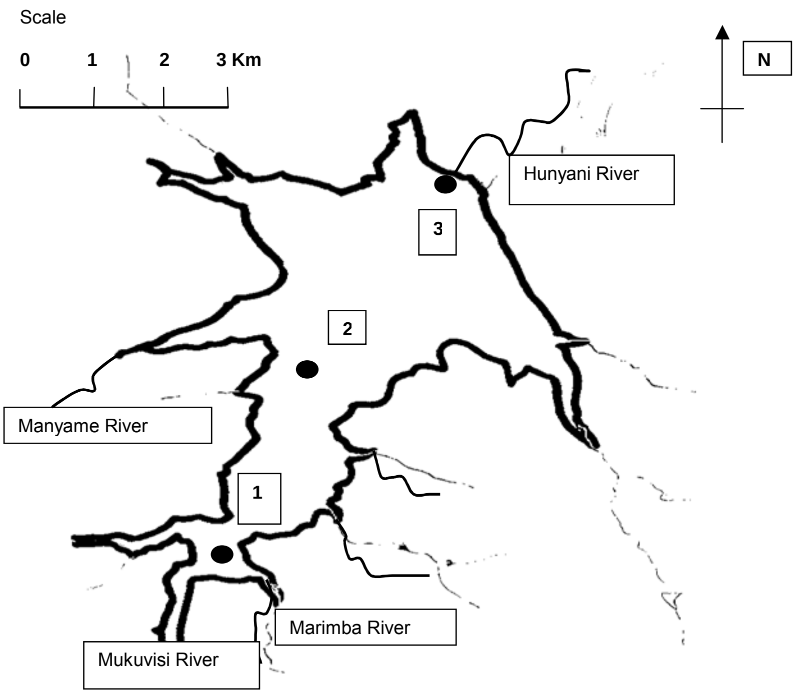

Figure 1: Sketch map showing sampling sites in Lake Chivero and the main tributaries; denotes a sampling station.

After the phytoplankton cell counts, the phytoplankton community's species diversity and distribution was calculated using the Shannon-Weiner diversity index.

Results

Upon microscopic examination of the phytoplankton samples collected from the lake, the following species were identified, counted and scored for frequency at each site (Table 1). The most abundant phytoplankton species was *Microcystis aeruginosa*, which accounted for an average of 64.3% of the phytoplankton sampled, followed by *Melosira* sp., which accounted for 19.3% of the counted species whilst the rest of the species constituted the remaining 16.4%. It was slightly unusual for *Melosira* sp. to co-dominate with *M. aeruginosa* because the time of sampling did not coincide with the winter period when this species dominates in Lake Chivero.

All the sites generally had the same floristic phytoplankton composition but site 2 had the highest number of phytoplankton as well as species identified, followed by site 3 then site 1 (Table 1). Overall, there were 25 species that were identified from the lake samples. A few specimen could not be identified properly under the light microscope and are herein referred to as unidentified green or blue-green algae. It is suspected that such species could be of the diatoms group where many species can only be identified under an electron microscope. The phytoplankton species diversity was low at the time of sampling, being 0.92 according to the Shannon Weiner diversity index formula. This showed that the lake was poor in species diversity and Hill's evenness number (E5) which was calculated to be 2.5 for the lake, reflected that the few phytoplankton species were not evenly distributed in the lake.

From the analysis of the gut contents of *O. macrochir* and *O. niloticus*, it was discovered that the dietary constituents of both fish species were comprised mainly of the phytoplanktoners *Microcystis* sp. and *Melosira* sp. Other common species found within the fish guts included *Cyclotella* and *Pediastrum* sp. (Table 2). Partly digested phytoplankton material was found within the gut of the fish and other gut contents that were observed upon separation of the gut constituents included detritus material, some zooplankton species like *Daphnia* sp., *Bosmina longistrosis*, *Keratella* sp. and *Paramecium* sp.; plant-like material and insect parts. This showed that the fish were not wholly planktivorous, but rather benthic omnivores. An elective index, which compared the food, specifically the phytoplankton species found within the fish guts and the overall phytoplankton in the water and computed this to the fish food preference, was calculated. This index was found to be 37.0% for O. macrochir and 33.3% for O. niloticus. The two fish species, therefore, on average, select these percentages of the phytoplankton species available to them in the lake.

Table 1. Phytoplankton floristic composition and density (per 1ml sample) from March to April 2003 (+ = species detected; - = species undetected in fish guts)

Phytoplankton Species	Fish guts	Site 1	Site 2	Site 3	Tot No.	Average	Freq (%)
Bacillariophyta							
Melosira sp.	+	308	1020	700	2028	676	100
Cyclotella sp.	+	204	182	28	414	138	100
Navicula	-	0	0	6	6	2	1.1
Pinnularia	-	0	12	9	21	7	2.7
Chlorophyta							
Staustram sp.	+	2	22	6	30	10	13.3
Pediastrum	+	35	65	16	116	39	80
Scenedesmus	+	0	3	75	78	26	61.5
Coelastrum	-	2	0	4	6	2	9
Chlorella	-	16	5	0	21	7	17
Volvox	+	3	15	9	27	9	16
Spirogyra	-	5	8	2	15	5	2.7
Chlamydomonas	+	33	6	0	39	13	10
Gonium	-	3	8	5	16	5	2
Cosmarium	-	0	0	4	4	1	1.1
Synedra	-	4	33	0	37	12	5
*Unidentified green algae	-	0	25	2	27	9	3.6
Cyanophyta							
Microcystis	+	1542	2795	3712	8049	2683	100
Lyngbya	-	15	3	0	18	6	4.4
Anabaena	-	23	56	8	87	29	3.6
Cylindrospermum	+	1	11	23	35	12	8
Spirulina	-	5	6	0	11	4	1.1
*Unidentified blue-green	-	0	8	0	8	3	2
Euglenophyta							
Euglena sp.	-	1	13	22	36	12	7.5
Phacus sp.	-	0	0	6	6	2	1.1
Haptophyta							
Rhodomonas	-	1	2	2	5	2	2
TOTAL	9 detected	661	1506	929	25	997	

The pH levels of the gut contents were slightly acidic ranging between 2.7 to 9.9 respectively, with an average pH level of 5.1 for *O. macrochir* and 6.7 for *O. niloticus*. The chi-square goodness of fit test performed reflected that there was a significant difference in the phytoplankton frequency in the fish and in the lake water with probability values of 0.01, 0.04 and 0.006 ($P< 0.05$) for *O. macrochir*, *O. niloticus* and the lake's phytoplankton community respectively (at 0.05% sig. level). The results suggest that the fish are selective feeders.

The correspondence Component Analysis showed that there were two axes that were most influential to the phytoplankton species abundance and distribution in the lake. The axes accounted for 76.1 and 100% of the variation in the species data whilst the rest of the variables were not significant contributors to the phytoplankton community with Eigen values of 0.19 and 1 respectively (Table 3).

Table 2. Phytoplankton species frequency of occurrence in the diet of two fish species.

Type of food (algae)	***O. niloticus* %**	***O. macrochir* %**
Melosira	100	90
Cyclotella	80	50
Microcystis	100	100
Unidentified green algae	20	0
Staustram	50	30
Pediastrum	70	30
Scenedesmus	30	20
Volvox	10	0
Chlamydomonas	30	20
Cylindrospermopsis	60	50
pH average	5.1	6.7
Selective index (%)	37	33.3

The two most influential factors were temperature and DO respectively.

Table 3. Canonical analysis of the environmental variables and phytoplankton community in Lake Chivero

Axes	1	2	3	4	Total inertia
Eigen value	0.19	0	0	0.061	0.256
Species-environment correlations	1.000	0	0	1.000	
Cumulative % variance of species data	76.1	100.0	0	0	0.256

Cluster analysis using grouped variables produced dendrograms (figs 2 and 3). The first (Figure 2) dendrogram shows that most of the phytoplankton species were closely associated with the physio-chemical parameters and were clustered according to genera with the exception of *Spirulina* sp., *Phacus* sp. and the unidentified green algae. The second dendrogram showed that there were five clusters in the species-environmental data. The clusters were (in order of high association):

1. Microcystin, pH, DO, depth, total P, total N, transparency and the N: P ratio (5);
2. Phytoplankton species at site 3, biomass and temperature (8);
3. Clusters 1 and 2 combined (25);
4. Chloride, phytoplankton at site 1 and at site 2 and the time variations (107);
5. Conductivity and the rest of the variables and species (248).

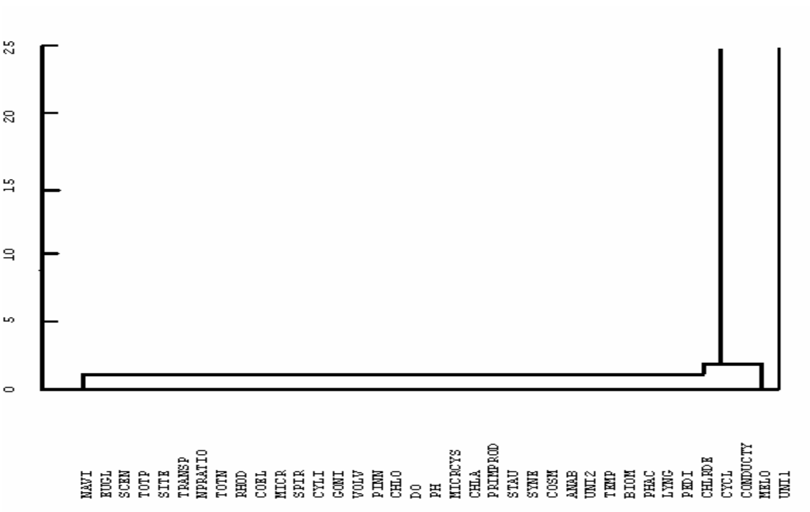

Figure 2. Dendrogram using average linkages between the phytoplankton community and some environmental variables.

The first cluster consisted mainly of the physio-chemical parameters whilst cluster 2 was comprised of biological data and cluster 3 was a combination of cluster 1 and 2, which was the phytoplankton at site 3, the temperature and biomass. Cluster 4 was made up of the phytoplankton species at site 2 and 3, which were associated with chloride and sampling times variations.

Conductivity (cluster 5) was the most unrelated variable to all the species and the other variables in the lake (Euclidean distance: 248 on Figure 3). There was a sub-cluster within cluster 4, which is labeled as cluster 6 in Figure 3. This cluster was made up of the phytoplankton at site 1 and 2. The CCA plot (Figure 4) also showed that conductivity was least related to the other variables (which were all closely associated) whilst the phytoplankton at site 3 was closely related to temperature.

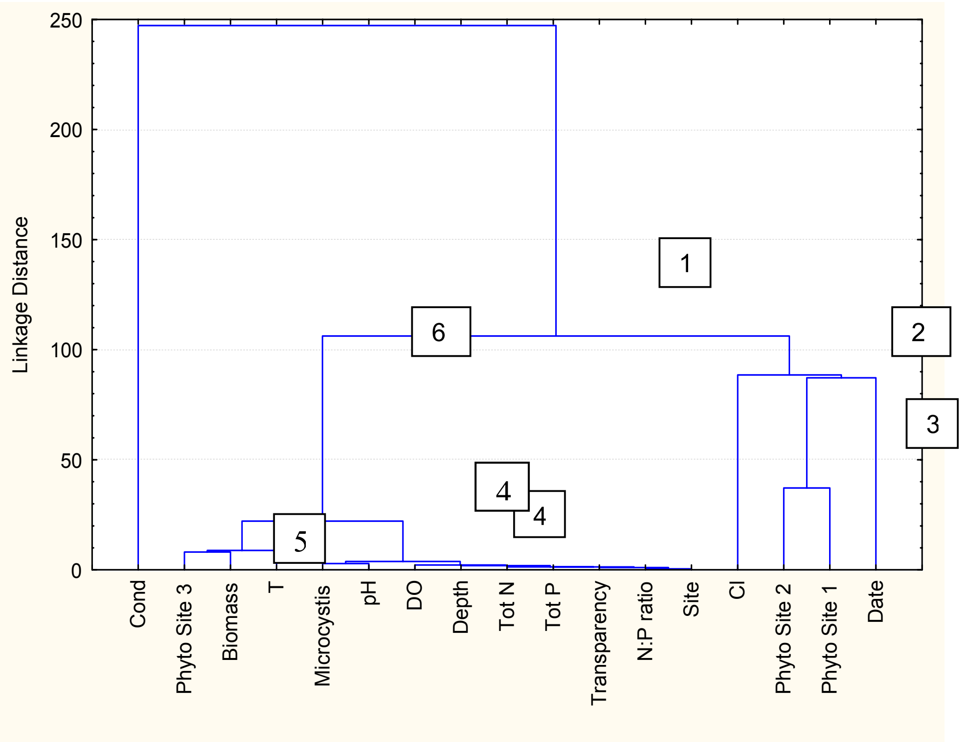

Figure 3. Dendrogram using Euclidean distances between the species and some environmental variables.

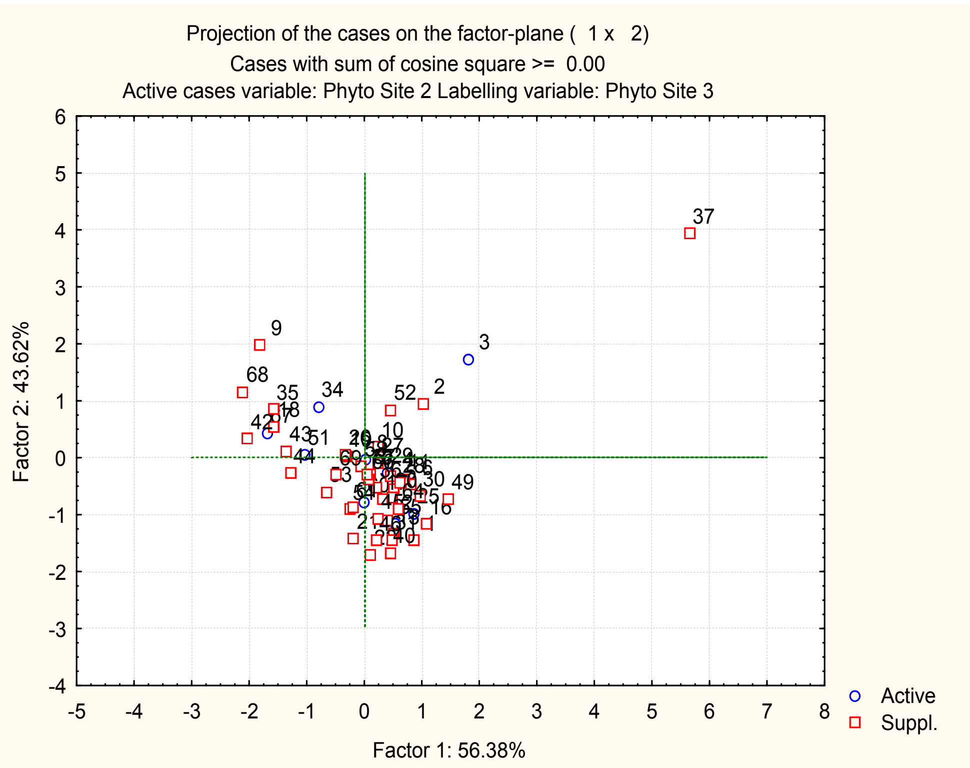

Figure 4. Scattergram of the phytoplankton species and environmental variables.

Discussion

The lake's enrichment not only supported higher phytoplankton densities, but also adversely affected the biodiversity in the reservoir. This was clearly indicated by the low phytoplankton diversity of the lake (0.92) in the present study. Only one cyanobacterial genera dominated the phytoplankton community, namely *Microcystis* sp. and another co-dominated, that is, the *Melosira* group, whilst the rest of the species' contribution was low. The low diversity in the species composition recorded was consistent with results found by other workers (Falconer 1973; Robarts, 1979; and Moyo, 1997) in the past but the species numbers in the present study were much lower. This could be attributed to the short time of sampling and the time or season as the species vary seasonally. The cyanobacteria, *Microcystis* sp. was expected to dominate the phytoplankton community during the sampling period (summer) as was found by Falconer (1973), Marshall and Falconer (1973), Robarts (1979), Magadza (1994) and the results were consistent with this expectation. However, the co-dominance of *Melosira*, a winter-dominating species, could be explained by the cold weather experienced during the time of sampling due to the rains (Zimbabwe Met. Report 2003), and since phytoplankton are short-lived, the cold, winter-like weather may have induced the growth of this particular species.

Although *Microcystis* sp. is a cosmopolitan phytoplankton species that adapts to different environments, its dominance in Lake Chivero can be an indicator of the hyper-eutrophic state of the lake. The source of this high eutrophic state arises from pollution exerted through industrial and sewage effluent and also from non-point sources such as agricultural run-off and urban wastes. The eutrophic characteristics continue to prevail despite the Lake's recovery in the late 1970s to early 1980s as the lake continues to deteriorate in water quality (Magadza, 2003).

The chi-square test showed that both *O. niloticus* and *O. macrochir* did not have the same proportion of phytoplankton in the guts as in the water from where they feed. One disadvantage of this test is the fact that it is based on frequency of occurrence of the food type in the fish guts, which does not accurately reflect the abundance of the phytoplankton and the fish selectivity. The selectivity index recorded varied with the results obtained by Minshull (1978), who examined the stomach contents of *O. macrochir* juveniles and found that they were unselective with very little difference between the phytoplankton in the stomach and the lake water. Such a difference may be attributed to the fact that the sampled fish were adults, which are highly specialised feeders (Marshall, 1982).

It is important to note that the fish sampled did not only feed around the sites where the phytoplankton samples were taken and this could have contributed to the differences in the phytoplankton in the fish guts and in the lake water. However, the presence of detritus material, zooplankton species and benthic algae (diatoms) like *Cyclotella* sp. in the fish guts suggested that the fish are not wholly phytophagous as they feed on both phytoplankton and epiphytic, benthic algal species. These results were consistent with those of Munro (1967), Moriarty (1973) and Minshull (1978), who found that O. niloticus and O. macrochir fed extensively on blue-green algae as well as on benthos material. It has been suggested that the microphytophagous nature of the two fish species has contributed to the fish's success in the lake as other fish species have failed to utilise this phytoplankton species as a food source (Marshall, 1982). The abundance of *Microcystis* sp. in the fish gut contents was also reported in studies by Marshall (1982) and Moyo (1997).

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References

Cronberg G., 1982. Phytoplankton changes in Lake Trummen Induced by restoration. Long-term whole-lake studies and food web experiments. *Folia Limnoligica Scandinavica* 18, 1-119.

Cronberg G., 1997. Phytoplankton in Lake Kariba. In: Moreau J. ed.. *Advances in the ecology of Lake Kariba.* University of Zimbabwe Publications, Harare, Zimbabwe.

Elenbaas P. F. M., 1994. *Field and lab work on ecology in Zimbabwe.* Centre for Development Cooperation Services, Vrije University, Amsterdam, Netherlands.

Falconer A.C., 1973. The phytoplankton ecology of Lake Mcllwaine, Rhodesia. M. Phil. Thesis, University of London.

Johansson S. and Olsson M., 1998. Investigations of toxic algal blooms in a drinking water reservoir of Harare, Zimbabwe. MSc Thesis, University of Lund.

Lund H. C. and Lund J.W., 1998. *Freshwater algae-their microscopic world explored.* Biopress Ltd, Bristol.

Magadza C. H. D., 1978. Phytoplankton in six hydroelectric lakes on the Waikato River, New Zealand, 1970-1972. *New Zealand Journal of Marine and Freshwater Research* 12, 29-40.

Magadza C. H. D., 1980. Comparative seasonal estimates of primary productivity in the Waikato River lakes during summer, autumn, and winter. *New Zealand Journal of Marine and Freshwater Research* 14:71-77.

Magadza C. H. D., 1994. Evaluation of eutrophication control in Lake Chivero, Zimbabwe, by multivariate analysis of zooplankton. *Hydrobiologia* 272, 277-292.

Magadza C. H. D., 2003. Lake Chivero: A management case study. *Lakes & Reservoirs: Research and Management* 8, 69–81

Marshall B. E. and Falconer A. C. (1973) Physio-chemical aspects of Lake Mcllwaine (Rhodesia), a eutrophic tropical impoundment. *Hydrobiologia* 42, 45-62.

Marshall B. E., 1982. The benthic fauna of Lake McIlwaine. In: *Lake McIlwaine. The Eutrophication and Recovery of a Tropical Man-made Lake.* (eds Thornton, J.A. and Nduku W.K.) Dr. W. Junk Publishers, The Hague, The Netherlands.

Minshull J. E., 1978. *A preliminary investigation of the Sarotherodon macrochir (*Boulenger*) on a shallow shoreline in Lake McIlwaine. Rhodesia*. MSc Thesis, University of Rhodesia.

Moriarty D.J.W., 1973. The physiology of digestion of blue-green algae in the cichlid fish, *Tilapia nilotica*. *London Journal of Zoology* 171, 25-39.

Moyo N. A. G., 1997. The causes of massive fish deaths in Lake Chivero. In: *Lake Chivero A Polluted Lake* (ed Moyo N.A.G.). University of Zimbabwe Publications, Harare, Zimbabwe.

Moyo N.A.G. and Mtetwa S., 2002. Water quality and pollution control. In: *Defining and mainstreaming environmental sustainability in water resources management in southern Africa.* (eds Hirji R., Johnson P., Maro P., Matiza-Chiuta T.). SADC Technical report.

Moyo N.A.G. and Worster K., 1997. The effects of organic pollution on the Mukuvisi River, Harare, Zimbabwe. In: Moyo N.A.G. (ed). *Lake Chivero A Polluted Lake*. University of Zimbabwe Publications, Harare, Zimbabwe.

Munro J.L., 1966. A Limnological survey of lake McIlwaine, Rhodesia. *Hydrobiologie,* 28:181-308.

Robarts R. D., 1979. Underwater light penetration, chlorophyll a and primary production in a tropical African lake (Lake McIlwaine, Rhodesia). *Archive of Hydrobiology* 86, 423-444.

Thornton J. A., 1982. The algal community. In: Thornton J.A. and Nduku W. K. (eds.). *Lake McIlwaine. The Eutrophication and Recovery of a Tropical Man-made Lake*. Dr. W. Junk Publishers, The Hague, The Netherlands

Zimbabwe Meteorological Report., 2003. Department of Meteorology, Harare region, Zimbabwe.

Dominance of introduced Nile tilapia, *Oreochromis niloticus* (L.) in Lake Victoria: A case of changing biology and ecosystem

Njiru M.[1], J.E. Ojuok[2], A. Getabu[2], M. Muchiri[1], I. G. Cowx[3] and J.B. Okeyo-Owuor[4]

[1] Fisheries Department, Moi University, P. O. Box 1125, Eldoret, Kenya. fax +254 56 530045, [2] Kenya Marine and Fisheries Research Institute (KMFRI), P.O. Box 1881, Kisumu, Kenya

[3] University of Hull International Fisheries Institute, Hull, HU6 7RX, UK

[4] School of Environmental Studies, Moi University, P.O. Box 3900, Eldoret, Kenya

[1]Email-rmnjiru2002@yahoo.com

Abstract

Nile tilapia, *Oreochromis niloticus* together with other tilapiines of *Oreochromis. leucosticus, Tilapia. zillii* and *Sarotherodon melanopleudra* (=*T. rendalii*) were introduced into Lake Victoria between 1951 and 1962 to boost the then declining fishery. Only *O. niloticus* was able to establish leading further to reduction in endemic tilapiines of *Oreochromis variabilis* and *Oreochromis esculentus*. *O. niloticus* currently forms the third commercially important species after introduced Nile perch, *Lates niloticus* and a native cyprinid *Rastrineobola argentea*, whereas other tilapias are extinct or are occasionally caught in the lake.

Information was collected from by bottom trawling and from published literature to ascertain possible factors leading to dominance of *O. niloticus*. Compared to other tilapiines, Nile tilapia is widely distributed, feeds on a variety of food items, grows to larger sizes, is highly fecund and can survive in a wide range of physical chemical parameters in the lake. These attributes could probably be the reason for its dominance over the tilapiine groups in the lake. Studies further show that ecology and biology of the *O. niloticus* has changed probably in response to changes occurring in the ecosystem. Management measures to sustain the fishery which include reduced fishing pressure, use of legal fishing methods, and control of environmental degradation have been discussed

Key Words: Overfishing, management, pollution; tilapia; ecology

Running title: Dominance of Nile tilapia in Lake Victoria

Introduction

Nile tilapia, *Oreochromis niloticus* (L) is presently widely distributed in many parts of the word (Trewavas, 1983). It occurs together with *Sarotherodon galilaeus* (L) and *Tilapia zillii* (Gervais) throughout much of its natural range in Palestine, the Nile, and across West Africa, and Lakes Turkana, Edward, George, Tanganyika and Albert in East Africa (Lowe McConnell, 1958; Trewavas, 1983). The herbivorous tilapiine, the Nile tilapia, *Oreochromis niloticus* (L.) was introduced in Lake Victoria in 1950s and 1960s to boost the then declining fishery (Welcomme 1967; Ogutu-Ohwayo 1990a). Currently Nile tilapia is the most commercially important tilapiine in Lake Victoria (Cowx *et al.* 2003; Njiru *et al*., 2005). This is in sharp contrast with the native species of 1950s and 1960s of *Oreochromis esculentus* (Graham) and *Oreochromis variabilis* (Boulenger) Today *O. niloticus* constitutes the third most important fishery in Lake Victoria, after Nile perch, *Lates niloticus* (L.) and a native cyprinind, *Rastrineobola argentea* (Pellegrin). Increase in *O. niloticus* is attributed to over fishing of endemic tilapiines thus reducing competition, while swamps clearance could have increased it spawning areas (Balirwa, 1998). Nile tilapia can also survive a wide range of pH, resists low levels of dissolved oxygen and feeds on a variety of food items (Balirwa, 1998; Njiru *et al*., 2004). The paper evaluates possible factors which have led to the dominance of introduced Nile tilapia in Lake Victoria.

Introduction of tilapiines in Lake Victoria

The first introduction of *O. niloticus* into Lake Victoria probably occurred in the early 1950s (Lowe-McConnell, 1958; Welcomme, 1966, 1967; Balirwa, 1992). The stocking in Kenya and Tanzania waters were between 1956 and 1958 with fry from Kajjansi fish ponds in Uganda. In Uganda, a massive introduction, involving tens of thousands of fry whose origin is not well documented stocking were carried out from Entebbe between 1961 and 1962. Apart from *O. niloticus,* other tilapiines introduced into Lake Victoria between 1951 and 1962 were *Oreochromis leucosticus, T. zillii and Sarotherodon melanopleudra* (=*T. rendalii*)- the last originating from Zambia (Welcomme 1966, 1967). The picture of introductions is somewhat confused as Trewavas (1983) suggests that *O. leucosticus* came originally, by accident, with *T. zillii* from Lake Albert. She also points that the lake was stocked with Lake Victoria subspecies of *O. niloticus* (= *O. niloticus vulcani*). The aim of introduction was to increase the declining native tilapiines and return the use of 5'' (127 mm) mesh gill nets because *O. niloticus* grows to a large size. Nile tilapia started appearing in commercial catches in 1960 and the objective on introduction was not realised even by 1963 because the species constituted only 1 per cent of the commercial catch (Welcomme, 1966; Balirwa, 1992). However, from 1965, it started featuring prominently in the commercial and encouraged a return of 5'' gillnet.

Methodology

Fish samples were collected monthly by bottom trawling (head rope 22.6 m, codend mesh size 24.5 mm) from the Kenyan waters of Lake Victoria (Figure 1) between June 1998 and December 2000. The sample sites were defined using a Global Positioning System (GPS), and depth (m) estimated by an echo sounder. Stations 1, 2, 3, 6, 10 and 11 were < 5 m deep, 4, 7 and 12 were between 5 and 10 m deep, stations 5 and 8 had depths > 10 m and station 9 was 20 m deep. Fish caught were weighed and biomass was estimated using the swept area method (Sparre and Venema ,1998). Ovaries used to estimate fecundity was preserved in Gilson's fluid and fecundity was estimated from total counts of ova in the most advanced state of development.

The samples for diet analysis were collected in June 1998 to December 2000, in July 2004, March 2005, June and September 2005 by bottom trawling. Fish stomachs preserved in 4% formalin. In the laboratory, the stomachs were dissected material constituting algae were thoroughly mixed with a small volume of water. A sub sample was then taken using a teat pipette and placed in a Sedgwick rafter cell, which carries a volume of 1ml. The food items were enumerated under a compound microscope (1000x) and the contents identified to the lowest possible taxonomic level using plankton keys. Selectivity of food items by *O. niloticus* was estimated using the Ivlev's index of electivity (*E*) (Strauss, 1979):

$(E) = (ri\text{-}pi)/\ (ri + pi)$,

where, *ri* = proportion of *i* food item in fish stomach, *pi* = proportion of food item *i* in the water sample. Published and unpublished data on tilapiines especially in Lake Victoria were also gathered and used in this paper.

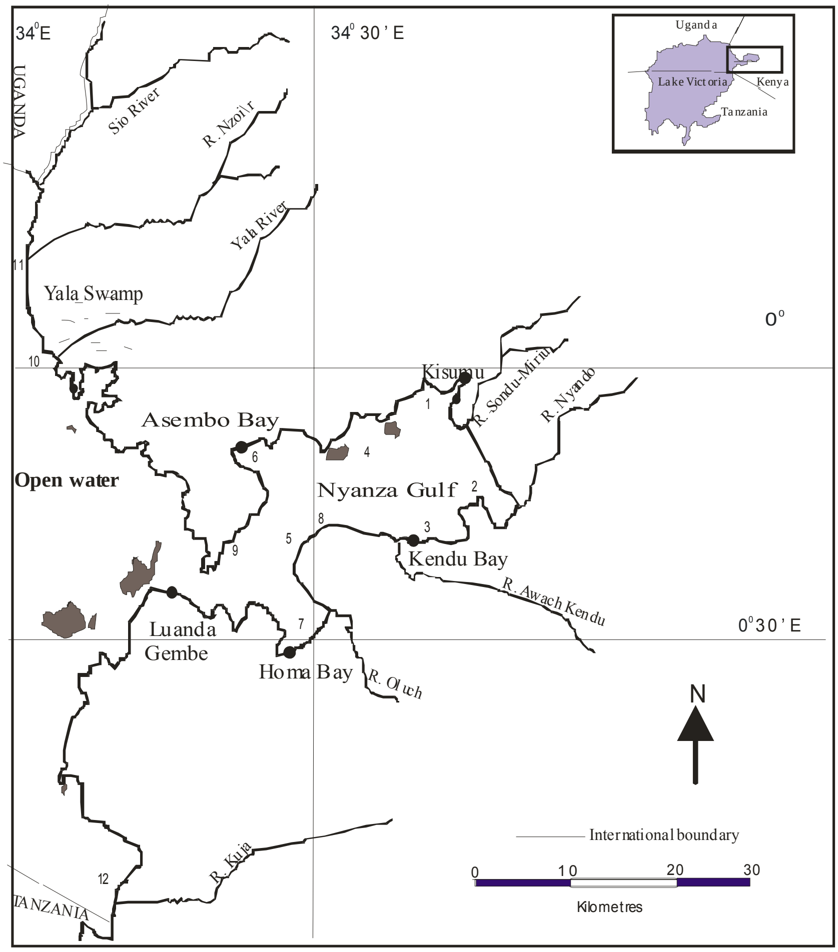

Figure 1. Map of Lake Victoria, Kenya showing the sampling sites.

Catch trends

The mean (±SD) catches of *O. niloticus* ranged from 6.99 ± 2.16 to 40.6 ± 21.9 Kgha^{-1} at station 8 (10 m) and 9 (20 m) respectively (Figure 2a). In 1970s *O. niloticus* was caught in areas less than 10 m and catches of 5 kg per 30 minutes were very rare in 1970s (Kudhogania and Cordone, 1974). In this survey tilapia was caught up to 20 m deep and up to 100 kg per 30 minutes haul was common. Comparison with previous bottom trawl surveys reveal that in the recent past catches of *O. niloticus* are on the rise (Figure 2b). Higher catches and movement of Nile tilapia into deeper waters could be due to changes in fish compositions in the lake. More than 60 % species of cichlids have drastically reduced or become extinct in Lake Victoria (Ogutu-Ohwayo, 1990a). Therefore, Nile tilapia experiences less competition for resources and could be expanding it ecological niche to occupy “empty” niches left by these cichlids.(Ogutu-Ohwayo 1990a). The spread of tilapia could also have been enhanced by the declining stocks of predatory *L. niloticus* (Figure 2b) Catches of perch have been on the declined since the late 1990s due to overexploitation (Njiru *et al*., 2002; Cowx *et al*., 2003). This means predation pressure especially on the young tilapia is reducing.

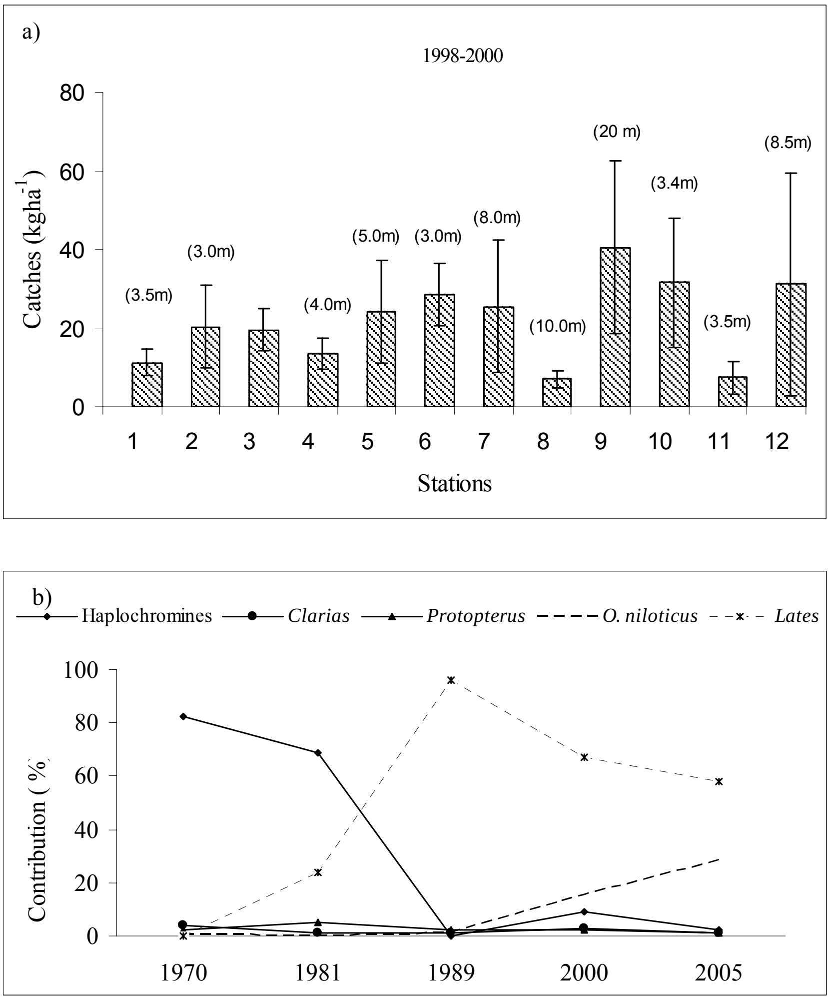

Figure 2. Bottom trawl catches in Lake Victoria, Kenya. a) catch per station in 1998-2000, b) contribution (%) over time (Adapted from Njiru *et al*. 2005).

Overfishing of native tilapiines

Increase in *O. niloticus* is attributed to over fishing of endemic tilapiines thus reducing competition. The history of the fishing industry on Lake Victoria up to mid-sixties is given by Mann (1969). Even in absence of catch record, there was virtually no impact on the stocks by subsistence requirements up to the 1900s when the gill nets were introduced as a new fishing technique. Lake Victoria then contained large stocks of *O. esculentus*. With the introduction of gill net, coupled with the growth of urban centres and communications around the lake, the fishing industry assumed a commercial role. By 1916, the catch rate per net per night (c.p.n) ranged from 25-100 *O. esculentus* in a 5-inch (127 mm) net of about 45 meters (EAFFRO 1955/1956). The gill nets become so popular that by the mid-1920s the number of tilapia had declined to 5 c.p.n. Despite measures to reduce the trend the c.p.n dropped from 3.1 in 1933-37 to 1.9 by 1945 and 1.2 in 1955. As *O. esculentus* was marketed by weight and not by size, this encouraged fishermen to fish with undersized gill nets. From 1956, gill nets of 4.5 inches (114 mm) mesh appeared (Balirwa, 1992). This resulted in increase in profits to fishermen and catches of *O. esculentus*. The smaller meshed nets of 3” (76 mm) became wide spear in the late 1950s and early 1960s. They mainly captured stocks of *O. variabilis* that grew to a smaller size (EAFFRO 1955/56). With increased fishing effort, *O. variabilis* declined. The introduction of *T. zillii* and *O. leucostictus* made a temporarly impact but was also overfished by the early sixties (Balirwa 1992).

Diet

Nile tilapia has more diversified diet and can be described as an opportunistic herbivore (Balirwa 1998; Njiru *et al*., 2004). The species previously described as herbivorous (Trewavas 1983), has a very diverse diet that includes insects, algae, fish, molluscs and detritus (Figure 3). The endemic tilapia, *O. esculentus* feeds almost entirely on diatoms (Welcomme 1967; Opiyo 1994) and *O. variabilis* ingests mainly on phytoplankton (Trewavas 1983; Fishbase 2005), *T. zillii* ingests mostly macrophytes (Fishbase 2005). There has been a shift from predominance of diatoms and green algae in the 1960s to the present domination of blue greens (Lung'ayia *et al*., 2000). Blue green algae offer poor quality food and have toxic groups (Lung'ayia *et al* 2000). The increase in blue green algae is due to increase eutrophication in the lake attributed to nutrient rich sewage effluents and agricultural runoffs (Lung'ayia *et al*., 2000). The replacement of diatoms is attributed to decline in silicon in the lake necessary for cell wall formation (Lung'ayia *et al*., 2000). Studies have further shown that Nile tilapia is able to digest blue green algae (Moriarty and Moriarty 1973). This study shown that when *O. niloticus* is feeding on algae it is able to actively select for the rare diatoms and green algae even when blue greens are more abundant species(Figure 4). The diversified feeding mode of *O. niloticus* probably gave the species higher survival rate in the changing Lake Victoria ecosystem.

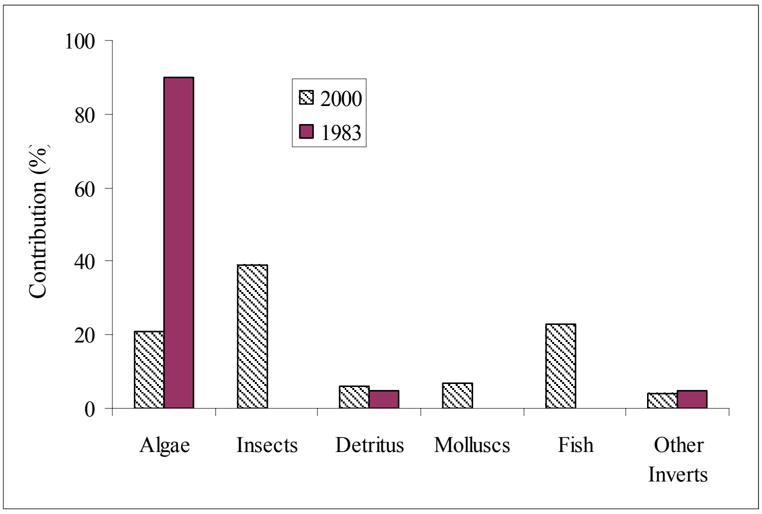

Figure 3. Contribution of the major food items ingested by *O. niloticus.* Source: 1983(Trewavas 1983), 2000 (Njiru *et al.* 2005).

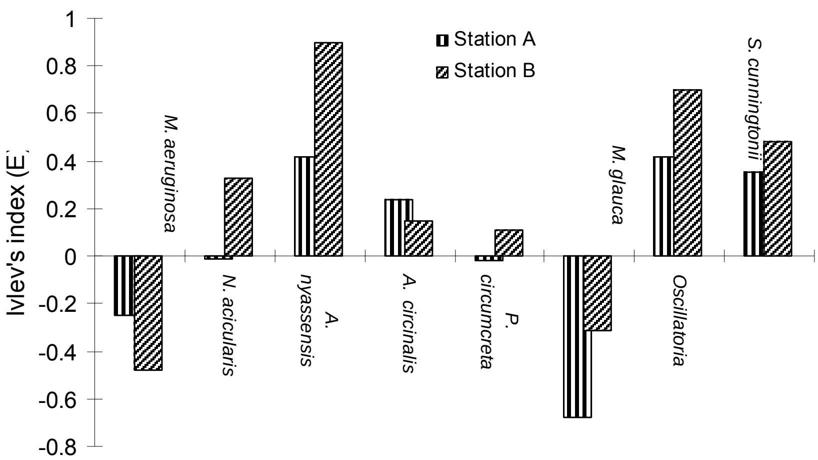

Figure 4. Selectivity of algae species by O. niloticus in Lake Victoria, Kenya. (M= Microcystis aeruginosa, M = Merismopedia, N=Nitzchia, A=Anabena, S= Synedra, P=Planktolynbya).

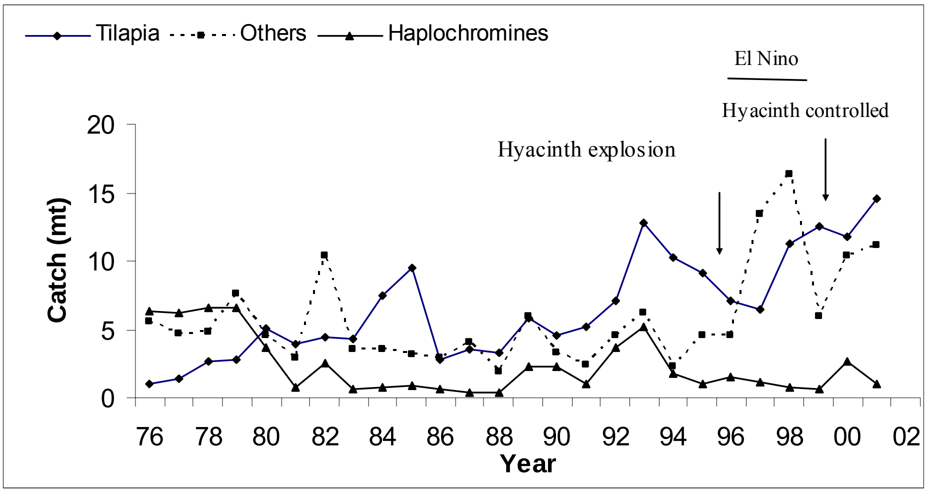

Figure 5. Annual catches of haplochromines, tilapia and other species mainly (Clarias, *Protopterus*) in Lake Victoria, Kenya.

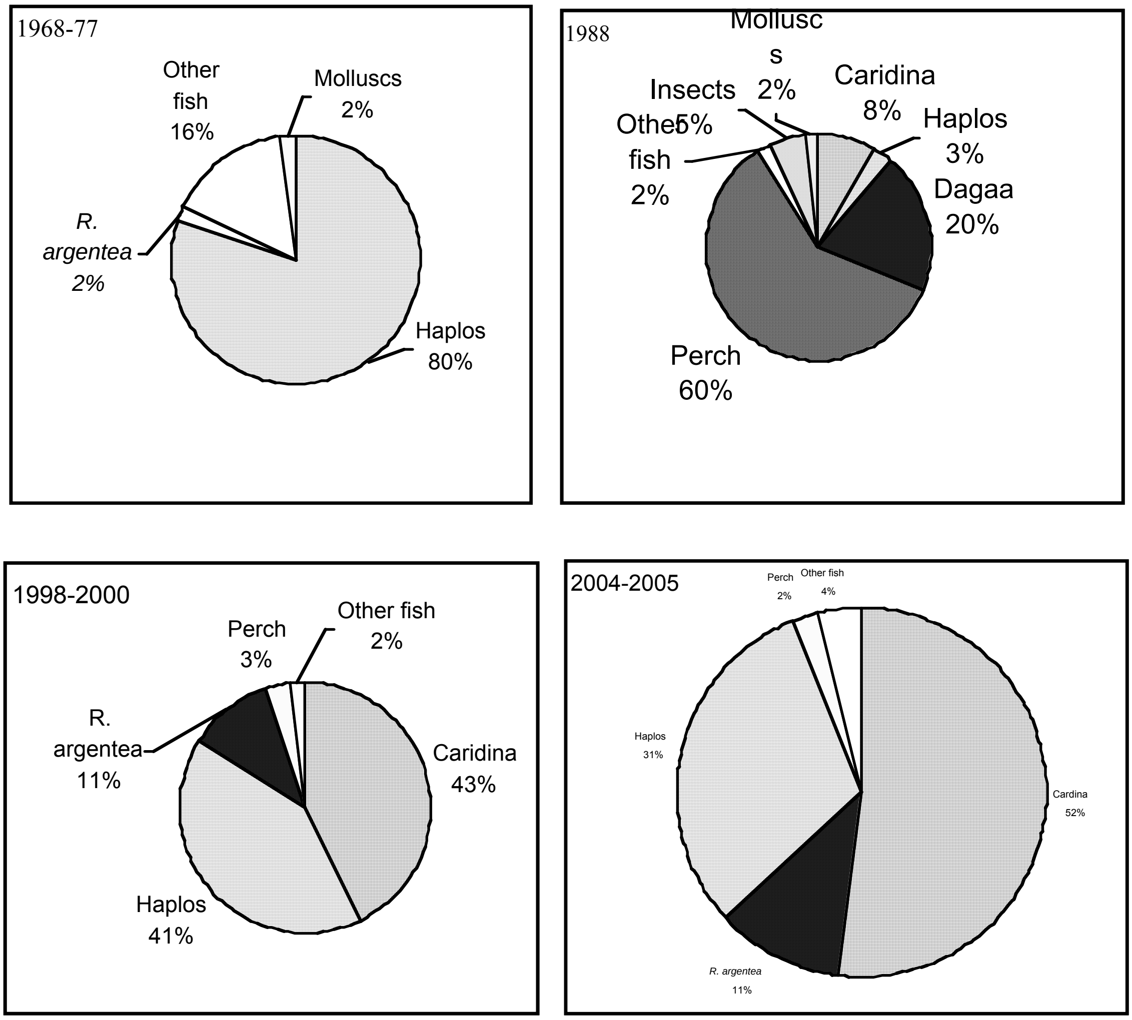

Figure 6. Diet of *O. niloticus* in Lake Victoria. Note *O. niloticus* does not appear as major diet even when catches are high. Source: 1968-77 (Ogutu-Ohwayo, 1990a), 1988 (Ogari and Dadzie ,1988).

Clearance of swamps

The near shore of Lake Victoria has been influenced by human activity in the drainage basin such as clearance of swamps and marginal vegetation and nutrients transport to the lake via run-offs (Lung'ayia *et al*., 2000). The effect of swamp clearance could have increased the spawning area for *O. niloticus* (Welcomme, 1966, 1967; Balirwa, 1998). The disappearance of water lilies and other aquatic weeds reduced the nursery grounds for *O. esculentus* and feeding niche of higher plants material (*Potamogeton* and *Ceretophylum*) which were fed upon by *T. zillii* (Welcomme 1967).

Water hyacinth

During infestation of Lake Victoria by water hyacinth, *O. niloticus* catches increased dramatically (Figure 5). Water hyacinth provided feeding and breeding grounds *O. niloticus* (Balirwa, 1998; Njiru *et al*., 2004). Diet of Nile tilapia by then consisted of mostly insects which were found in association with the hyacinth (Njiru *et al*., 2004). Infestation of the lake by the hyacinth also led to a drastic reduction in beach seining, a common fishing method in shallow waters, allowing the fish time to reproduce and grow. The declining trend thereafter was attributed to reduction in hyacinth and over fishing by fishers who targeted the species. Under the hyacinth water oxygen levels could go as low as 0.01 mgl^{-1}. Nile tilapia which has a higher tolerance to low oxygen concentrations (Kolding, 1993) excluded the predator Nile perch whose critical dissolved oxygen is 5 mgl^{-1} (Kaufman, 1992). Thus, localised deoxygenation could have played a role in tilapia increase by occupying areas that Nile perch could not.

Massive stocking and El Niño

The massive 1960s stocking off Entebbe followed by rising lake levels could have enhanced spread of *O. niloticus* (Balirwa, 1992). Under favourable conditions tilapiines may spawn at frequent intervals throughout their productive life (Lowe-McConnell, 1955; Balirwa, 1992, 1998). Heavy rains of 1960-1963 led to flooding of marginal areas of the lake and creation of new beaches and lagoons, offering more breeding and feeding areas (Balirwa, 1998). Recent increase in catches in 1998-1999 could possibly be linked to the effect of El Niño rains (Oct. 1997-March 1998) (Figure 5**)**. The rains increased lake water volume, flooding adjacent lands and increased feeding and breeding areas, thus increasing recruitment.

Fecundity

Nile tilapia in Lake Victoria is more fecund and shows a well-defined reproductive strategy, which has probably contributed to its success in the lake. Lowe-McConnell (1955) reports that *O. variabilis* produces 323-547, *O. variabilis,* 324-1627, *O. leucostictus*, 99-950, *O. niloticus* 340-3706 and *T. zillii* 1000-7061 eggs in Lake Victoria. Recent studies show the fecundity of *O. niloticus* has increased from 864-6316 eggs for fish of 28-56 cm TL (Lung' ayia, 1994), to 905- 7619 eggs for fish of 28-51 cm TL (Njiru *et al.,* in press). Adult *O. niloticus* also competed with *O. variabilis* for breeding grounds. High survival rate of the *O. niloticus* could be probably be due to its eggs and young brooding behaviour. The lack of establishment by *T. zillii* and *T. rendalii* could be due to the fact that they lay and raises their young on the bottom substratum of the lake, and their eggs could suffered high predation from then abundant haplochromines (Lowe-McConnell, 1955).

Growth and predation

Among all the tilapias Nile tilapia grows to the largest length and size (Fishbase 2005). The maximum size attained by Nile tilapia in Lake Victoria is 60 cm TL (Njiru 2003). Studies from Lake Naivasha, Kenya found *O. leucostictus* to attain a maximum length of 37 cm TL, *T. zillii* 30 cm TL (Ojuok personal communication). Maximum sizes obtained for the tilapiines are 4324g for *O. niloticus,* 2200g for *O. esculentus* and 300 g for *T. zillii* (Fishbase, 2005). Growing to bigger size could probably have reduced predation of *O. niloticus* by Nile perch. Following the establishment of Nile perch there was a drastic reduction in native icthyofaunal diversity (Ogutu-Ohwayo 1990a) and this could have promoted the spread of *O. niloticus.* Nile perch diet studies reveal very little *O. niloticus* is ingested by Nile perch (Figure 6). Both introduced species are able to coexist may be due to ecological separation which has evolved over long time in their native habitats in the Nile, and lakes Turkana, Edward, George, Tanganyika and Albert in East Africa.

Hybridisation

Tilapia are known to interbreed both under natural and artificial conditions (Lowe-McConnell ,1958; Welcomme, 1967; Trewavas, 1983). Hybrids of *O. variabilis x O. niloticus,* and *O. esculentus x O. niloticus* hybrized under experimental conditions (Lowe-McConnell, 1958). Preliminary studies in northern waters found such hybrids (Balirwa, 1992). A characteristic feature of all hybrids is the dominance of an *O. niloticus* morphological features. Therefore for all practical purposes, the native tilapias of *O. variabilis* and *O. esuclentus* have been swamped up and the common tilapia in Lake Victoria is some form of *O. niloticus* (Balirwa, 1992; 1998).

Conclusion

Several factors have been advanced on why *O. niloticus* has replaced and dominated the tilapia fishery of Lake Victoria. These factors include a wide distributed range, utilisation of variety of food items, big sizes and reduced predation, higher fecund and survival in a wide range of physical chemical parameters in the lake.

However, recent studies have shown that the dominant tilapia is also under threat from fishing pressure (Njiru 2003; Njiru *et al* in press). The fish matures earlier (ripe male 21.0 cm TL, female 22.7 cm TL) than in 1990s when it was reported to mature at 35 cm TL (Getabu, 1992). Fecundity, growth (K), fishing mortality (F), exploitation rate (E) are increasing whereas asymptotic weight (W_∞) and length (L_∞) is declining (Table 1).

Table 1. Growth parameters (K yr^{-1}, W_∞ g, L_∞ cm), mortality (Z, M, F yr^{-1}) and exploitation (E) estimates of *O. niloticus* from Victoria (Source: Njiru 2003).

Parameter								
W_∞	L_∞	K	Z	M	F	E	Data source	
5930	64.60	0.25	0.82	0.54	0.28	0.34	1985-1986	(Getabu ,1992)
-	63.10	0.35	1.71	0.72	0.99	0.58	1989-1990	(Dache, 1994)
4534	59.50	0.66	2.42	1.07	1.34	0.55	1998-2000	(Njiru ,2003)

These changes in biological and population characteristics of *O. niloticus* could be tactics to maximize survival and reproductive success (Njiru *et al.* in press), possibly linked to a population response to overfishing (Cowx *et al.,* 2003). The number of boats on the lake has increased from 4000 in 1950s to over 40 000 in 2000 supporting 12 000 and 120 000 fishers, respectively. There is also a substantial amount of illegal and undersized nets in the lake, which mostly target juveniles fishes all the species (Cowx *et al.,* 2003). However, even under this pressure *O. niloticus* still grows to large sizes and maintains a good condition (close to 1) (Njiru *et al,.* in pressure). This implies that if proper management measures are instituted the fishery of *O. niloticus* can continue providing protein and income especially to the local communities around the lake who depend mostly entirely of the fishery of Lake Victoria.

Recommendations

Management measures should include restriction on entry to the fishery which is open access by limiting and imposing licences on boats and gears. Ban on the undersized nets and other illegal fishing method should be enforced. The recommended gill mesh size for *O. niloticus* fishery is 5 inches (127 mm), but 2-3 inches were frequently encountered during the present study. Beach seining, although banned it is still prevalent in Lake Victoria. This scenario has led to intense exploitation of the *O. niloticus.* There is a need to involve community in management of the lake resources because fisheries management undertaken by the government through the Fisheries Department apparently has not succeeded. Communities are the first beneficiaries of the lake resources and if properly sensitized they can be willing to protect the resource. Alternative livelihoods, such as aquaculture and other agriculture activities should also be encouraged to reduce pressure on the fishery. There is lack of funding to do research and management of Lake Victoria fisheries. Most of the money got from the lake is not ploughed back, thus is need to advance a harmonized levy to be impose on the fishery and same should be used in management of the lake.

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References

Balirwa, J. S., 1992. The evolution of the fishery of *Oreochromis niloticus* (Pisces: Cichlidae) in Lake Victoria. *Hydrobiologia* **232**, 85-89.

Balirwa, J. S., 1998. *Lake Victoria wetlands and the ecology of the Nile tilapia*, *Oreochromis niloticus* (L) PhD thesis, University of Wageningen, A.A. Balkema, Rotterdam.

Cowx I.G.., Van der Knaap M., Muhoozi, L.I and Othina A. (2003) Improving fishery catch statistics for Lake Victoria. *J. Ecosys. Health Mgmt* **6**, 299-310.

Dache S.A.O., 1994. Observations on the fisheries, growth and mortality of *Oreochromis niloticus* (tilapia) in Nyanza Gulf of Lake Victoria. *In.* E. Okemwa, E. Wakwabi & A. Getabu eds. *Proceedings of the second EEC regional seminar on recent trends of research on Lake Victoria fisheries*, 25-27 Sept. 1994, Kisumu, Kenya. ICIPE Press, Nairobi, Kenya, pp. 59-65.

EAFFRO, 1955/56. East Africa Freshwater Fisheries Research Organisation, Jinja, Uganda. *Annaul. Report*. 23 pp.

Getabu, A., 1992. Growth parameters and total mortality in *Oreochromis niloticus* (L) from Nyanza Gulf, Lake Victoria. *Hydrobiologia* **232,** 91-97.

Getabu A., 1994 Mortality rate, exploitation and recruitment in *Oreochromis niloticus* (L) in Nyanza Gulf of Lake Victoria, Kenya, pg 43-52 In: Okemwa E, Wakwabi E. and Getabu, A. eds. *Proceedings of the second EEC regional seminar on recent trends of research on L.Victoria Fisheries 25-27 Sept.1994 Kisumu, Kenya*. ICIPE Press.

Kaufman L, 1992. Catastrophic change in species rich freshwater ecosystem: lessons of Lake Victoria*. Bioscience* **42**, 846-858.

Kolding J., 1993. Population dynamics and life-history styles of Nile tilapia, *Oreochromis niloticus,* in Ferguson's Gulf, Lake Turkana, Kenya. *Environmental Biology of Fishes* **37**, 25-46

Kudhogania A.W. and Cordone A.J., 1974. Batho-spatial distribution patterns and biomass estimation of the major demersal fishes in Lake Victoria. *African Journal of Hydrobiology and Fisheries*. **3**, 15-31.

Lung' ayia H.B.O., 1994. Some aspects of the reproductive biology of the Nile tilapia, *Oreochromis niloticus* (L.) in the Nyanza Gulf of Lake Victoria, Kenya. *In.* E. Okemwa., E. Wakwabi and A. Getabu eds. *Recent Trends of Research on Lake Victoria Fisheries*. ICIPE Science Press, Kenya.

Lung'ayia, H.B.O., M'Harzi, A., Tackx, M., Gichuki, J., and Symoens, J.J., 2000. Phytoplankton community structure and environment in the Kenyan waters of Lake Victoria. *Freshwater Biology* **43**: 529-543.

Lowe-McConnell, R.H., 1955. The fecundity of tilapia species. *East Africa Agriculture*. *Journal*. **11,** 45-52.

Lowe-McConnell, R.H., 1958. Observations on the biology of *Tilapia nilotica* Linné in East Africa waters*. Rev. Zool. Bot. Afri.* **57**, 130-172.

Mann, M. J., 1969. A resume of the evolution of the *Tilapia* Fisheries of Lake Victoria up to year 1960. EAFFRO Ann. Rep. 355: 18-41.

Moriarty C. M. and Moriarty D. J. W., 1973. Quantitative estimation of the daily ingestion of phytoplankton by *Tilapia nilotica* and *Haplochromis nigrispinnis* in Lake George, Uganda. *Journal of Zoology* **171**: 15-23.

Njiru, M., Othina, A., Getabu, A. Tweddle, D., & Cowx, I.G., 2002. ''The invasion of water hyacinth, *Eichhornia crassipes* Solms (Mart.), a blessing to Lake Victoria fisheries''*. In.* I.G. Cowx, ed. *Management and Ecology of Lake and Resevoirs Fisheries*. Fishing News Books, Blackwell Science, Oxford, UK.

Njiru M., 2003. *Ecology and population characteristics of Nile tilapia, Oreochromis niloticus (l.) In Lake victoria, Kenya.* Phd Thesis, Moi University, Eldoret, Kenya. 199pp

Njiru, M.,Okeyo-Owuor, J. B, Muchiri, M., & Cowx I.G., 2004. Shift in feeding ecology of Nile tilapia in Lake Victoria, Kenya. *African Journal of Ecology* **42**, 163-170.

Njiru, M., Waithaka, E., Muchiri, M., van Knaap, M. and Cowx, I.G., 2005. Exotic introductions to the fishery of Lake Victoria: What are the management options? *Lakes & Reservoirs: Research and Management* **10**, 147-

Njiru M., Ojuok J.E., Okeyo-Owuor J.B., Muchiri M., Ntiba M.J.and Cowx I. G. (In press). Some biological aspects and life history strategies of Nile tilapia *Oreochromis niloticus* (L.) in Lake Victoria, Kenya**.** *African Journal of Ecology*

Ogari J and Dadzie S., 1988. The food of the Nile perch, *Lates niloticus* (L.) after the disappearance of the haplochromine cichlids in the Nyanza Gulf of Lake Victoria (Kenya). *Journal of Fish Biology* 32, 571-577

Ogutu-Ohwayo R., 1990a. The decline of the native fishes of lakes Victoria and Kyoga (East Africa) and the impact of introduced species, especially the Nile perch, *Lates niloticus* and the Nile tilapia, *Oreochromis niloticus*. *Env. Biol. Fish*. **27**, 81-96.

Ogutu-Ohwayo, R., 1990b. Changes in the prey ingested and the variations in Nile perch and other fish stocks in Lake Kyoga and the northern waters of Lake Victoria (Uganda). *Journal Fish Biology* **37**, 55–63.

Opiyo S.V. and Dadzie S., 1994. Diet utilization in *Oreochromis esculentus* (Graham) in Lake Kanyaboli, Kenya. *Fisheries management and Ecology **1,** 79-90.*

Sparre P. and Venema S. C., 1998. Introduction to Tropical fish stock assessment. Manual Part 1. *FAO Fisheries Technical Paper* no 306 uss, R. E., 1979. Reliability estimates for Ivlev's selectivity index, the forage ratio and a proposed linear index of food selection. *Transactions of American Fisheries Society* **108**: 344-355.

Trewavas E., 1983. *Tilapiine species of the genera* Sarotherodon, *Oreochromis and Danakilia*. London: British Museum (Natural History) Publications No 878 p

Welcomme R. L., 1966. Recent changes in the stocks of *Tilapia* in Lake Victoria. Nature 21: 52-54

Welcomme, R. L., 1967. Observations on the biology of introduced species of *Tilapia* in Lake Victoria. *Rev. Zool. Bot. Afri*. **76**, 249-279.

Witte, F., & VAN Densen, W.L.T., 1995. *Fish Stocks and Fisheries of Lake Victoria. A Handbook for Field Observations*. Cardigan, Britain, Samara Publishing, Cardigan UK.

The potential of lake-river interface habitats as refugia for endangered fishes in Lake Victoria

Nkalubo W.N.[1], Balirwa J.S. [1], and R. Ogutu-Ohwayo[2]

[1]Fisheries Resources Research Institute (FIRRI), P. O. Box 343, Jinja, Uganda,
[2]Lake Victoria Fisheries Organisation (LVFO), P. O. Box, 1625, Jinja, Uganda,
[1]Email-firi@firi.go.ug

Abstract

The dramatic decline in fish species diversity in East Africa's Lake Victoria (surface area: 68,800 km^2) has been associated with the introduction of the predatory fish, the Nile perch (*Lates niloticus* L.). Due to the importance of biodiversity in ecosystem sustainability, the decimation of an endemic fish fauna comprising at least 350 species in less than two decades by the Nile perch may seem to be in contradiction with its emergence as an important export commodity. The potential of *in-situ* conservation greatly depends on identifying habitats that can act as refugia for remnants of the endemic fish fauna. For Lake Victoria, satellite lakes and macrophyte-dominated shores have been recognised as *in-situ* refugia, but little is known about the impact of the Nile perch on the endemic fauna of the adjoining Victoria Nile.

Experimental gill-netting in a 5 km – long interface zone upstream the present Owen Falls Dam conducted between May and August 2001 aimed at establishing whether despite its immediate connection to the Nile perch dominated open water habitats of Lake Victoria, the interface still contains fish species that have essentially become extinct in the lake. Despite the hydrological separation of fishes in the main lake by the dam from fish populations downstream in the river, the lake-river interface habitats still contain several endemic fish species (e.g. *Bagrus docmak*) that are no longer caught in the main lake. 17 fish species from nine families, majority endemic, were collected. The study thus illustrated the ecological importance of structural features (rocks, current) in the *in-situ* survival of endangered species of Lake Victoria.

Key words: Fish diversity, Faunal refuge, predation

Introduction

Wildlife and forest reserves have been carved out of terrestrial ecosystems to preserve species otherwise threatened by unregulated human activities such as agriculture. In aquatic systems, reserves may protect fish from diverse threats including over-fishing, predation by introduced exotic species, pollution and also hydropower projects in the case of rivers. However due to the physico-chemical properties of water, some threats such as pollution and the mobility of non-indigenous fishes, aquatic reserves in a single water body may be less effective than terrestrial reserves, which can be physically zoned off.

The drastic decline or disappearance of many indigenous species that coincided with the expansion of the Nile perch population in the 1980s in lakes Victoria and Kyoga, led to subsequent changes in the lakes' fish stocks and hence dramatic changes in their ecology (Balirwa *et. al*., 2003). An estimated two thirds of the *ca.* 350 small (<15cm long) haplochromines became extinct in less than two decades (Witte *et al.*, 1992a,b; Lowe McConell, 1997). Recent studies, however, have shown that habitats such as rocky shores and offshore rocky islands (Seehausen, 1999), satellite lakes (Nagayi, 1999) and macrophyte-dominated habitats (Balirwa, 1998; Chapman *et al*., 2002) provide refugia for the endangered indigenous fishes of the Lake Victoria basin. Fish surveys of the Nile system in Uganda since 1987 have shown that the Victoria Nile as a whole contains many of the species, which were once a major fishery of lakes Victoria and Kyoga (Balirwa, 1990; Musenero, 2000; Atkins/ FIRRI, 2001), but have now disappeared in the two lakes. Therefore, the upper Victoria Nile (Figure 1) which by virtue of the dynamic, spatial and temporal hydrological changes that occur in large rivers (Vannote *et al*., 1980), may also contain important refugia for the endangered fishes.

The major objective of this study was to describe the fish fauna of the 5-km interface zone between Lake Victoria and its outflow to the Victoria Nile, and to detect ecological factors within this zone that may contribute to fish refugia. The study focused on fish species distribution, their frequency of occurrence, and species diversity in the macro-habitats of the sub-zones between the lake and the dam.

Materials and methods

Study Area and site selection

A 5-km long interface zone (Latitudes $0^0$24'N to $0^0$26'N; Longitudes $33^0$11'E to $33^0$12'E) between Lake Victoria and the River Nile, the sole outflow from Lake Victoria, was investigated between May and August 2001 (Figure 1). The upstream section of outflow from Lake Victoria also known as the Upper Victoria Nile is characterised by a gradual change in current from lake to river. Prior to 1954, the headwaters of the River Nile comprised of a 15 m high Ripon Falls. The construction of the Owen Falls Dam in this area formed a physical barrier to fish movements between the lake and the rest of the river and also submerged the Ripon Falls and thus the out-flow regime.

During the study, the lake-river interface was divided into three transects: the upstream shallow (5.8m) area reflecting lake-like conditions (Transect I), the mid-section (15.4m) corresponding to the submerged Ripon Falls (Transect II) and the deeper

(22.5m) area upstream of the Owen Falls Dam (Transect III). The transects also had location sites: (Triangle-Kiryowa (Transect I); Bukaya-Kiira (Transect II); Sunset-Nytil (Transect III) with identifiable dominant features including landing sites.The location sites were used as bases for field sampling.

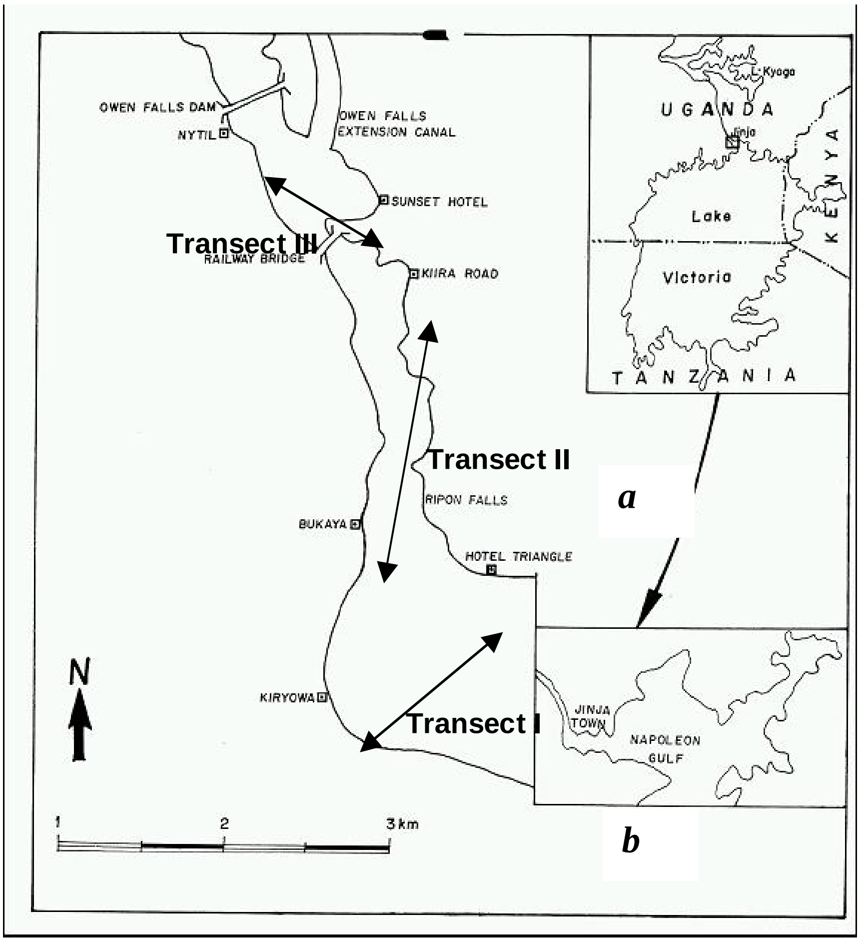

Figure 1. Map showing the study area (with inset a, the Victorian Nile connection between Lake Victoria and Lake Kyoga; inset b, Napoleon Gulf, the part of Lake Victoria connecting to the Upper Victoria Nile). Study sites are represented by Transects I to III.

Field Sampling and data analyses

Three fleets of multi-filament multimesh gill nets (25 mm to 203 mm (25.4, 38.1, 50.4, 63.5, 76.2, 88.9, 101.6, 114.3, 127, 139.7, 152.4, 177.8, 203.2mm i.e. 1” to 8” in half inch mesh increments) were used to sample fish in the three identified transects. In each transect, experimental fishing was carried out from both the eastern and western banks of the interface zone. One fleet of nets made up of the different mesh sizes was set as close to the bank (about 2m) as was physically possible; the second fleet was set about 30-50m offshore, and the third fleet was set 60-100m offshore. In view of the fast flowing water, the fleets of nets were heavily anchored with stones to prevent drifting.

On the three sampling occasions carried out, the gill nets were set between 1700 and 1800 h and retrieved between 0600 and 0700h the following day. Fishes were identified following procedures in Greenwood (1966) and CLOFFA (Daget *et al.*, 1984,1986). Owing to taxonomic difficulties, most haplochromine fishes were not identified beyond genera, though trophic and taxonomic groups while noted were grouped under the *Haplochromis* genus as in Witte *et al.* (1992a) and biometric data (total length in cm, fresh mass in grams) were recorded. Fish richness (number of taxa) and percentage composition (numbers and mass) were determined for each transect.

Diversity indices were calculated to compare the structure of the fish communities among sampling sites using the Shannon Weaver Diversity Index (H’). $H' = -\Sigma p_i . \ln P_i$ (Shannon and Weaver, 1949), where P_i= proportion of total number of species found in the i^{th} species (n_i/N). Fish species evenness (as an indication of similarity in species abundance or occurrence) was also calculated using the formula $E = H' / \ln S$ where E= Evenness, H’= Shannon Weaver Diversity Index and lnS= natural log of number of species observed in a site sampled.

Results

Fish species of the Lake Victoria-Victoria Nile Interface

A total of 1471 fish specimens representing 17 fish species in nine families (Table 1), were collected from the study area over the sampling period. Four species (*Astatoreochromis alluaudi*, *Ptyochromis savagei*, *Pundamilia macrocephala*, *Paralabidochromis rockcribensis*) were the most common among the haplochromine cichlids; other haplochromines could not be identified to species level and beyond trophic groups (molluscivore, zooplanktivore and insectivore).

Table 1: Longitudinal distribution (presence/absence) of fish species caught in the Lake Victoria- Victoria Nile Interface habitats.(Tri = Triangle; Kir = Kiryowa; Buk = Bukaya).

		ZONE					
		TRANSECT 1		***TRANSECT 2***		**TRANSECT 3**	
FAMILY	**FISH SPECIES**	**Tri**	**Kir**	**Buk**	**Kiira**	**Sunset**	**Nytil**
CYPRINIDAE	*Barbus altianalis*	-	-	+	-	-	-
BAGRIDAE	*Bagrus docmak*	-	-	+	+	-	+
CHARACIDAE	*Brycinus jacksonii*	-	+	-	-	+	+
CHARACIDAE	*Brycinus sadleri*	-	+	-	-	-	-
CLARIDAE	*Clarias alluaudi*	-	+	-	-	-	-
CENTROPOMIDAE	*Lates niloticus*	+	+	+	+	+	+
MORMYRIDAE	*Mormyrus kannume*	+	+	+	+	+	+
CICHLIDAE	*Haplochromis spp.*	+	+	+	+	+	+
CICHLIDAE	*Tilapia zillii*	+	+	-	+	+	+
CICHLIDAE	*Oreochromis leucostictus*	-	+	-	-	-	-
CICHLIDAE	*Oreochromis niloticus*	+	+	+	+	+	+
PROTOPTERIDAE	*Protopterus aethiopicus*	-	+	-	-	-	-
MOCHOKIDAE	*Synodontis afrofischeri*	+	+	+	+	+	+
MOCHOKIDAE	*Synodontis victoriae*	-	-	-	+	+	+

Excluding the haplochromine cichlids, four other fish species (*L. niloticus*, *M. kannume*, *O. niloticus* and *S. afrofischeri*) were encountered in all transects (Table 1). *Brycinus sadleri*, *C. alluaudi*, *O. leucostictus* and *P. aethiopicus* occurred only in Transect 1 and *Barbus altianalis* only in Transect 2. Haplochromine fishes collectively dominated the abundance by 28% in terms of numbers; followed by *L. niloticus* and *M. kannume* in equal proportions (26% each, Figure 2). The introduced species (*L. niloticus,* 31.3%, *O. niloticus,* 16.5% and *Tilapia zillii,* 3.1%*)* contributed 51% of the biomass compared with 49% to native species (*M. kannume,* 33%, *Haplochromis spp.*, 6.6% and others e.g. *Synodontis*, *Bagrus docmak*, *Barbus altianalis*, *Brycinus*, 9.4%).

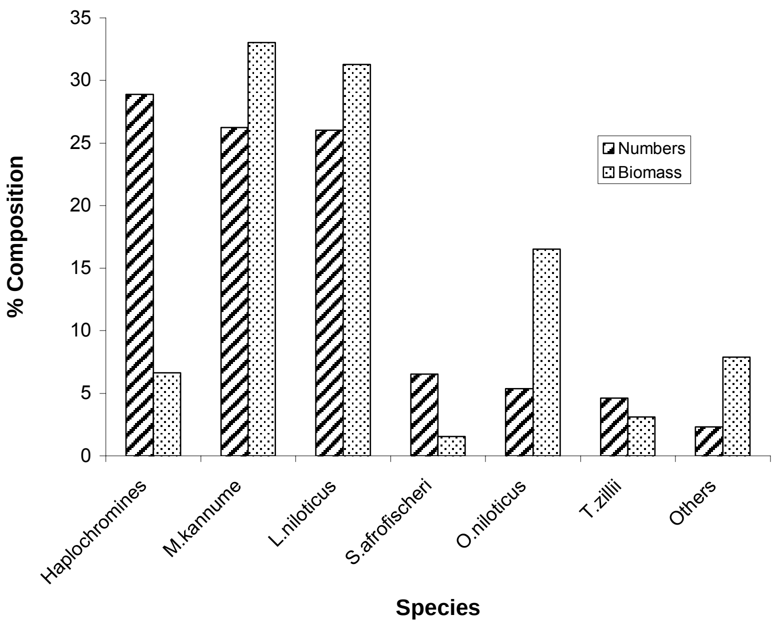

Figure 2. Percentage Composition (by number and biomass) of the fish species encountered in sampled sites.

The relative abundance of species varied among transects (Figure 3a). In terms of numbers, haplochromines and *T. zillii* were the most represented in Transect I (54%; 10.8%). *Mormyrus kannume* and *L. niloticus* were most numerous in Transects II (55.8%; 28.7%) and III (27.5%; 37.9%) compared to Transect I (8.7%; 12%). The frequency of occurrence of *S.afrofischeri* and *O.niloticus* in transects I and III was comparable. In general, the high abundance of individual species yielded high biomass values within the different transects (Figs 3a and 3b). However, on comparing the different species within individual transects, the low biomass values resulting from high abundance (e.g. the haplochromines) implied a small size for these fish, and vice versa for the corresponding high biomass values with low abundance (e.g. *M. kannume*) indicating large sized fish (Figs 3a and 3b). Kiryowa (Transect I) registered a higher number of taxa including several species that did not occur in samples from other sampling sites (Table 1). The Shannon Index (H') calculations also revealed Kiryowa (H' =1.69) to have had the highest fish diversity, but the within transect analysis indicated that transects II registered the lowest diversity (H' =1.21) and species evenness (E =0.5) while transect III showed the highest values in both indices (Table 2).

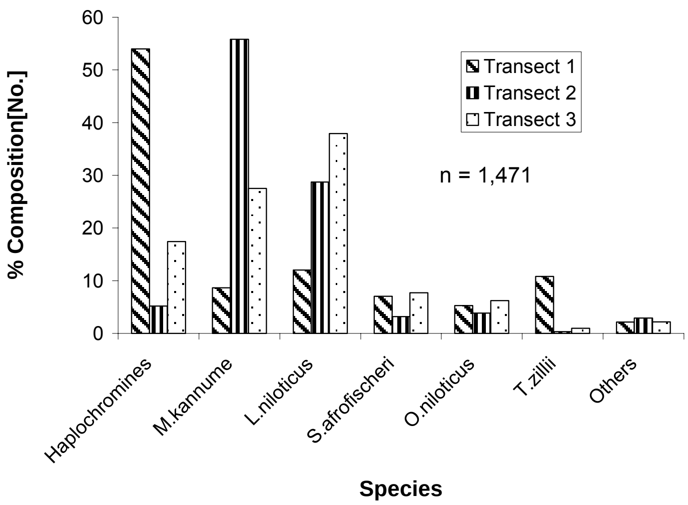

Figure 3a. Variation in relative abundance across transects.

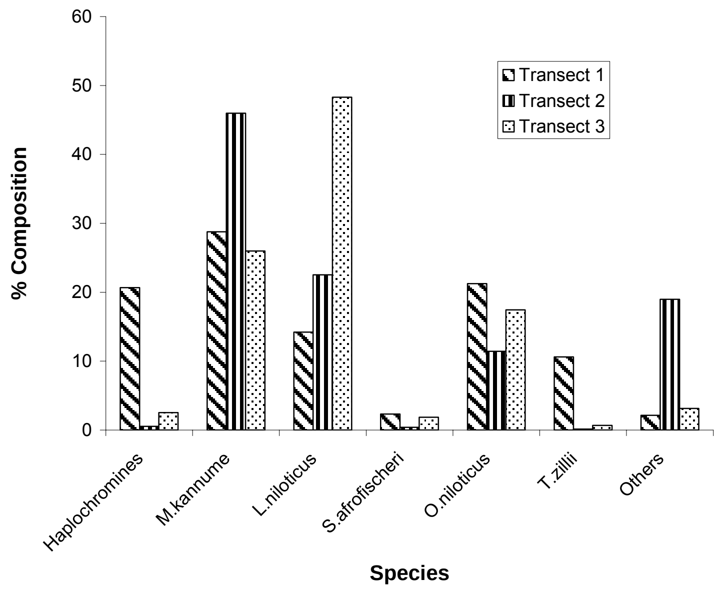

Figure 3b.Variation in relative abundance across transects.

Discussion

The investigated area comprises the headwaters of the upstream-most zone of the River Nile also referred to as the Upper Victoria Nile. Prior to the construction of the Owen Falls Dam in the 1950s, there was a direct connection of flowing water from Lake Victoria into the river. The falls and rapids in this stretch then known as the Ripon Falls, were a natural barrier to upstream migration of fishes from the downstream river Nile fishes. With the construction of the Owen Falls Dam, 5 km downstream from the open lake, it would be expected that downstream fish movements would be blocked off, and that the fishes introduced into Lake Victoria in the 1950s/1960s would exert the same influence in this interface zone as they did in the main lake.

Despite the hydrological separation of fishes in the main lake by the dam from fish populations downstream in the river, this study however revealed that lake-river interface habitats still contain several endemic fish species (e.g. *Barbus altianalis, Bagrus docmak, Mormyrus kannume*) rarely caught in the lake. Also as evidenced by Seehausen (1996a,b), and others (Seehausen *et.al.,* 1998), habitats (such as rocky littoral banks and macrophytes) encountered in Transect I affected species diversity through increased habitat heterogeneity. The abundance of *Haplochromis spp.* and *T. zillii* in the upstream (Transect I) section of the interface zone compared to downstream (Transects II and III) areas depicts demonstrations by McNeely (1986) and Matthews (1998) of habitat diversity overriding the typical longitudinal patterns. The highest fish species diversity observed in transect I, which depicted lake like conditions, showed the efficacy of this stretch in promoting the survival of endangered species. Typical riverine stretches (e.g. Transect II) lacked the potential of harbouring such species diversity, thus, the localisation of certain fish species to specific habitats (e.g. rocky banks and macrophyte dominated stretches) depicted a variation in the distribution patterns of the identified species along this interface zone. The presence of such high flow velocities along river stretches as was associated with this transect, probably does not only limit species survival rates, but also hinders their occurrence in such agitated areas.

Possible implications of the dam

Dam construction has been reported (Scudder & Conelly, 1985) to affect the variability of the riverine fishes. Following the construction of the dam across the Nile River, the water level extended over part of the river creating a reservoir, which replaced the flowing water with a more static lacustrine water body (Olowo & Chapman, 1999; Welcomme, 2001). The existing undercurrent resulting from the creation of this reservoir probably provide lotic conditions over a short section (Transect III) thus promoting the survival of lacustrine species such as *Haplochromis spp.* and *T. zillii* (though in this case not as in the open lake) (Nkalubo, 2001). However, the negative effects on original Victoria Nile fish fauna arising from the dam construction, that have also contributed to the diminished abundance, disappearance and even extinction of some of these native species, especially the migratory ones can not be ignored.

Structural features (current, rocks) as is illustrated by this study should therefore be highly regarded as factors of ecological importance that could be used to define fish refugia. The ability to enhance the persistence of native species with the exotics elucidates their vital role in conserving the declining fish diversity in the lake region. Therefore, it is recommended that further hydropower projects considered along the River Nile should include provision for the preservation of such features (e.g. rocky stretches) with adequate flow regimes.

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References

Atkins W.S & FIRRI, 2001. *AES Nile Power- Bujagali Hydropower Project*. Haplochromine Habitat Study. Report No. AF6097/70/dg/1215 Rev.2.0. 67p.

Balirwa J. S., 1990. *Perspectives of Conservation and Management of the River Nile System in Uganda. With special reference to the Fisheries. The Conservation and management of river.* An International conference, University of York, 10-13th Sept. 17p.

Balirwa J.S., *1998. Lake Victoria wetlands and the Ecology of the Nile Tilapia,*

Oreochromis niloticus (Linne). PhD thesis of Wageningen Agricultural University, The Netherlands. 247p.

Balirwa J.S., Chapman C.A., Chapman L.J., Cowx I.G., Geheb K., Kaufman L., Lowe-McConnell R.H., Seehausen O., Wanink J.H, Welcomme R.L. & F. Witte, 2003. Biodiversity and Fishery Sustainability in the Lake Victoria Basin: An unexpected Marriage? *BioScience* 53(8): 703-715.

Chapman L.J., Chapman A., Nordie G. and A. Rosenberger, 2002. Physiological refugia swamps, hypoxia tolerance and maintenance of fish diversity in the Lake Victoria region. *Comparative Biochemistry and Physiology* Part A 133 421-437.

Daget J., Gosse J.P & Thys van den Audenaerde D.F.E eds., 1984. *Check List of the Freshwater Fishes of East Africa*, Volume I. ORSTOM, Paris. 410p.

Daget J., Gosse J.P & Thys van den Audenaerde D.F.E eds., 1986. *Check List of the Freshwater Fishes of East Africa*, Volume II. ORSTOM, Paris. 520p.
Greenwood P.H., 1966. *The fishes of Uganda*, *2nd Edition*. The Uganda society, Kampala. 133p.
Lowe McConnell R., 1997. EAFRO and after: a guide to key events affecting fish communities in Lake Victoria (East Africa). *South African Journal of Science* Vol. 93.
Matthews W.J., 1998. *Patterns in Fresh water Fish Ecology*. Chapman and Hall. 756p.
McNeely D.L., 1986. Longitudinal patterns in the fish assemblages of an Ozark Stream. *Southwest. Nat.,* 31,195-208.
Musenero R., 2000. *The distribution patterns and feeding ecology of Nile perch, Lates niloticus (L.) in the Victoria Nile (Uganda).* M.Sc. thesis of IHE Delft, The Netherlands. 69p.
Nagayi J.K. 1999. *Population characteristics of Oreochromis esculentus in satellite lakes of the Victoria and Kyoga lake basins*. M.Sc. thesis of Makerere University, Kampala. 86p.
Nkalubo W. 2001. *Fish populations and trophic characterisation of key stone species at the Lake Victoria-Victoria Nile interface habitats.* MSc thesis of IHE Delft, The Netherlands. 141p.
Olowo J.P and L.J. Chapman, 1999. Trophic shift in predatory catfishes following the introduction of the Nile perch into Lake Victoria. *Afr. J Ecol* 37:457-470
Scudder T. and M. Conelly, 1985. *Management systems for riverine fisheries*. FAO Fish Tech. Pap. (263): 85p
Seehausen O., 1996a. A reconstruction of the ecological composition of the cichlid species flock in Lake Victoria before and after the Nile perch boom. *In*: W. van densen and R. Lowe-McConell eds.. *Lacustrine fish Communities in S.E. Asia and Africa.* Samara Publishing Ltd., Cardigan, Dyfed. U.K.
Seehausen O. ed., 1996b. *Lake Victoria cichlids- taxonomy, ecology and Distribution. Verduyn cichlids*. Germany. 304p.
Seehausen O., Lippitsch E., Bouton N. and H. Zwennes, 1998. Mbipi, the rock dwelling cichlids of Lake Victoria: a description of three new genera and fifteen new species. *Icthyological exploration of freshwater*, 9: 129-228.
Seehausen O., 1999. A reconsideration of the ecological composition of the cichlid species flock in Lake Victoria before and after the Nile perch boom *In*. W. L. T van Densen & M. J Morris eds. *Fish and fisheries of lakes and reservoirs in Southeast Asia and Africa*. Westbury Publishing, Otley, UK p281-293.
Shannon C.E and Weaver W., 1949. *The mathematical theory of communication.* Urbana, 11: University of Illinois Press
Vannote R. L., Minshall G.W., Cummins K.W and J.R. Sedell, 1980. The River Continuum Concept. *Can. J. Fish & Aq. Sc.* 37. 130-137.
Welcomme R.L., 2001. *Inland fisheries- Ecology and management*. Published for Food and Agricultural Organisation of the United Nations by Blackwell Oxford, London. 358p.
Witte F., Goldscmidt T., Goudswaard P. Ligtvoet W., Van Oijen M. and J. Wanink, 1992a. Species Extinction and Concomitant ecological changes in Lake Victoria. *Neth. J. Zool*. 42 (2-3). 214-232.
Witte F. Goldschmidt T, Wanink J., Van Oijen M., Goudswaard K., Witte-Maas E and
N. Bouton, 1992b. The Destruction of an endemic species flock quantitative data on the decline of the haplochromine cichlids of Lake Victoria. *Env. Biol. Fish.* 34 : 1-28.

Abundance and composition of zooplankton in Lake Victoria, Kenya

Owili M[1], R. Omondi[1] and J. Njuru[2]

[1]Kenya Marine and Fisheries Research Institute, P.O. Box 1881, Kisumu- Kenya

[2]Ministry of water and irrigation, P. O. Box 1922, Kisumu- Kenya

[1]E-mail: owili@yahoo.co.uk

Abstract

Lake Victoria has undergone major ecological changes due to the introductions of alien fish species since the 1950's and the increased nutrient input resulting in eutrphication. This trend is expected to continue with the invasion by water hyacinth in the 1990,s and the subsequent macrophytic succession coupled with habitat degradation and pollution in the catchments. Zooplankton has potential value as assessors of trophic conditions since they respond quickly to environmental change and may be effectively indicators of changes in water quality. It is therefore important to monitor such changes in composition and abundance of zooplankton in order to understand the changes in the lake's trophic conditions and the environment. Zooplankton community structure in Lake Victoria, Kenya was investigated between August 2001 and May 2002 using a-80µm mesh size net. Physico-chemical parameters were measured *in situ* using YSI multi-parameter hydrolab.

There were spatial variations in physico-chemical parameters measurement values and zooplankton abundance, composition and distribution. Secchi transparency and dissolved oxygen increased offshore while conductivity and pH decreased offshore. The mean total zooplankton densities decreased progressively offshore from 203.6±66.8-ind./l. to 9.4±1.6 ind.l at KL1 (2.5m) and KP2 (59m) respectively. The higher abundance in the gulf is attributed to higher nutrient input resulting in higher primary production.

Key words: Zooplankton, Lake Victoria, indicators, and eutrophication

Introduction

The role of zooplankton in the functioning and production of aquatic ecosystems is vital (Downing, 1984; Mavuti & Litterick, 1991). As major primary consumers many zooplankters convert algal production into animal material for carnivorous invertebrates and fish. Zooplankton, especially Crustacea, forms an important component of the diet of carnivorous invertebrates such as chaoborids, chironomids and *Caridina nilotica* and fishes (Ogari & Dadzie, 1988; Hymblyn, 1966) in Lake Victoria. Larvae of most fish species such as *Lates niloticus*, *Oreochromis niloticus* in the lake rely on these organisms as their first food after resorption of yolk sac, while *Rastrineobola argentea* and some haplochromines feed on zooplankton throughout their lives. Zooplankton affect phytoplankton population through grazing which in turn has profound effects on water quality and can be useful as indicators. There is, therefore, a strong direct relationship between the dynamics of zooplankton population and fishery production, and need to further understand the role of zooplankton in the food web and in the energy transfer pathway for the major fish species of Lake Victoria.

The community structure, species diversity, and standing stock biomass of zooplankton vary in lakes depending on the prevailing limnological and trophic conditions. These attributes are influenced by the relative strength of two interacting forces. (Brooks & Dodson 1965; Carvalho, 1984; Anderson, 1978), One, being predation and grazing mainly by fish which is a major factor in structuring the zooplankton communities in freshwater intems of species composition and body size (Hrbacek, 1962;and Maciej, 1994), and supply of nutrients to the phytoplankton and the algal food for zooplankton which controls the biomass.

One of the characteristic features of tropical lakes and reservoirs is the small-bodied size zooplankton taxa, which is attributed to size based high and persistent selective predation by the planktivorous fish. The large bodied zooplankton individuals are therefore more vulnerable to extinction in tropical waters as compared to temperate areas (Brook & Dodson, 1965).

Zooplankton community of Lake Victoria has not been studied extensively (Mavuti & Litterick ,1991), but a remarkable change in composition and relative abundance from one dominated by large-bodied herbivores to one dominated by small-bodied species has been reported (Mwebaza-Ndaula, 1994; Gophen et al, 1995; Wanink, 1998). During the 1930's and 1950's the large bodied calanoids and cladocerans dominated the zooplankton community (Worthington, 1931, Rzoska, 1957) This has probably been due to changes in water transparency, food availability and fluctuations in abundance of predators. The changes in zooplankton and fish community have resulted in the simplication of the food wed of the lake. The depletion of the haplochromine community due to predation by the introduced *Lates niloticus* has resulted in reduced grazing pressure which turn has brought changes within the zooplankton community.

Understanding how physical, chemical and biological processes interact and influence water quality to impact fisheries production is very essential since changes in both the temporal and spatial scales of the physico-chemical environment determine phytoplankton and zooplankton dynamics in aquatic ecosystems and ultimately determine fisheries production in Lake Victoria.

Objectives

To determine, abundance, distributions, and composition of zooplankton taxa in Nyanza, Lake Victoria, Kenya.

Materials and Methods

Study area

The study was carried out between August 2001 and May 2002 at stations established along the Nyanza Gulf and the main lake on the Kenyan portion of Lake Victoria (Fig.1). Eleven stations were sampled along the Nyanza Gulf from KL1 (2.5m) to KP2 (58m). These stations were part of the established lake-wide water quality sampling sites under the Lake Victoria Environmental Management Project (LVEMP).

Physico-chemical parameters were measured in situ using YSI Multi-parameter Hydrolab. Transparency (Secchi depth) was determined with a 25cm diameter secchi disc painted black and white.

Zooplankton samples were collected using a 1.0 m long Nansen type plankton net of 80 μm mesh size and mouth opening measuring 30 cm diameter. The net was hauled vertically through the water column noting the depth. The zooplankton samples were preserved in 5% formalin. In the laboratory each sample was made to a known volume, thoroughly shaken for uniform distribution and a sub-sample taken and placed in a counting chamber. Identification of the zooplankter was done using relevant. Estimates of abundance of crustacean zooplankton were made from counts of sub-samples under a Leica dissection microscope, at a magnification of x 25. The number of individuals per

Litre of lake water was determined by taking into account the, Volume of the sample, number of organisms in the sub-sample, volume of the lake water filter by the vertical haul and the depth of the haul.

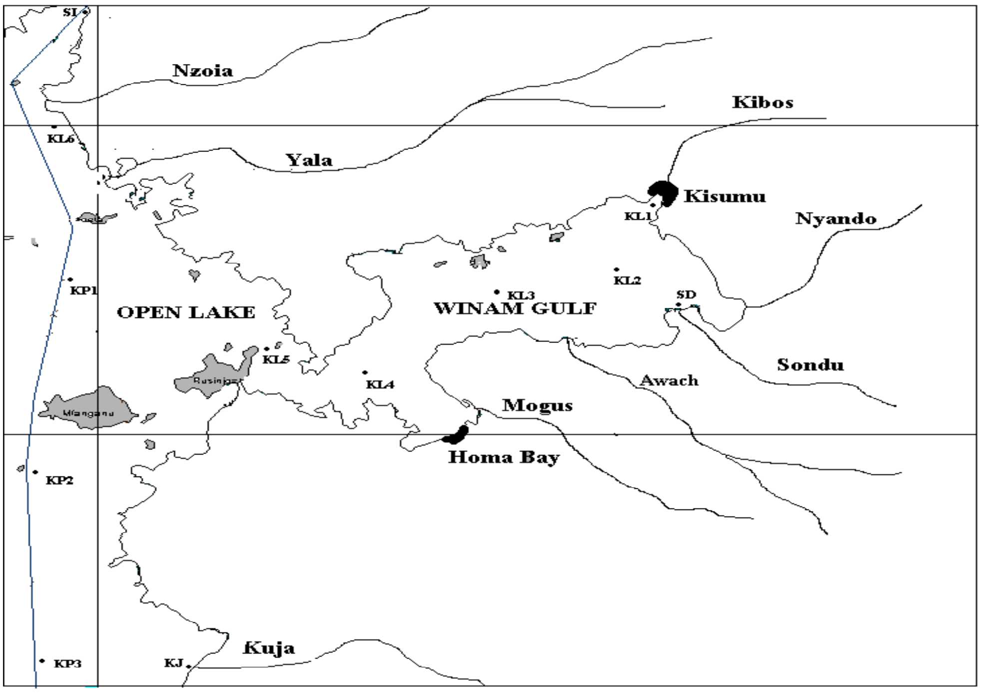

Figure 1. Map showing the study area, Lake Victoria, Kenya.

Results

Water surface temperatures were generally higher in the gulf than in the open lake (Table 1). Mean water surface temperatures ranged form 27.22±0.84ºC at KL1 to 25.67±0.7ºC at KP2. Water transparency (Secchi depth) increased with depth from 0.52±0.2 at Sondu miriu along the gulf to 3.1±3.4 at KP2 in the open lake. Conductivity was higher in the gulf than the open waters ranging 167.8±11μS/cm at KL1 to 100.4±3.5μS/cm at KP2. pH ranged from 7.37±0.5 at KP3 to 8.59±0.7 at KL2. Oxygen levels were generally high and increased with increasing depths to a maximum of 8.44±0.6 mg/l at KP2. The shallow inshore waters in the gulf were more turbid than the offshore water.

Table1. Physico-chemical characteristics of Lake Victoria, Kenya.

	Stations										
Parameters	**KL1**	**KL2**	**KL3**	**KL4**	**KL5**	**KL6**	**KP1**	**KP2**	**KP3**	**Sondu m**	**Kuja**
Mean Temperature (¼C)	27.22±0.2	26.6±0.1	27.19±1	26.46±1	26.58±0.5	26.99±0.7	25.67±0.2	25.97±0.2	25.95±0.2	26.32±0.6	25.76±0.43
Dissolved Oxygen (mg/l)	7.04±0.8	7.62±1.1	7.57±1.1	5.5±1	6.52±1.9	8.39±2.3	7.28±1.6	8.44±2.3	7.95±1.5	6.51±0.9	7.03±0.76
Conductivity (µS/cm)	167.82±1	161.3±3.8	158.96±6	156.82±8	113.7±2.5	105.3±5.3	101.3±2	100.36±4	101±4.3	122.63±9	100.5±1.74
Ph	7.75±0.3	7.86±0.5	8.59±0.7	7.44±0.4	8.09±0.7	8.17±0.56	7.92±0.4	7.9±0.77	7.37±0.5	9.91±0.25	7.49±0.6
Secchi depth (m)	0.84±0.2	0.95±0.2	1.1±0.1	1.54±0.4	1.92±0.4	1.6±0.5	2.6±0.37	3.1±0.46	2.26±0.2	0.52±0.2	0.98±0.59
Turbidity (NTU)	16.7±10.1	11.5±6.3	7.6±2.9	5.2±3.9	6.7±3.9	9.5±8.8	3.1±3.2	1.4±0.5	2.7±1.5	22.5±1.0	9.7±3.6
Depth (mean)	2.5	4	6	14	24	13	45	58	35	2	7

Zooplankton abundance

Mean densities of total zooplankton recorded at the eleven stations sampled are presented (Fig.2.). There were spatial variations in abundance, distribution and diversity Total zooplankton densities decreased generally along the gulf from KL1 to KP3 (littoral to pelagic waters). The littoral stations such as KL1, KL2, Sondu-miriu and Kuja recorded significantly higher densities**.** The densities decreased progressively from 203.6±66.8-ind. /l. to 9.4±1.6 ind.l at KL1 (2.5m) and KP2 (59m) respectively. Total zooplankton densities were 106.5±24.7 ind./l, 67.2±31.1,37±13.7 ind./l, and 149.1±67.2 ind./l 77.4±25.7 ind./l for KL2 KL3, KL4 Sondu-miriu and Kuja River mouths respectively. Cladocera densities decreased generally towards the open lake with higher mean densities of 35.53±16.6 ind.l^{-1} and 32.4±15.9 ind.l^{-1} at KL1 and Sondu –Miriu river mouth respectively while the deeper stations recorded very low densities.

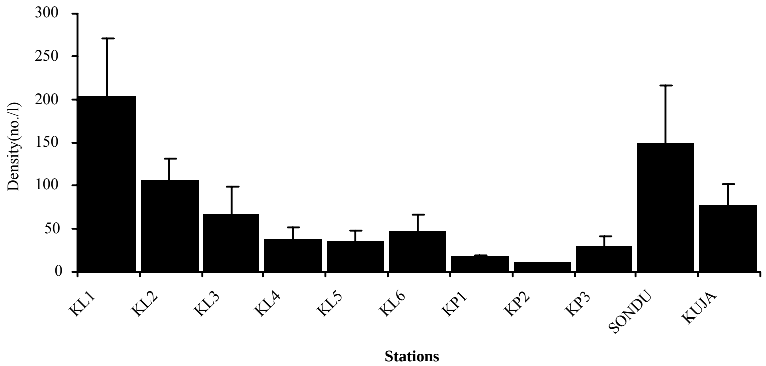

Figure 2. Spatial variation in zooplankton abundance (Mean±S.E.) in Lake Victoria, Kenya that the river mouths recorded relatively higher zooplankton densities.

Zooplankton composition and distribution

The zooplankton of Lake Victoria consists of three major groups namely: Copepoda Cladocera and Rotifera. Copepoda accounted for between 70% and 98% of the total zooplankton and are well distributed throughout the Lake (Fig.3).

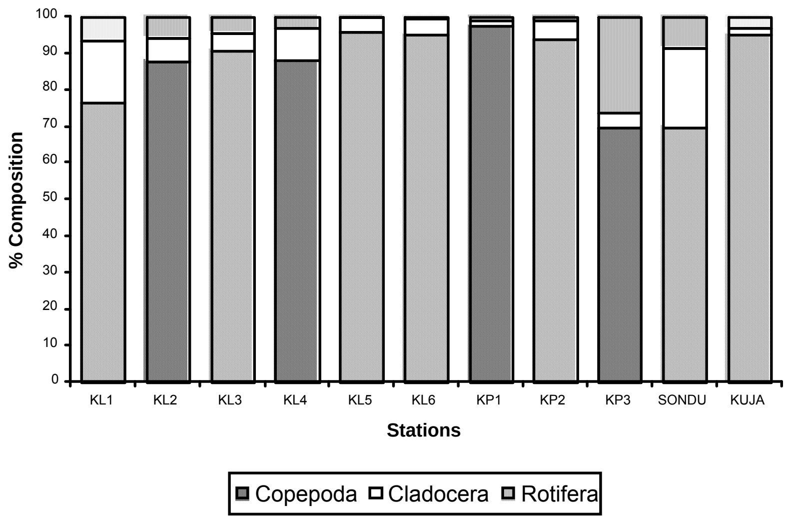

Figure 3. Spatial distribution of different zooplankton taxa in Nyanza Gulf.

Among copepods, immature individuals dominated in the shallow stations in the gulf (KL1-KL3) and at the mouth of Rivers Sondu and Kuja. The immature (Nauplii) had the highest proportion at the mouth of River Kuja (72%) followed by KL2 (60%). Cyclopoida, on the other hand, dominated the other groups in both the gulf and open waters. The highest proportion of Cyclopoida (60%) was in KL5.

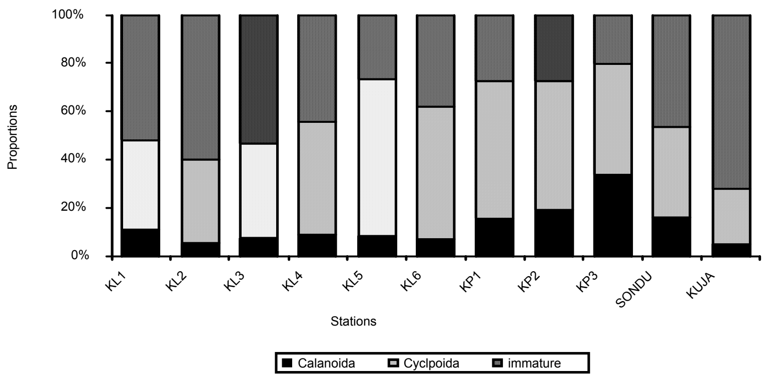

Figure 4. Percentage composition of copepod groups.

Cyclopoid copepoda occurred in higher proportions compared to Calanoida. Calanoida generally occurred in higher proportions in the deep water stations than in the shallower areas and a considerable number were also found at the mouth of River Sondu miriu. Both calanoid and cyclopoid copepods increased in proportions offshore (Fig.4) Nauplii and copepodites were persistently high in proportions in shallow stations, which decreased offshore. Rotifers were underestimated because the mesh size (80µm) used, however they occurred in low numbers but were high inshore than offshore. Cladocerans were also relatively low in proportion.

Taxonomic composition and species diversity

Copepoda was represented by 10 species dominated by *Thermocycops neglectus* followed by *T.emini.* Other Cyclopoid copepods recorded were *T.incisus, T.decipiens, Tropocyclops confinnis; T.tenellus and Mesocyclops* spp. Of the two species of calanoids found in Lake Victoria, *Tropodiaptomus stuhlmanni* dominated the shallower stations while *Thermodiaptomus galeboides* dominated the deeper regions (Table.2).

Seven species of Cladocera were recorded in the study ((Fig.5). The highest occurrence of cladocerans was observed at the Sondu River mouth. The family Daphnidae was represented by three species (*Daphnia lumholtzi, D.barbata*, and *D.longispina*), while the other three families viz. Sididae, Moinidae and Bosminidae with one species each i.e. *Diaphanosoma excisum, Moina micrura*, *Bosmina longirostris* respectively. These three species were widely distributed along the gulf. Among the cladocerans *Moina micrura*,

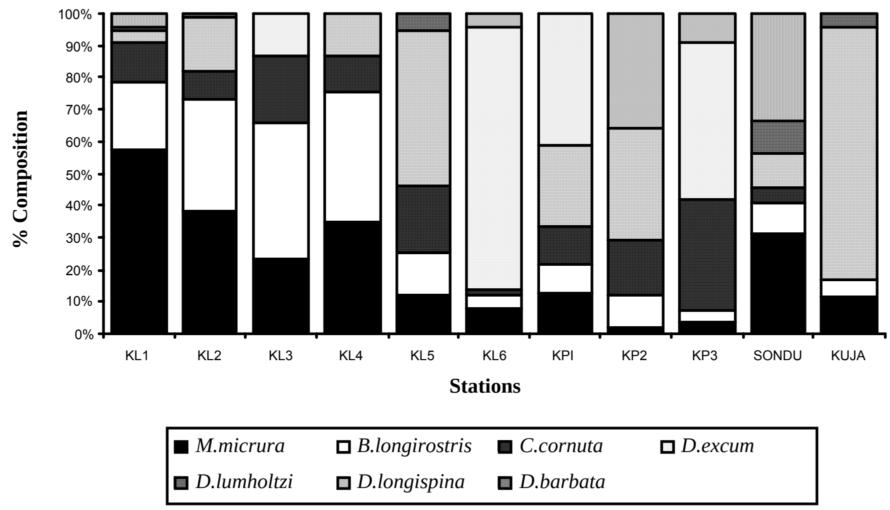

Figure 5. Spatial distribution of different cladoceran species in Nyanza Gulf.

Bosmina longirostris, *Ceriodaphnia cornuta, and Diaphanosoma exiscum* had a wide distribution throughout the lake, while *Daphnia lumholtzi was* rare. *Daphnia barbata* was restricted at KL1 and at Sondu miriu River mouth. *M. Micrura* and *B. longirostris* dominated the cladoceran population in shallow gulf stations, while the deeper waters were dominated by *D. excisum* and *Daphnia longispina*. *D longispina* was restricted to the open waters at KP1, KP2 and KP3 and KL6.

Rotifera contributed least among the three taxa, but was the most diverse group and were more or less restricted to the gulf. A total of 24 species of Rotifera were recorded. Their densities, decreased progressively from KL1 to KP2. However the highest contribution of rotifers was found at KP3 and was due to mainly one species, *Epiphanes sp*. The family Brachionidae dominated with 11 species**.** *Brachionus angularis, B.calyciflorus, B.caudatus and B.falcatus* were the most dominant species*. B.plicatilis, B.bidentatus, B.rubens Platyias patulus* were common especially in the gulf. Other rotifera species encountered included *Keratella tropica, K.cochlearis, Polyarthra spp, Hexarthra mira, Euchlanis sp, Epiphanes micruorus, Lecane spp. Keratella tropica, Filinia spp, Asplanchna brightwelli*, *Trichorcerca spp, Filinia longiseta*, *F.terminalis*, and *F.opoliensis* among others.

Table 2. Species of Zooplankton recorded in both littoral and pelagic zones (+ = recorded; - = not recorded or very rare).

TAXA	**LITTORAL**	**PELAGIC**
	Cladocera	
Bosmina longirostris	+	+
Ceriodaphnia cornuta	+	+
Daphnia lumhorltzi	+	+
D.barbata	+	-
D.longispina	-	+
Diaphanosoma excisum	+	+
Moina micrura	+	+
Calanoida		
Thermodiaptomus galeboides	+	+
Tropodiaptomus stuhlmani	+	-
Cyclopoida		
Mesocyclops spp.		
Thermocyclops emini	+	+
T. incisus	+	+
T. neglectus	+	+
T.decipiens	+	-
T.oblongatus	+	-
Tropocyclops confinnis	+	+
T. tenellus	+	+
Rotifera		
Ascomorpha spp	+	+
Asplanchna brightwell	+	+
B.angularis	+	+
B.bidentatus	+	-

B.plicatilis	+	-
B. calyciflorus	+	+
B. caudatus	+	+
B.falcatus	+	+
B.rubens	+	-
B.quadridentatus	+	-
Euclanis sp	+	-
Filinia longiseta	+	+
F.terminalis	**+**	**+**
F. opoliensis	+	+
Keratella cochlearis	+	+
K. tropica	+	+
Platyias patulus	+	-
Lecane spp	+	-
Polyrthra spp	+	-
Synchaeta spp	-	+
Hexarthra mira	+	-
Trichorcerca longiseta	+	+
Trichocerca spp	+	+
Epiphane spp	**+**	**+**

Discussion

Physicochemical condition observed is indication of the ever-changing Lake Victoria environment. The higher turbidity in the inshore gulf is normally a product higher suspended organic matter brought in by the rivers as well as the high algal blooms which is a common feature. The increased nutrient input results in eutrophication which enhance the thriving of high algal biomass and eventually control zooplankton production.

The zooplankton community structure, distribution pattern observed is largely related to both physico-chemical and biotic conditions prevailing in different ecological habitats. The changes in the dynamics of Lake Victoria zooplankton have largely been attributed to changes in water quality (bottom-up control) and predation pressure (up down control effects). The groups with high predation pressure are likely to be low in abundance compared to those that are always not selected for. Competition could also a factor controlling the zooplankton structure. The pattern of abundance observed could be explained in terms of the Lake's productivity as the inshore (Gulf) and river mouths tend to be more productive and thus supporting more zooplankton community than the less productive offshore waters. Higher Rotifera species diversity indicates increased eutrophication in the Gulf. Physicochemical factors such as turbity can also affect zooplankton community structure, by enhancing different predation pressures (Owili & Omondi, 2003) on the various zooplankters and primary production as well.

Zooplanktivorous fishes are known to be selective feeders and would go for the prey that they can capture at minimum cost. Cladocerans are reported to be more vulnerable to fish predation than copepods and rotifers due to their large size and slow mobility (Owili, 1999), This vulnerability of cladocerans could partly explain their low abundance in Lake Victoria, however it should be noted that the diet choice largely depend on prey availability and the fish will go for an alternative prey in the event of scarcity of the main prey. Fish in shallow areas tend to go more for Cladocera while in offshore they tend to feed more on copepods (personal observation). The wide distribution and the high abundance of cyclopoids in the lake are of great ecological importance on assumption that they are readily available for fish as food and therefore likely to contribute to fish production in the lake.

Increased Copepoda proportions offshore could be due to reduced predation pressure as the fish reduce in abundance. Though predation is major factor in structuring the zooplankton community, Wannik (1998) however argued that intensified predation alone cannot explain the decline of the larger zooplankters in Lake Victoria after the replacement of zooplanktivorous haplochromines by *R.argentea* and Nile perch, since the present lumped biomass of the two species is lower than the biomass of the original haplochromine-dominated zooplanktivorous.

Low relative abundance of rotifera observed is due to underestimation given that the mesh size of the net used was rather being for maximum harvesting of this group. The observed trend agree with previous work done along the gulf (Mavuti & Litterick, 1991). However, the difference in densities could be due the difference in mesh size of the net used and also the habitat sampled since the earlier studied many littoral sites.

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References

Andersson, G. Berggren, H. Cronberg, G.amd Gerlin C., 1978. Effects of Planktivorous and Benthivorous fish on organisms and water chemistry in Eutrophic lakes *Hydrobiologia* Vol**. 59**, I pp 9-15

Books, J.L. and S.L. Dodson, 1965. Predation, body size and composition of plankton. *Science* 150: 28-35

Carvalho, L, 1984. Influence of production by fish and water turbidity on *Daphnia gessener* population in an Amazon flood plain lake, Brazil. *Hydrobiologia*, **113**, 243 – 247.

Downing, J.A., 1984. The Assessment of secondary production. *In.* J.A. Downing & F.H. Rigler eds. *A manual of methods for Assessment of secondary productivity in freshwaters*. I.B.P. Handbook No. 17, 358 pp.

Fernando C.H. 1994. Zooplankton, fish and fisheries in tropical waters: *Hydrobiologia* 272, 105-123.

Gophen, R.E., P.B.O., Ochumba and L.S., Kaufman, 1995.Some aspects of perturbation in the structure and biodiversity of the ecosystem of Lake Victoria (East Africa). *Aquat. Living Resour*.**8**: 27-41.

Hrbacek J., 1962. Species composition and amount of zooplankton in relation to fish stock. *Rozpr Csau rada matem prir ved,* 72, 1-116. *Hydrobiologia* **208**:131-140.

Maciej Gliwicz, Z. 1994 Relative significance of direct and indirect effects of predation by planktivorous fish on zooplankton.*Hydrobiologia* **272**:201-210

Mavuti, K.M.and M.R., Litterick, 1989. Composition, distribution and ecological role of zooplankton in Lake Victoria, Kenya. *Verh. Internat. Limnol*, 24, 1117-1122.

Mwebaza – Ndawula, L.M, 1994 – Changes in Relative abundance of zooplankton in Northern Lake Victoria. *Hydrobiologia,* **272,** (1-3), pp 259-264.

Ogari, J and S. Dadzie. 1988. The food of the Nile perch *Lates niloticus* (L), after the disappearance of the *haplochrimines cichlids* in the Nyanza Gulf of Lake Victoria (Kenya). *J. F. Biol.* 32(4): 571-577.

Owili M. A, 1999. *Zooplankton- fish interaction in the littoral zone of Nyanza Gulf, Lake Victoria.* LVFRP/TECH/99/07. For Report on forth FIDAWOG workshop held in Kisumu, 16th-20th August 1999.

Owili M.A. and Omondi R., 2003. Predator- prey relationship between zooplankton and *Rastrineobola argentea* and Juvenile *Lates niloticus* in the Lake river interface habitats in Nyanza Gulf of Lake Victoria. *Environmental Ecosystem Health Society* AEMS, 2003, Canada.

Rzoska, J., 1957. Notes on the crustacean plankton of Lake Victoria. *Proc. Linn. Soc. Lond.* **168**: 116-125.

Siegfried, C.A, Bloomfield J.A. and Sutherland J.W., 1987. Acidification, vertebrate and invertebrate predators and the structure of zooplankton communities in Adirondack lakes

Wanink, J.H., 1998. The *Pelagic Cyprinid Rastrineobola argentea as a Crucial Link in the Disrupted Ecosystem of Lake Victoria*: *Dwarfs And Giants –African Adventures*. PhD Thesis, Rijksuniversiteit Leiden

Worthington, E.B., 1931. Vertical movement of freshwater macro plankton. Int.Rev.ges. *Hydrogr*. **25**:394-436

Dual role of Vembanadu Lake (Ramsar site) in Arabian Sea coastal productivity

Joseph Sebastian Paimpillil[1], ThresiammaJoseph[2] , K.K. Balachandran[2]

[1] Envirosolutions, Center for Earth Research and Environment Management, Cochin 17, India,

[2] National Institute of Oceanography, Regional Center, Cochin 18, India.

[1]Email-psjoseph@eth.net,

Abstract

The forcing of nutrient rich ground water through the narrow coastal strip between lake and sea - thickly populated and with improper sanitary conditions- into coastal Arabian Sea takes place on the attainment of sufficient water level difference between sea and lake to overcome the frictional resistance. The indications of nutrient rich water of 'external sources' spreading offshore from localized coastal pockets were visible in the recent studies. These sources had changed the oligotrophic coastal ecosystem into productive one with approximately 3 times greater primary productivity than the peak reported values. The long-term trends for nutrients in Vembanad Lake had a two-fold increase in dissolved phosphate and silicate in 42 years. The nutrients NO_3 and PO4 were present at very low levels up to mid 60s but had increased during 80's. It is estimated that the lake is receiving 42.4 * 10^3 mol d^{-1} inorganic PO_4 and 37.6 * 10^3 mol d^{-1} of inorganic nitrogen with an export of 28.2 * 10^3 mol d^{-1} inorganic PO_4 and 24* 10^3 mol d^{-1} of inorganic NO_3 to coastal waters. Major supplies of the nutrients into coastal regions come from diffuse sources rather than direct discharges. Attempts for reversing the eutrophication trends require management strategies for watersheds reaching far inland from the coastal region and restoration of wetlands and floodplains that act as nutrient traps. Drastic reductions in inflows due to climate changes, poor flushing resulting from sedimentation linked with floods or strong stratification are particularly susceptible to eutrophication of estuarine system. The increase in nutrients in the ground water flow into coastal region to be controlled by improving the living conditions and sanitary facilities of the coastal region to avoid the possibility of changes in species diversity of the region.

Key words: coastal fertilization

Introduction

Out of 20 wetlands identified in India for conservation and management, two are in the southern Kerala state of India (Sasthamkotta Lake and Vembanad-Kol Wetland). The Vembanadu Kole wetlands (Figure 1) are the largest brackish and humid tropical wetland on country's south west coast. This Lake is shrinking rapidly due to increased human activity from 315 Sq.km in 1912 to 120 sq.km and Ernakulam district accounts for a major portion of this shrinkage. The lake has lost an area around 125 sq.km between 1983 and 2003: mainly reclaimed for constructing residential apartments, roads and bridges. Average depth of lake had dropped to 3 m from 6.7 m during the last 50 years.

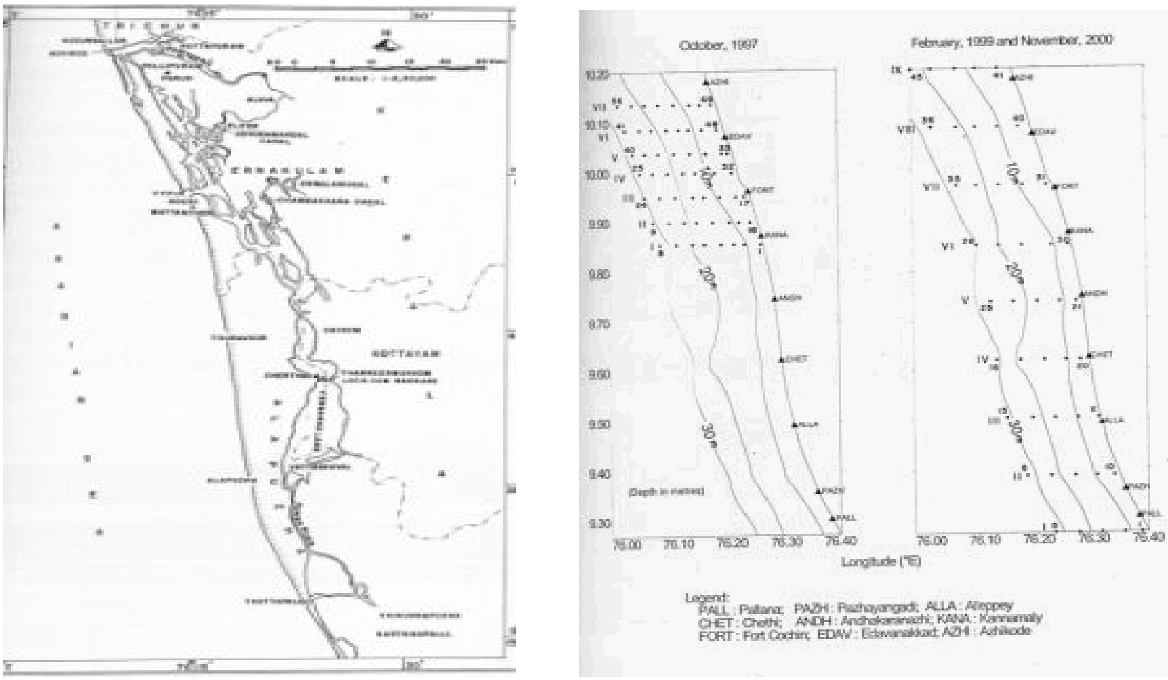


Figure 1. Vembanadu Lake and sampling transects in coastal Arabian Sea.

The man made impacts in the coastal ocean and in estuaries has increased rapidly over the last decades affecting the natural dynamic equilibrium and the biotic composition of the respective ecosystems. The main causes for such changes are introduction of untreated and partially treated sewage rich in organic substances and plant nutrients from human settlements, urban areas and certain industries, leaching of nutrients from soils and agricultural fields and animal husbandry. The booming city of Cochin has population of nearly 1.5 million (Anon, 1998) and 60% of the chemical industries of Kerala are situated in this area Cochin backwaters are the largest of its kind on the west coast of India with an area of 256 Km^2. The 16 major and several minor industries situated in the upstream region of the backwaters discharge nearly 0.105 Mm^3d^{-1} of effluents (Anon, 1996). The industrial typology includes fertilizer, pesticide, radio active mineral processing, chemical and allied industries, petroleum refining and heavy metal processing and fish processing. The fertilizer consumption in Kuttanad region (the main agricultural field draining to Cochin backwater) alone is reported to be 20,239 t y^{-1} (Anon, 1998). The backwater receives organic wastes (~ 260t d^{-1}, Anon, 1998) and an annual dredge spoil from the harbor area to the tune of 10^7 m^3. The west coast of India is environmentally more sensitive than the east coast as it is bordering one of the most sensitive ecosystems, the Arabian Sea. If there is a possible threat to the well being of the living resources of EEZ of India, then the coastal waters of southwest coast of India, and in particular, Cochin region is the prime location prone to trigger it.

Conventional understanding of coastal waters of southeastern Arabian Sea is that activation of mud banks by monsoon forcing triggers intense geochemical processes leading to high productivity. Mud banks, as they appear only during monsoon and disappear with its retrieval, are unique in their formation and functions, and have turned out to be economically important for its rich biological resources. As far as the chemical features are concerned, the general picture so far emerged out is that except during the monsoon periods, the southwest coastal waters remained oligotrophic and surface chlorophyll a typically ranges from 0.1 to 5.3 $mg.m^{-3}$, while primary productivity ranges from 100 to 360 $mgC.m^{-2}.d^{-1}$. Recent studies as the one discussed here contradict these findings and show that even after the monsoon period, fresh injection of nutrients by hitherto unknown processes fertilize the coastal waters that are either permanent or quasi-permanent in nature. One of the major mudbank regions of southwest coast of India was selected for observation that indicates episodic introduction of nutrients into the coastal waters during periods when mud banks are passive.

Results and discussion

Methods and materials

Water and sediment samples were collected from 17 stations in the vicinity of Cochin harbor area. The sampling stations covered an area extending from bar mouth up to 9 km towards the northern, the Ernakulam and Mattancherry channels. Water samples were collected using Niskin bottles from the surface, mid-depth (where the depth > 5m) and near bottom were kept packed in iceboxes and brought to the shore laboratory.

Nutrients were analyzed calorimetrically (Grasshoff et.al. 1983), and chlorophyll a by UNESCO (1966) procedure within 6 hours of collection. For trace metal analysis, extreme care was taken in sampling and sub-sampling. Water samples were collected in acid-washed polythene jerry cans and kept in iceboxes till filtration. Known volumes of samples were filtered through pre-weighted Millipore filter paper (0.45μm) and the filtrate was acidified to pH >2 using concentrated hydrochloric acid. The dissolved metals were extracted using Ammonium Pyrrollidine Dithocarbamate (APDC) and Methyl Isobutyl Ketone (MIBK) at pH 4.5 and brought back to aqueous layer by back-extraction with concentrated nitric acid and made up to 20 ml with mille-q water (Smith and Windom, 1972). The extracts were analyzed in the flame AAS (PE Analyst 100) for dissolved trace metals, Viz: Cu, Cr, Co, Fe, Mn, Ni, Pb and Zn. Sediment samples were collected at all stations with a Van Veen grab. Sediments were scooped carefully from the middle portion without being disturbed, using a clean plastic spoon, to pre-cleaned plastic containers and were kept in iceboxes and then transferred to freezer until analysis was carried out. The sample, after drying were finely powdered and I gm. Weighted accurately and digested with a mixture of HF-$HClO_4$-HNO_3, evaporating to dryness each time until the digestion was complete (Loring and Rantala, 1977), and brought into solution in 0.5 M HCl (25ml) and analyzed using flame AAS.

Results and discussion

Estuarine water quality

A comparison of the environmental parameters collected presently with the earlier data can indicate the estuarine system behavior over the years. The available data from the bar mouth region are used as several workers had sampled this region since 1965 and this is the point where the pollutants are brought down to the minimum by dilution before being exported to the coastal waters. The phosphate and nitrates were present in very low levels up to mid 70s' from where, due to the combined effect of increased industrial and agricultural activities, the levels increased during 80s' and 90s'. During 1965, the surface phosphate and nitrate were 0.75 and 2.0 μM, which has increased to 2.9 and 6 μM respectively by 2000 even though, between the years it show still higher levels. The trend also shows a build up of nitrogen and phosphorus fractions after 1975 and from 1980 onwards, the concentrations remained high. Enrichment of phosphorus with respect to nitrogen is more leading to mesotrophic waters . However, this enhanced nutrient levels have not lead to any oxygen depletion in the environment, possibly because the river discharge and tidal exchange may be sufficient enough to renew the estuarine waters and prevent deoxygenating even during the lean discharge period. But the maximum nutrient levels reported for the upstream regions were quite alarming. The build up for inorganic phosphate had started after 1973, and the subsequent increase in waste discharge had ultimately resulted in extreme levels of ammonia, phosphate and nitrate in the estuarine region. During 1980-81, the study region had nitrate and phosphate levels up to 40 and 12 µM with its upstream peaks of 108 µM and 186µM (Saraladevi, 1986). Sankaranarayan *et al.,* (1986) have reported phosphate levels up to 88µM during 1982-83, in the northern upstream stations. The present study recorded phosphate levels from 5 to 40µM for the same region. During 1990, the nutrient maximum reported from this estuarine region was 98.48 for nitrate and 15.11 µM for phosphate (Kunjikrisha Pillai, 1991). Sheeba (2000) also had reported nutrient enrichment in this system and recorded nitrate up to 451µM and phosphate up to 33 µM at the bar mouth alone.

The deterioration of environmental quality of Cochin backwaters seems to depend on several factors, all related to human interventions. One of the recent estimate shows that in spite of receiving 42.4 x10^3 mol.d^{-1} of inorganic phosphate and 37.6 x 10^3 mol.d^{-1} of inorganic nitrate from Periyar side of the estuary, the export to the coastal waters is only 28.2 x10^3 mol.d^{-1} of inorganic phosphate and 24 x10^3 mol.d^{-1} of inorganic nitrate (Hema Naik, 2000). Thus, the estuary acts as a sink for the nutrients, flushing out only a portion of the pollution load that it receives. The enhancement with respect to these nutrients in Cochin Backwaters shows the signs of eutrophication.

Heavy metals

Scientists held fast to the idea that phosphorus and nitrogen were the only nutrients vital in determining the distribution of the world's algae populations. They believed that the higher the levels of these non-metallic elements became, the more phytoplankton would grow and the greater the populations of zooplankton and fish would be. Scientists recently accept the role of trace metals such as iron or zinc in the growth of algae and zooplankton. Most common types of plankton contained a regular number of trace metals in their chemical make-up such as zinc, iron, and copper. The Trace metals in recent estuarine sediments had shown a post - monsoon enrichment of Zn, Pb, and Cu and in the northern and southern limbs of the estuary. The dissolved iron had an opposite trend to that of it in the sediments. The strong influence of fresh water modifies the sediment to leach out chromium as inferred from low values (<30µg/g) at the bar mouth. The entire region is enriched with manganese (141- 337 µg/g) with the lowest values around the bar mouth. High values of Zinc (>1000 µg/g) were noted in the east channel and low levels (~90 ug/g) at the bar mouth. The dissolved zinc had enrichment in the backwater with 116 ug/l in the year 1986 and with 879 µg/l in 1991. Nearly 80 tons of Zinc seems to have accumulated in the water body. The copper content in sediment ranged between 5 - 53 µg/g with high values in the northern limb and the dissolved copper had a range of 1 -3 µg/l. The distribution of Nickel and cobalt were similar and a concentration of Ni (0.60 µg/g) indicates the absence of nickel pollution. Significant correlation of iron with other metals (except Mn) indicates that elemental accumulation in sediments may be controlled by precipitation of iron on to organic matrix. The significant correlation between metals (except Mn) shows a common source of metals. Natural processes control the distribution of most metals, while Zn is influenced more by anthropogenic input. Cochin bar mouth and harbor region was not enriched in metals to greater levels. The northern part, an enrichment of metals, especially Zn is evident. Absence of build up in harbor and bar mouth may be due to periodic dredging and removal of recent sediment deposit.

Ground water flux and potential of coastal productivity bloom

The quality of coastal waters is coupled closely to the drainage of uplands. Primary attention has been given to river hydrography, but recent evidence shows that other transport mechanisms, particularly the discharge of groundwater, are important in areas covered by unconsolidated sediment or lime shell beds. Human activities on coastal watersheds provide the major sources of nutrients entering shallow coastal ecosystems. The land-use pattern in the two adjacent coastal districts had 287 households/km^2 . The human population along the coastal belt with more than 70 % of households without proper sanitation facilities has resulted in concomitant increases in widespread use of septic tanks and nutrient inputs to coastal waters, particularly from regions occupying limestone beds. The ground water quality of the region had shown nitrate in sediment extract up to 12 µM, ammonia (in water) 8µM, urea (in water) 14 µM, urea (sediment extract) 15 µM (Lizen Mathews, 2000).

As far as the chemical features of the coastal region are concerned, the general picture so far emerged out is that except during the monsoon periods, the southwest coastal waters remained oligotrophic and surface chlorophyll a typically ranges from 0.1 to 5.3 mg.m^{-3}, while primary productivity ranges from 100 to 360 mgC.m^{-2}.d^{-1} .The present study in the post monsoon season had revealed highest value of 14 mg/m3 for chlorophyll a, approximately 3 times greater than the peak values reported so far from these waters. A band of N/P > 15 funneling out from coastal region provided an indication of 'external source' of nitrogenous compounds into the coastal waters. The long-term (decadal) trend of chlorophyll showed a "greening" of the near-shore waters. The present investigation represents the period when the mud banks were not activated and the results showed fertilization of the coastal water by injection of nutrients by hitherto unknown processes. The high nitrate-N, ammonia concentrations, enriched particulate organic carbon (> 3.5 mg/l) and Chlorophyll a (14.8 mg/m3) at localized coastal regions indicated a clear near shore nutrient source. It is difficult to point out a definite source for the high nutrient values, as the fresh water discharge was at the minimum. These sources of nutrients deserve identification as it was traced in a region far away from any river mouth and the injection of nutrients was observed during non-monsoon months when mud banks were passive.

Phytoplankton productivity boosting in southwest coastal Arabian Sea is partially attributed to the possibility of nutrient rich ground water discharges to coastal sea through the narrow strip of porous lime shell bed separating the Vemabadu Lake and the sea. The necessary forcing for the ground water flow is gained when the fresh water level in lake and the sea level reached a critical value. With regard to the hydraulic mechanism, it would appear that apart from the trending faults and water level variations in the lake, existence of several passages depending on the thickness of the lime shell bed also contribute to the sub aqueous flow. It is likely that the subterranean flows during the present observations were very weak (ooze), it can be expected to be much larger during the southwest monsoon period, when the sea level is at its annual minimum and the water level of the lake, approximately 5 feet above than normal (Lapointe, B.E. & Clark, M.W., 1992) due to the increased river discharge. A head of 5 ft. of water exerting a pressure of 2 lb/sq.inch may be insufficient to push suspended solids (Du Cane, 1968), but is good enough to erode frictional resistance and force fresh water flow into the coastal region. It is ultimately this flow of fresh water through these passages that stratifies the water column and diverts the converging waves to unsettle the sediment.

The periodicity of groundwater flows depend on the critical rise in the water level of the Vembanadu Lake, which depends on monsoon variability and flash floods associated with severe cyclones. If sufficient critical fresh ground water flow required to induce stratification in the coastal waters is available, the nutrient rich flow can induce high primary coastal productivity. The possibility of heavy rains and flash floods linked with cyclones are high with ongoing climate variability, such critical conditions can occur during other seasons and can shift to similar locations in the coastal region. Any noticeable change of the current oligothrophic nature of the coastal region can contribute to the removal of atmospheric carbon through photosynthesis and later by plank tonic grazing and sinking. The phytoplankton blooms can alter the food chain and the impact of climate variability can slowly adjusts the overall coastal ecosystem.

Seasonal ground water flux

During a typical pre-monsoon (February) month, the nitrogenous nutrients remained low except for the southern transects centered on Chethi and Alleppey. The phosphate concentrations did not show any spatial or vertical variation in the water column, but higher concentrations of ammonia, nitrate and silicate were observed at selected regions starting in the near shore regions and extending offshore . The Nitrate-N concentrations point towards a clear source between Chethi and Pazhayangadi, where it peaked up to > 8 μM and decreased towards offshore. A similar trend was observed for ammonia-N with the source centered on Chethi (at about 15 m depth). It may be assumed that the ammonia released were either rapidly utilized by phytoplankton or oxidized within the system itself where the waters were saturated with dissolved oxygen. Distribution of silicate-Si was similar to that of nitrate (4 – 10 μM), higher than the corresponding values reported for the waters of Southeastern Arabian Sea. The input of these nutrients supported high primary production up to 14 mg/m^3 of chlorophyll a (peak column production of 1529 $mgCm^{-2}d^{-1}$), approximately 3 times greater than the peak values reported so far from these waters (Qasim *et al.,* 1978). The peaks in chlorophyll a and ammonia showed a preference of ammonia among the nutrients for primary production. It is difficult to point out a definite source to these high nutrients during this period, as the fresh water discharge was at the minimum.

During the post monsoon (November), homogenous mixed layer prevailed in the entire region. While the physical characteristics were more or less stable, there was considerable variability in the nutrients and in chlorophyll a concentration. A marked decrease in sub-surface dissolved oxygen (2.8 - 4.8 ml/l) was the characteristic feature of this period, which was concomitant with enriched nitrite (0.5 – 2.0 μM), phosphate (0.4 – 2.8 μM) and silicate (0.5 – 14 μM). The ammonia (1 – 7 μM) and nitrate (1 – 6 μM) were also elevated at some regions along southern transects. The enriched particulate organic carbon (> 3.5 mg/l) and Chlorophyll a (14.8 mg/m^3) were also the notable features of this period. It is likely that chlorophyll a values were proportionate to carbon production indicating a strong positive relationship binding it with nutrient related factors rather than seasonal or diurnal fluctuation. The elevated nitrite and phosphate levels around Cochin may be due to the input from the backwaters. Higher values of nitrite, POC and chlorophyll a towards the southern offshore waters off Pallana were conspicuous and the regions with high nitrite had nitrate levels up to 6 μM and the low levels of ammonia had ruled out the nitrification as a significant process responsible for nitrite accumulation. The remarkable co-existence of nitrite with nitrate strongly suggested that the nitrite production should mostly be due to assimilatory reduction. This was further substantiated by the high concentration of chlorophyll a (4 – 9.8 mg/m^3) on these transects.

The N/P ratio in the coastal waters was below 15 during November, possibly due to the disproportionate release of P from mudbank sediment. However, a band of N/P > 15 funneling out from Alleppey region was indicative of an 'external source' of nitrogenous compounds into the coastal waters. A comparison of long-term (decadal) trend in the chlorophyll data of this region showed "greening" of near shore waters (Devassy, V. P, 1983). This suggests that phytoplankton standing crops had increased historically, possibly in response to watershed nutrient inputs. These sources of nutrients deserve identification as it was traced to a region, far away from any river mouths.

The current observations in general indicated the presence of a nutrient source between Chethi and Pallana. This region has mud banks but the release of nitrogenous compounds cannot be accounted from sediments. The injection of nutrients was in non-monsoon months when mud banks were passive and a new influence of Vembanadu Lake on the coastal waters is very clear. One of the recent estimate shows that in spite of receiving 42.4 x10^3 mol.d^{-1} of inorganic phosphate and 37.6 x 10^3 mol.d^{-1} of inorganic nitrate from Periyar side of the estuary, the export to the coastal waters is only 28.2 x10^3 mol.d^{-1} of inorganic phosphate and 24 x10^3 mol.d^{-1} of inorganic nitrate (Hema Naik, 2000) and the lake acts as a sink for the nutrients, flushing out only a portion of the pollution load that it receives

Dual role of Vembanadu Lake

The coastal Arabian Sea ecosystem is plagued by submarine ground water discharges and runoff of from the paddy fields at the southern extend of Vembanadu Lake. The fertilizer runoff from big farms through wetlands can trigger sudden explosions of marine algae capable of disrupting ocean ecosystems and even producing "dead zones" in the sea. Many harmful blooms are artificially fueled by fertilizer runoff from farms, which dump tons of excess nitrogen into rivers that eventually flow into the sea. Addition of these human-caused nutrients in the coastal region causes extra blooms of phytoplankton. Each fertilization and irrigation event may trigger a noticeable phytoplankton bloom near to the estuary mouth, right after each irrigation events, within a matter of days. Common type of marine algae prefers urea, to inorganic fare such as ammonium and nitrate that occurs naturally in the ocean. Although urea as a source of pollution is generally ignored, research shows that urea represents an average of one-third of the total nitrogen uptake supporting growth of phytoplankton in regions where red tides can occur. A series of images from an orbiting satellite equipped with special light sensitive instruments may be required to detect phytoplankton blooms floating near the surface of the sea in order to identify the possibility of a one-to-one correspondence between an irrigation event and a massive algal bloom. The coastal eddies actually pull the plumes along the coastal region.

Conclusions

The introduction of nutrients into Vembanadu Lake and the ground water fluxes through porous coastal strip between wetland and coastal ocean seem to induce large variability in the coastal water quality, primary productivity. Major supplies of the nutrients into coastal regions come from diffuse sources rather than direct discharges. Continuous nutrient entry through such process is bound to upset coastal water productivity pattern. A sub aqueous injection of nutrients into the coastal waters through this region is possible even after the rainy season. This assumption need further study to establish cause and effect mechanisms and quantify actual trends created by increased nutrient loading.

Attempts for reversing the trends leading to eutrophication require management strategies for watersheds reaching far inland from the coastal region and restoration of wetlands and floodplains that act as nutrient traps. Drastic reductions in inflows due to climate changes, poor flushing resulting from sedimentation linked with floods or strong stratification are particularly susceptible to eutrophication of estuarine system. The occurrence of red tides and fish mortality off the Kerala coast may be much more frequent for flooding triggered by increase of tropical cyclones, El Niño effect or intensification of monsoons. The increase in nutrient rich ground water flow through the narrow strip into coastal region to be controlled by improving the living conditions and sanitary facilities of the thickly populated coastal region to reduce the possibility of slow changes in species diversity of the region.

References

Anonymous, 1996. *Pollution potential of industries in coastal areas of India. Coastal Pollution Control Series*. Central Pollution Control Board Report. COPOCS/9/1995-96.

Anonymous, 1998. *NEERI- carrying capacity based developmental planning of Greater Kochi Region*. Phase I Report.

Devassy, V. P., 1983. Mahasagar, Bull. Bull. Nat. Inst. *Oceanogr.* 7: 101-105.

Du Cane, 1968. Report of the special committee on the movement of mudbanks, Cochin Govt.Press, 57.

Grasshoff, K.. Ehrhardt, M & Kremling, K. eds. 1983. *Methods of seawater analysis.* Verlag Chemie, Weinheim.

Hema Naik, 2000. *Budgets for Periyar estuary, Kerala. Presented at "Regional Training Workshop on Biogeochemical Budgeting and Socio-Economic modeling for Coastal Scientist"*. APN/SASCOM/LOICZ, 18-22 September, Colombo.

Kunjikrisha Pillai V., 1991. *Studies on the hydrobiology and pollution of the Vembanadu Lake and adjacent waters.* Ph.D. Thesis. Cochin University of Science and Technology, Cochin, 148.

Lapointe, B.E. & Clark, M.W. 1992. Nutrient inputs from the watershed and coastal eutrophication in the Floriday Keys. *Estuaries.* 15(4), 465-476.

Loring D.H. and Rantala R. T. T., 1977. *Geochemical analysis of marine sediments and suspended particulate matter.* Fisheries and Marine Service Technical Report No. 700, Environment, Canada.

Lizen Mathews, 2000. *Studies on the role of sediments on the nutrient dynamics and fertility of Kuttanad waters.* Ph.D Thesis, CUSAT, 103 p.

Sankaranarayan V.N., Udaya Varma P., Balachandran K.K., Pylee A. and Joseph T., 1986. Estuarine characteristics of the lower reaches of the river Periyar (Cochin backwaters). I*ndian J. Mar. Sci*. 15:166-170.

Saraladevi K., 1986 *Effects of industrial pollution on the bentic communities of a tropical estuary.* Ph.D. Thesis. Cochin University of Science and Technology, Cochin:. 380.

Sheeba P., 2000. *Distribution of benthic infauna in the Cochin backwaters in relation to environmental parameters.* Ph.D. thesis. Cochin University of Science & Technology, Cochin. 241.

Smith R.G and Windom H. L. 1972. *Analytical handbook for the determination of As, Cd, Co, Cu, Fe, Pb, Mn, Ni, Hg and Zn in the marine environment*. Georgia Marine Science Center, Tech. Rep: 72-76.

UNESCO. Determination of photosynthetic pigments in seawater, 1966. *In: Monographs on Oceanographic methodology* - 1

Qasim, S. Z., Wafar, M. V. M., Sumithra Vijayaraghavan, Joseph, P., Royan. And Krishna Kumari, L., 1978. *Ind. J. Mar. Sci*., 7: 84-93.

Macroinvertebrate assemblages of the inflow, the outflow and the upper littoral zone of the prealpine Lake Bohinj, Slovenia

Mihael J. Toman, Gorazd Urbani and Branka Tavzes

University of Ljubljana, Biotechnical Faculty, Department of Biology, Vena pot 111, SI – 1000 Ljubljana, Slovenia

Email: mihael.toman@bf.uni-lj.si

Abstract

Lake Bohinj, oligotrophic dimictic glacial water body, is located in the alpine area of Slovenia, with an elevation of 530 m above sea level, maximum depth of 45 m, retention time 4 months and 11.35 km of diverse upper littoral zone. Lake's turbulent inflow stream Savica (5.0 m^3/s) with more or less constant temperature during year is a habitat of diverse macroinvertebrate community and has decisive effect in lake summer stratification. Mainly stony littoral zone and the macrophyte area provide microhabitats for different macroinvertebrate assemblages. The outflow Jezernica (8.34 m^3/s) is characterised by high temperatures in summer and drift of organic matter from epilimnium strata.

Macroinvertebrate assemblages, both in lake and running waters were studied with the special reference to the substrate over a four year period from 2001 to 2004. In the littoral, four sampling sites with different substrates (stones, pebbles, submerged macrophytes and emergent macrophytes) were selected whereas in the rivers mainly stony substrate was present. In addition, several ecological factors were measured and analysed, including water temperature and temperature profile in lake, oxygen saturation and concentration, concentration of nutrients, pH and conductivity, suspended solids, Secchi depth and chlorophyll a concentrations in the lake. Taxa composition of macroinvertebrate assemblages in the littoral zone and in the river bed were studied. Detrended correspondence analysis (DCA) was used to compare sampling sites. Results of the DCA analysis showed that in the inflow and the outflow of the lake more similar assemblages were found compared to the sites in the littoral zone. From year to year the variability in the assemblages was also much lower at lotic sites than at different substrates of the littoral zone. Taxa richness of most sites varied considerably during those four years, but at the sites with pebble substrate variation and number of taxa were the lowest. In addition also Shannon-Wiener and Simpson's diversity indices and evenness were calculated. The lowest values were observed at the inflow, but at different littoral sites similar average values were calculated, but values varied considerably from year to year.

Key words: glacial lake, littoral zone, macroinvertebrates

Introduction

The understanding of lake ecosystem is incomplete if there is no estimation of its catchment area, ecology of inflow and outflow and mainly the littoral area as the interface buffer zone (Wetzel 2001). Not only the vegetation as the most productive component, but also the macroinvertebrate community has important influence on matter transport and energy budget in the lake ecosystem (Jonasson *et al.,* 1990) and could be used in bio-assessment models (Broders K.P. *et al.*, 1998). The present study examined macroinvertebrate assemblages both in lake upper littoral zone and in the inflow and outflow riverbed and represent the first data concerning Lake Bohinj and its system of running waters.

Lake Bohinj - located in Triglav National Park at 46°16'N, 13°48'E, at an altitude of 530 m a.s.l - is the largest and deepest prealpine glacial lake in western part of Slovenia. From the geological point of view the catchment area is comprised mainly of Triassic and Jurassic carbonate rocks. The lake is 4.35 km long and 1.25 km wide with a volume of water mass of 100 x 10^6 m^3, maximum depth of 45 m and the water retention time between 3 and 4 months. Littoral area with a steep slope has diverse substrate, stones and boulders in the north side near inflow mouth, part of muddy substrate in the west, submerged macrophytes in the south and patchiness of emergent macrophytes. Number of macrophyte species is more or less constant over a period of years, but with evident change in their abundance and distribution. The dominant species is *Myriophyllum spicatum* (Urbanc-Bercic 1995), the greatest species diversity is among *Potamogeton* genus (Urbanc-Bercic & Gaberscik, 1996). Nutrient loading is very low therefore the lake is in oligotrophic condition.

The main inflow stream Savica carries fresh water with constant annual temperature between 6 and 8 °C. The study section was 25-30 m wide with a max depth in riffle area of 0.5 m. Stones, pebbles and gravels dominate in river bed, a few boulders were covered with water mosses. Due to turbulence stream water was well oxygenated (100%), with low conductivity (125-170 µS/cm), alkaline pH (8.2-8.5), very low nitrate (mean value 2.0 mg/L) and orthophosphate (mean value 0.04 mg/L) loading. Flow and current velocity depend on precipitation intensity in catchment area.

The outflow stream Jezernica is warmer during summer (18-23°C), carrying epilimnium water. Mezo- and macrolital dominate in river in the riffle area under study. During our investigations the river was 28-30 m wide with a max depth of 0.8 m, pH 8.1-8.5, low conductivity (170-200 µS/cm), low nitrate (1.2-1.6 mg/L) and orthophosphate (0.01-0.02 mg/L) concentrations. Periphyton community, mainly diatom community was well developed.

Materials and methods

Four different substrates (emergent macrophytes, submersed macrophytes, stones and pebbles) in the lake littoral were sampled, whereas in inflow as well as in outflow one sampling site with mainly stony substrate was chosen. Samples of macroinvertebrates were collected in the summer period from 2001 to 2004. Hand-net with 0.5 mm mesh-size was used by kicking 3 minutes. Animals from littoral were also removed by hand from stones and submersed macrophytes. All samples were preserved in 4% formalin, animals sorted and identified to the highest possible level. Each level of identification was considered as one taxon.

Several abiotic factors were measured in the field, e.g. oxygen concentration, oxygen saturation, pH, conductivity, Secchi depth, others were analysed in the laboratory e.g. concentrations of nutrients, suspended solids and chlorophyll *a*.

Results from sampling sites were compared using Detrended Correspondence Analysis (DCA). Four metrics were calculated i.e. taxa richness, evenness, Shannon-Wiener and Simpson diversity index. All statistical analyses were provided by the programme PC-ORD (McCune & Mefford 1999).

Results and discussion

All together, 104 macroinvertebrate taxa were recorded. Taxa richness of three littoral sites (EM, SM, SL) varied considerably during four year sampling, only at the site with pebbles (PS) variation and number of taxa were low (Figure 1a).

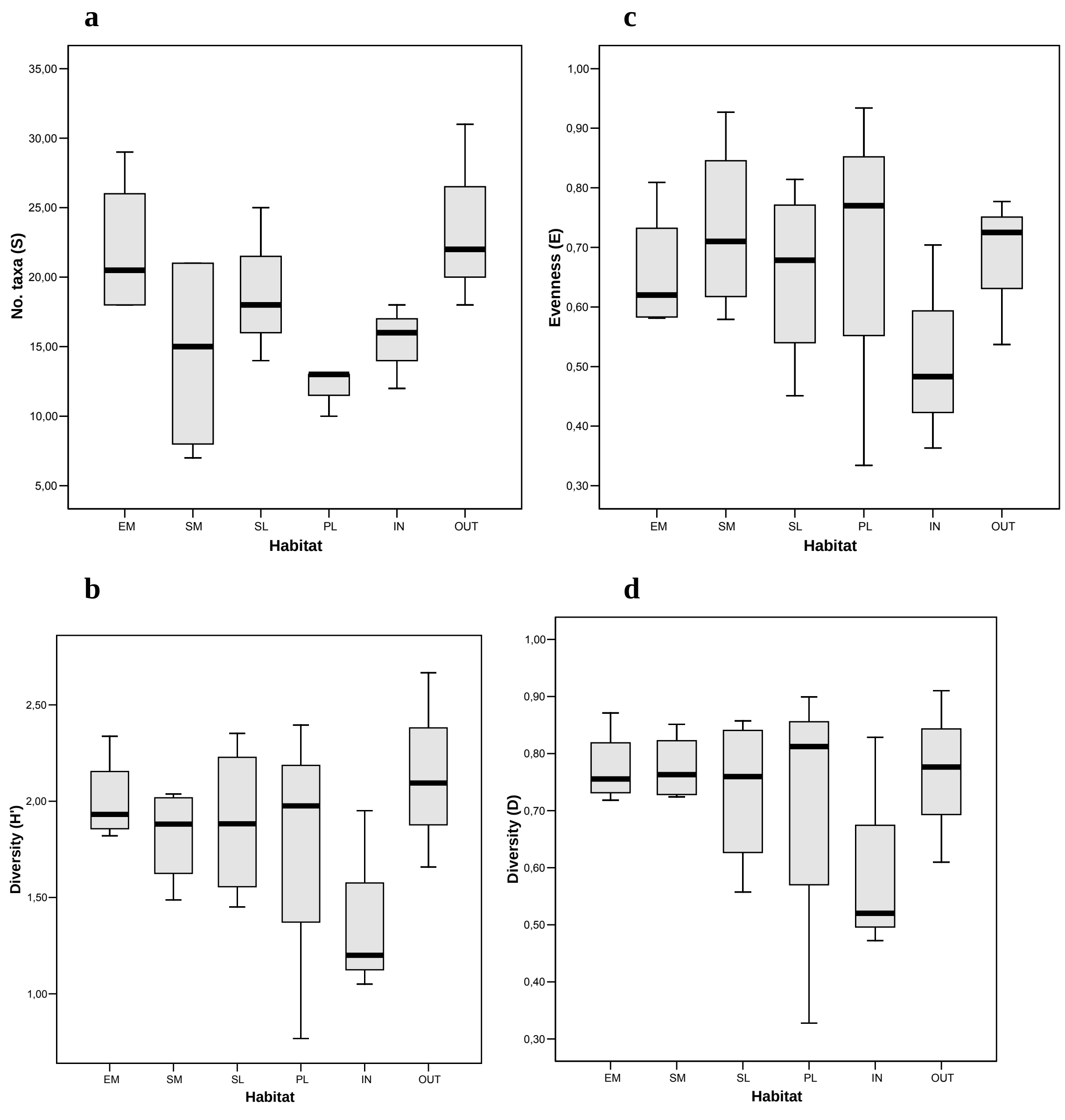


Figure 1. Absolute deviations of the metrics a) No. of taxa (S), b) Shannon-Wiener diversity index (H'), c) Evenness (E), d) Simpson's diversity index (D) for four littoral sampling sites (EM-emergent macrophytes, SM- submersed macrophytes, SL-stones, PL-pebbles), inflow (IN) and outflow (OUT).

Low number of taxa was recorded also at inflow, whereas the number of taxa recorded at outflow was similar to the three littoral sites. Shannon-Wiener and Simpson's diversity index and evenness as well confirmed the results of taxa richness (Figure 1b,c,d). However, the diversity and evenness at pebble substrate in the lake littoral are high and comparable with other littoral substrates. At different littoral sites and at the lake outflow similar average values were calculated, but values varied considerably from year to year. On the other hand, evidently low diversity and evenness at the inflow are probably due to the constant low water temperature.

Results of the DCA analysis showed that in the inflow and the outflow of the lake more similar assemblages were found comparing to the sites in the littoral zone (Figure 2).

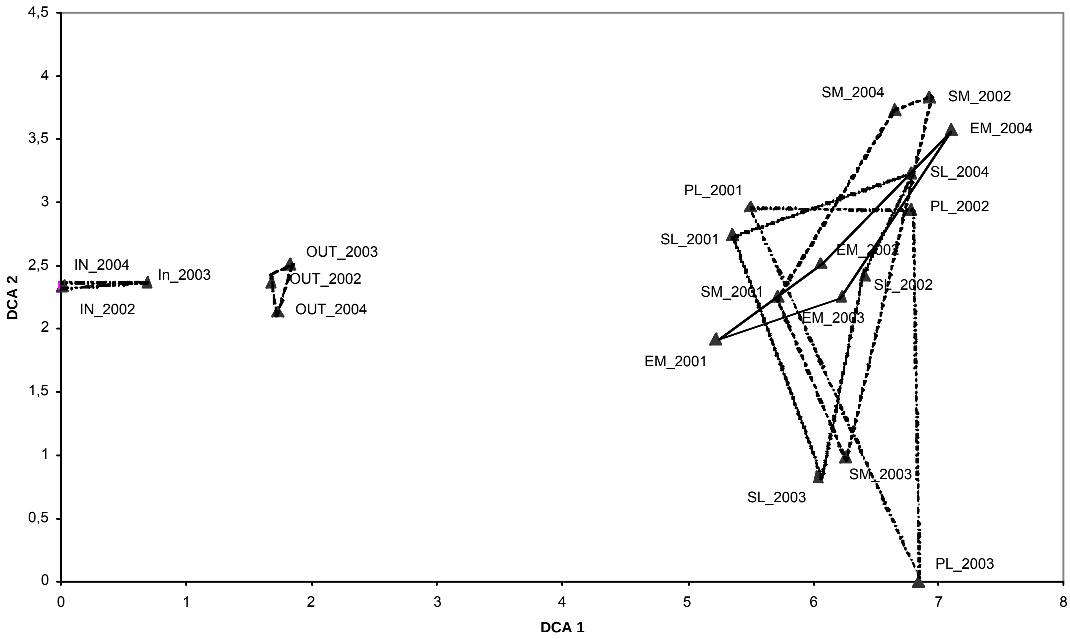


Figure 2. DCA ordination diagram with littoral sampling sites (EM-emergent macrophytes, SM- submersed macrophytes, SL-stones, PL-pebbles), inflow (IN) and outflow (OUT) collected from 2001 to 2004.

There were no obvious differences in the macroinvertebrate assemblages structure between substrates in the lake littoral (Figure 3). However, year to year variability in different littoral assemblages was high, much higher than at the lotic environment. The reason is probably due to the fact that headwater streams (in our case lake inflow) and lake outflows are the most stable freshwater environments (Giller & Malmquist, 1998).

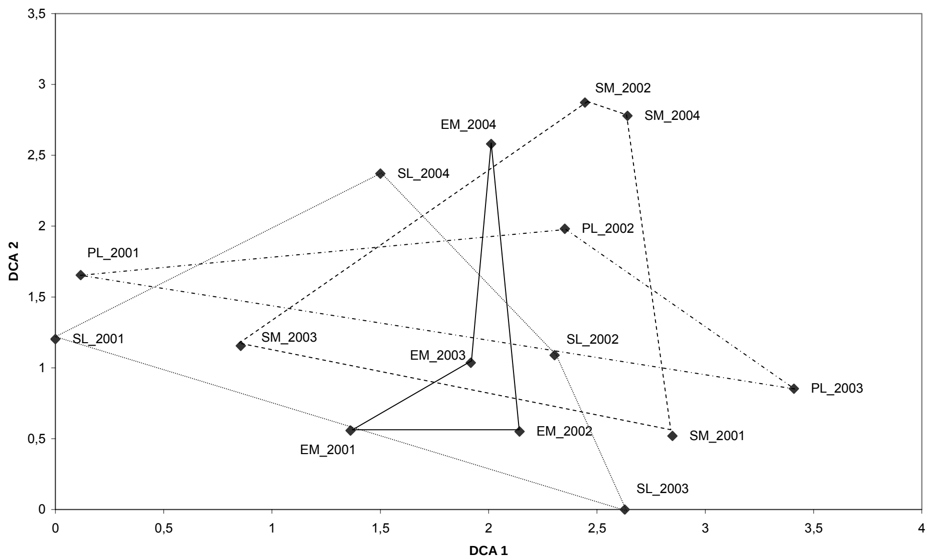


Figure 3. DCA ordination diagram with littoral sampling sites (EM-emergent macrophytes, SM- submersed macrophytes, SL-stones, PL-pebbles) collected from 2001 to 2004.

References

Brodersen K.P., Dall P.C. & Lindegaard C.,1998. The fauna in the upper stony littoral of Danish lakes: macroinvertebrates as trophic indicators. *Freshwater Biology* 39, 577-592.

Giller P. S. & Malmquist B., 1998. *The biology of streams and rivers*. Oxford University Press, Oxford.

Jonasson P.M., Lindegaard C. & Hamburger K., 1990. Energy budget of Lake Esrom, Denmark. *Verh.Internat.Verein.Limnol.* 24, 632-640.

McCune B. & Mefford M. J., 1999. *Multivariate Analyses of Ecological data, Version 4.* MjM Softwarer Design, Gleneden Beach, Oregon, USA.

Urbanc-Bercic O., 1995. Aquatic vegetation in two pre-alpine lakes of different trophic levels (Lake Bled and Lake Bohinj): vegetation development from the aspect of bioindication. *Acta bot. Galica* 142, 563-570.

Urbanc-Bercic O., Gaberscik A., 1996. *The change of aquatic vegetation in lake Bohinj from 1986 to 1996*. In: Proceedings of the Int. Workshop and 8th Macrophyte Group Meeting IAD-SIL, pp. 69-72, Ljubljana, Slovenia.

Wetzel R.G., 2001. *Limnology, Lake and River Ecosystem*. Academic Press, San Diego.

Reservoirs of Sri Lanka: A major source of animal protein for rural communities

Upali S. Amarasinghe

Department of Zoology, University of Kelaniya, Kelaniya 11600, Sri Lanka

E-mail: zoousa@kln.ac.lk

Abstract

In Sri Lanka (64,652 km^2), there are over 200 large (750 – 7,793 ha) and medium-sized (250 – 750 ha) reservoirs with a cumulative extent of over 130,000 ha, which support capture fisheries. In addition, there are over 15,000 small (<50 ha) village reservoirs with a total extent of about 39,000 ha. The inland fishery of Sri Lanka is at present essentially a capture fishery in perennial reservoirs. The reservoir capture fishery is a relatively recent development after introduction of the exotic cichlid species, Oreochromis mossambicus into Sri Lankan freshwaters in 1952. When total extent of reservoirs, which support commercial fisheries are considered, with the mean annual fish yield of 122 kg ha^{-1} in Sri Lankan perennial reservoirs, the total annual inland capture fisheries production is estimated to be around 9,600 tonnes.

Through an effective management strategy, which involves a mechanism for social mobilization, culture-based fisheries can be developed in small, village reservoirs. Recent studies indicate that an average fish yield of about 450 kg ha^{-1} can be achieved during a single culture cycle within a year from the culture-based fisheries in these village reservoirs. As there are concerted efforts to develop culture-based fisheries, at least 10% of the total extent of village reservoirs (about 9,000 ha) can be stocked annually with fish fingerlings to enhance inland fisheries production. As inland fishery is a source of relatively cheap animal protein for rural communities, future prospects of this sector for food security need to be properly understood to give a high priority for inland fisheries research and development in national development plans.

Key words: Reservoir fisheries, food security; culture-based fisheries

Introduction

Reservoir construction can be considered as one of the major anthropogenic factors making radical changes in the landscape of the planet earth. Although this has been a recent phenomenon in the global scale, in some countries like Sri Lanka, this enterprise dates back to even the period of written history (De Silva 1988). In Sri Lanka, two broad categories of reservoirs are found. They are, large perennial reservoirs (200 - 6300 ha), which have been constructed by damming rivers the total cumulative extent of which is about 130,000 ha and over 15,000 small (<50 ha) village reservoirs with a total extent of over 95,000 ha (Anon. 2000) most of which are non-perennial. The reservoir density in Sri Lanka (2.6 ha for every km^2 of island) is the highest density of reservoirs in the world. Sri Lanka does not possess natural lakes.

As recorded in stone inscriptions, there were subsistence scale fisheries in reservoirs and irrigation canals to exploit indigenous species for domestic consumption even during the reigns of ancient kings (Siriweera, 1994). Nevertheless, the commercial scale inland fishery in Sri Lanka was established after the introduction of exotic cichlid species, *Oreochromis mossambicus* into Sri Lanka reservoirs in 1952 (De Silva 1988). The trends in the growth of the inland fishery of Sri Lanka have been well documented and most detailed studies of fisheries in tropical reservoirs or lakes outside Africa have been reported in Sri Lanka (e.g., Fernando & Indrasena, 1969; De Silva, 1988; Amarasinghe, 1998). In this paper, an attempt is made to appraise the potential role of reservoirs as a source of animal protein for rural communities with a view to identifying appropriate technology for achieving high yields.

Materials and methods

Cumulative extents of perennial reservoirs with commercial fisheries in different administrative districts were estimated from the unpublished records available at the National Aquaculture Authority of Sri Lanka. Cumulative extents of small village reservoirs in these administrative districts were obtained from the published information (Anon, 2000). Mean fish yield in perennial reservoirs, estimated from surveys of commercial fisheries at least for 20 days per month over a period of two years (Amarasinghe *et al*., 2002), were used to compute reservoir capture fisheries production in each administrative district.

It has been shown that small village reservoirs most of which retain water from the intermonsoonal rainy season in November – December to peak dry season in August – October, can be used to develop culture-based fisheries through stocking and recapture of Chinese and Indian major carps after growth period of 7 - 9 months (De Silva, 1988). The total extent of these non-perennial reservoirs is about 39,000 ha (Mendis, 1977). In addition, there are small perennial reservoirs where the commercial fisheries are not practised the cumulative extent of which is estimated to be over 56,000 ha (Anon. 2000). Considering technical and socioeconomic constraints (Jarchau *et al*., 2005; De Silva *et al*., 2004) for culture-based fisheries development in small village reservoirs (e.g., fingerling scarcity, low biological productivity, wildlife reserves, level of willingness of rural communities to get involved in culture-based fisheries), a conservative estimate of 10% of the available extent of small reservoirs was considered for determining potential for culture-based fisheries production. Mean fish yield of culture-based fisheries trials in 22 small reservoirs of Sri Lanka estimated by Wijenayake *et al*., (2005), was used to compute fish production potential in small village reservoirs. For determination of fingerling requirement for stocking non-perennial reservoirs, relationship between stocking density and fish yield was computed from the data reported by Wijenayake *et al*., (2005).

Data on per capita consumption of marine and freshwater fish in different districts were gleaned from the URL www.ststistics.gov.lk. Based on this information, status and the future trends in the reservoir fishery of Sri Lanka, as a source of providing animal protein for rural communities were evaluated and strategies were suggested for achieving the maximum potential.

Results

Mean fish yield of capture fisheries in perennial reservoirs of Sri Lanka, based on the data presented by Amarasinghe *et al*., (2002), is estimated to be 122.3 kg ha^{-1} yr^{-1}. District-wise annual fish production, estimated on the basis of the cumulative extents of reservoirs with commercial fisheries in individual districts of Sri Lanka (Source: National Aquaculture Development Authority of Sri Lanka), is shown in Figure 1. Accordingly total annual capture fisheries production from major reservoirs is estimated to be about 9,646 metric tons. Per capita consumption of fish in individual districts (Figure 2) shows that in the districts with high freshwater fish production from perennial reservoirs, freshwater fish consumption is higher than in other districts. According to Figure 2, it is also evident that in the administrative districts situated in the central part of the country (e.g., Kandy, Nuwara Eliya), per capita fish consumption is very low. In the coastal districts (Colombo, Gampaha, Puttalam, Galle, Matara etc.), per capita consumption of marine fish is high. Relatively high per capita fish consumption with a significant proportion of freshwater fish is reported for Anuradhapura and Polonnaruwa districts where the total extents of large perennial reservoirs are high. There is a highly significant relationship ($p < 0.001$) between cumulative reservoir area and per capita freshwater fish consumption (Figure 3a). On the other hand, no significant relationship is evident between cumulative reservoir area and per capita marine fish consumption (Figure 3b) indicating that the reservoirs play a significant role in providing fish protein to the people in the inland districts.

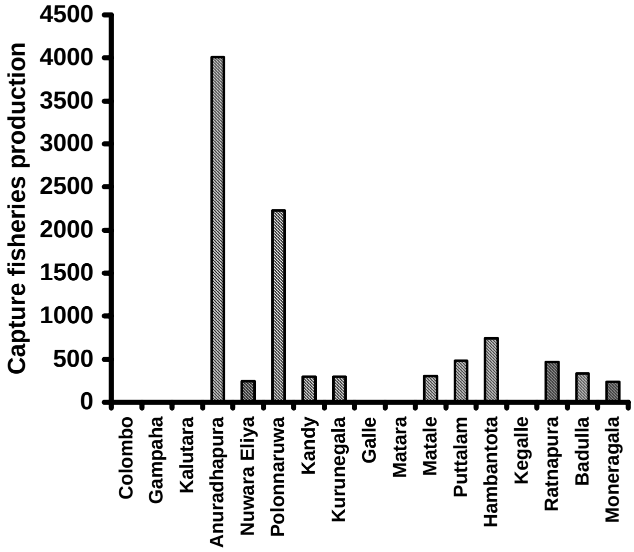


Figure 1. District-wise annual freshwater fish production (metric tons), estimated on the basis of the cumulative extents of reservoirs with commercial fisheries in individual districts of Sri Lanka.

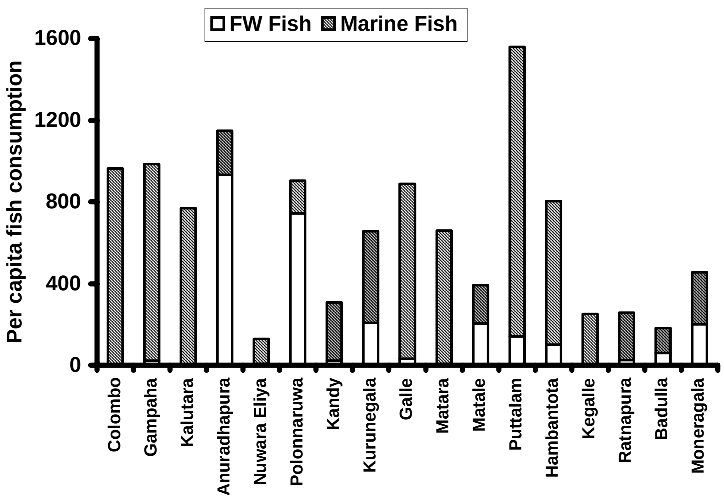
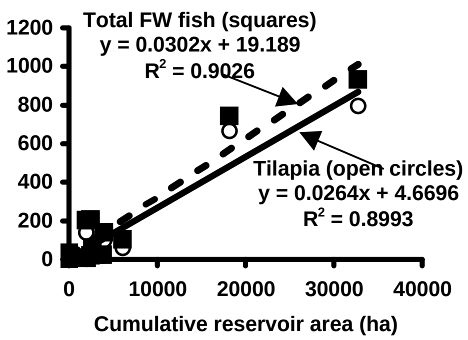
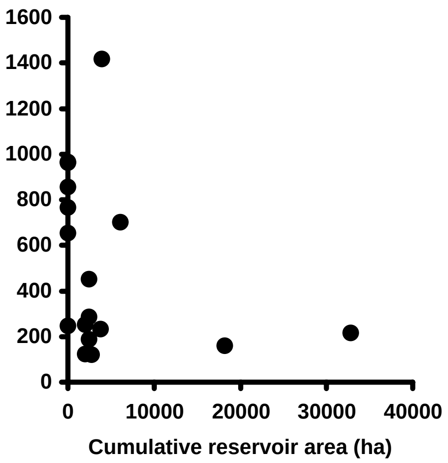


Figure 2. Per capita consumption of freshwater and marine fish (g/month) in individual districts in Sri Lanka (Source: www.statistics.gov.lk).



(a)



(b)

Figure 3. Relationships between (a) cumulative reservoir area and per capita freshwater fish consumption (tilapia and all freshwater fish); (b) cumulative reservoir area and per capita marine fish consumption, in different districts of Sri Lanka.

Culture-based fisheries potential in small village reservoirs in each district, estimated on the basis of mean yield of 22 trials in such reservoirs (449.8 kg ha^{-1} yr^{-1}) assuming that 10% of total reservoir area in each district can be utilized for the development of culture-based fisheries, is given in Table 1. As high densities of small village reservoirs are found in inland areas of the dry zone of the country (e.g., Anuradhapura, Kurunegala, Vavunia), this reservoir resource is a promising means for providing animal protein for rural communities.

Table 1. Cumulative extents of small reservoirs in different administrative districts (Source: Anon. 2000) and their culture based fisheries potential (CBFP) as estimated for 10% of total reservoir extent.

District	Total reservoir extent (ha)	CBFP (metric tons/year)
Ampara	1,800.7	81.0
Anuradhapura	33,373.5	1,501.1
Badulla	148.6	6.7
Batticaloa	3,723.2	167.5
Colombo	0	0
Galle	0	0
Gampaha	8.5	0.4
Hambantota	744.3	33.5
Kalutara	0	0
Kandy	100.9	4.5
Kegalle	7.3	0.3
Kurunegala	19,250.4	865.9
Mannar	2,189.4	98.5
Matale	2,736.2	123.1
Matara	48.3	2.2
Moneragala	2,021.8	90.9
Mulathivu	3,705.7	166.7
Nuwara Eliya	73.1	3.3
Polonnaruwa	1,720.5	77.4
Puttalam	6,282.8	282.6
Ratnapura	251.8	11.3
Trincomalee	3,936.2	177.1
Vavunia	13,266.6	596.7
Jaffna	0	0

The relationship between stocking density and culture-based fisheries yield, as determined from the data presented by Wijenayake *et al*., (2005), is shown in Figure 4. This indicates that the optimal stocking density of fish fingerlings which produces the highest culture-based fisheries yield in small village reservoirs of Sri Lanka is about 2,400 fingerlings per ha. Fish fingerlings should be stocked in small village reservoirs during November – January period for the success of this aquaculture strategy, as indicated in the calendar of activities (Figure 5). For the utilization of about 9,500 ha of small village reservoirs to develop culture-based fisheries, about 22 million fish fingerlings should be made available during this period.

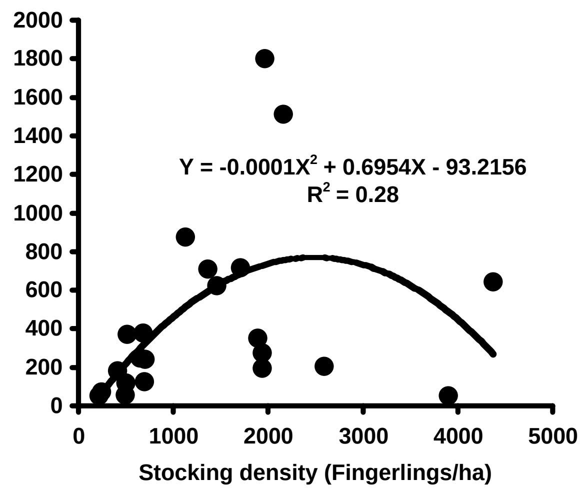


Figure 4. Relationship between fish yield and stocking density of seasonal reservoirs (estimated from data of Wijenayake *et al.,* 2005).

Discussion

Almost entire reservoir resource in Sri Lanka, with the exception of recently constructed hydroelectric reservoirs supports agricultural food production in the country. As shown in the present analysis, extent of perennial reservoirs in each district is directly related to per capita freshwater fish consumption. As such, in addition to agricultural production, major perennial reservoirs of Sri Lanka support animal protein production in the form of fish production. This demands for assuring sustainability of reservoir capture fisheries at their optimal levels. This is of particular importance because marine fish is consumption is much low in inland districts possibly due to the availability of good quality freshwater fish locally.

Also, there is a significant potential for the development of culture-based fisheries in small, village reservoirs of the country. Recent studies indicated that an average fish yield of about 450 kg ha^{-1} can be achieved during a single culture cycle within a year from the culture-based fisheries in these village reservoirs (Wijenayake *et al*., 2005). However, in order to achieve success of this strategy, a strong extension mechanism is needed to obtain active community participation for its implementation. Also, it is necessary to produce a large number of fish fingerlings (about 22 million) to make available during the correct time (Figure 5) for stocking in the small reservoirs. A strategy for developing a strong cooperation between agricultural and fisheries authorities is presented in Figure 6. Obviously, this needs to be developed through strong extension mechanism.

MONTHS																	
M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O
INDUCED BREEDING OF MAJOR CARPS		FRY REARING	FINGERLING REARING			STOCKING IN RESERVOIS			CULTURE PERIOD							HARVESTING	

Figure 5. Correct timing of culture-based fisheries in seasonal reservoirs of Sri Lanka. Rainy reason is shaded.

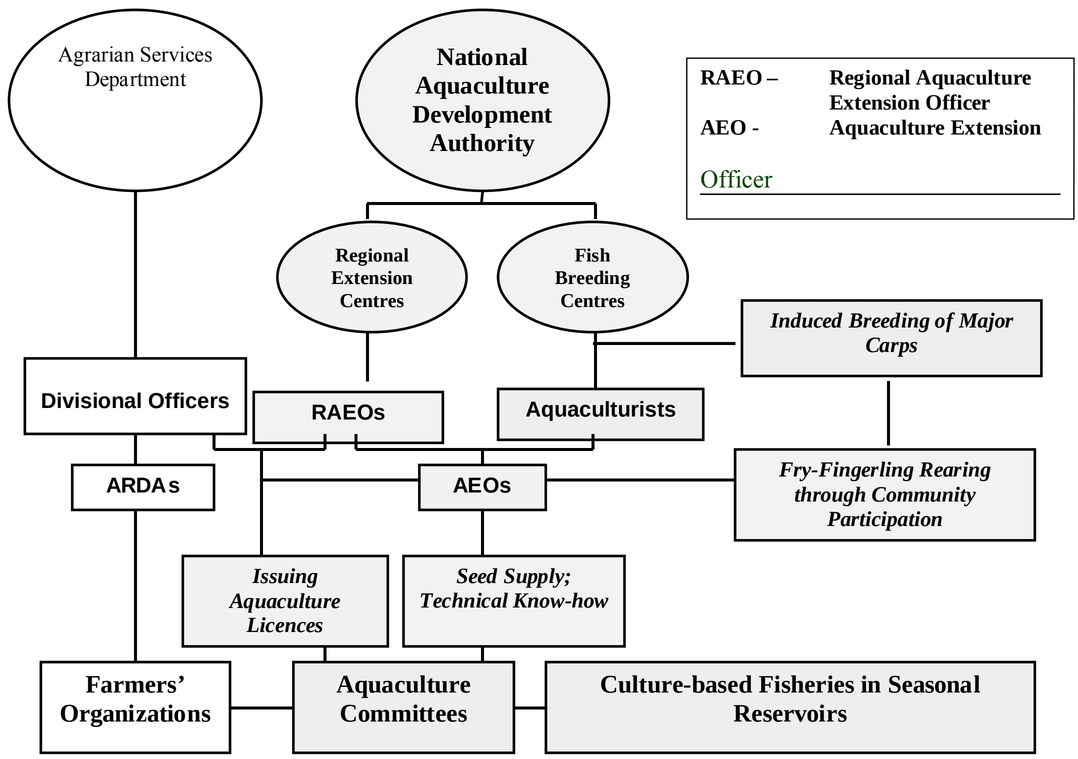


Figure 6. Proposed strategy for developing a strong cooperation between agricultural and fisheries authorities for the development of culture-based fisheries in small, village reservoirs of Sri Lanka. Shaded boxes indicate links to be developed through aquaculture extension mechanisms.

Amarasinghe and Nissanka (2005) have indicated that during different periods between early 1950s and mid 1990s, inland fisheries production in major reservoirs of Sri Lanka has shown both positive and negative trends. For example, introduction of exotic cichlid species, *O. mossambicus* into Sri Lankan freshwaters in 1952 was responsible for dramatic increase of inland fish production in the country (Fernando & Indrasena, 1969). During the early 1980s, fisheries cooperative societies (FCS) in Sri Lankan reservoirs were functioning effectively due to the reason that a boat subsidy scheme was implemented through FCS, which also facilitated implementing fisheries regulations effectively (Amarasinghe & De Silva, 1999). Conversely during the period 1990 - 1994, government funding for monitoring and stocking programmes was interrupted due to a policy decision to stop state support for inland fisheries development, which has resulted in marked decline of about 70% in annual inland fish production due to overfishing (Amarasinghe & De Silva ,1999).

This evidence suggests that under the existing system of centralized management, an effective mechanism is necessary to implement state sponsored monitoring procedures. It has been reported that in some Sri Lankan reservoirs, fishing communities were also actively involved in making decisions for management of the fisheries (Amarasinghe & De Silva, 1999). Such a co-management procedure in which the centralized administration authority and fishing community share the responsibilities of making decisions for fishery resource management (Berkes, 1994; Sen & Raajaer-Nielsen, 1996) is shown to be useful for preventing over-exploitation of fishery resources.

In contrast to small (<250 ha) reservoirs, stocking of fish fingerlings is shown to be ineffective for enhancing fish yields in large (> 750 ha) reservoirs due to low recovery rates (Amarasinghe ,1998).

The present synthesis highlights the importance of reservoir fishery in Sri Lanka as a source of relatively cheap animal protein for rural communities. Future prospects of this sector for food security should therefore be properly understood to give a high priority for inland fisheries research and development in national development plans.

Acknowledgements

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References

Amarasinghe U. S., 1998. Reservoir fisheries management in Sri Lanka: Achievements, mistakes and lessons for future. *Internat. Rev. Hydrobiol*. 83 (Special Issue), 523-530.

Amarasinghe U. S. & De Silva S. S., 1999. The Sri Lankan reservoir fishery: a case for introduction of a co-management strategy. *Fish. Manage. Ecol.* 6, 387-399.

Amarasinghe U. S. & Nissanka C., 2005. Fisheries in perennial reservoirs of Sri Lanka: Strategies for management. *In.* M. J. S. Wijeyaratne & U. S. Amarasinghe, eds. *Participatory Approaches to Reservoir Fisheries Management: Issues, Challenges and Policies*). pp. 163-177. German Technical Cooperation (GTZ) and Sri Lanka Association for Fisheries and Aquatic Resources, Colombo.

Amarasinghe U. S., De Silva, S. S. & Nissanka C., 2002. Evaluation of the robustness of predictive yield models based on catchment characteristics using GIS for reservoir fisheries in Sri Lanka. *Fish. Manage. Ecol.* 9, 293-302.

Anon., 2000. *Data Book for Village Irrigation Schemes of Sri Lanka* (23 volumes). Ministry of Agriculture and Lands, Department of Agrarian Services, Colombo.

Berkes F., 1994. Co-management: Bridging the two solitudes. *Northern Perspectives* 22: 18-20.

De Silva S. S., 1988. Reservoirs of Sri Lanka and their fisheries. *FAO Fish. Tech. Pap*. 298, 128 p.

De Silva S. S., Wijenayake W. M. H. K., Gunaratne A. B. A. K. & Amarasinghe U. S., 2004. Use of GIS tools to develop a scale for the selection of non-perennial reservoirs for culture-based fisheries activities. *In.* T. Nishida, P. J. Kailola & C. E. Hollingworth, eds. *GIS Spatial Analyses in Fishery and Aquatic Sciences* (Volume 2) pp. 559-572. Fishery and Aquatic GIS Research Group, Japan.

Fernando C. H. & Indrasena H. H. A. 1969. The freshwater fisheries of Ceylon. *Bull. Fish. Res. Stn Ceylon* 20, 101-134.

Jarchau,P., Lidzba C. & Jayasekara S., 2005. Feasibility assessment of fish farming in seasonal reservoirs: Packaging technical, social and financial aspects. In: *Participatory Approaches to Reservoir Fisheries Management: Issues, Challenges and Policies* (eds M. J. S. Wijeyaratne & U. S. Amarasinghe). pp. 193-205. German Technical Cooperation (GTZ) and Sri Lanka Association for Fisheries and Aquatic Resources, Colombo.

Mendis A. S., 1977. The role of man-made lakes in the development of fisheries in Sri Lanka. *Proc. IPFC* 17(3), 247-254.

Sen S. & Raajaer-Nielsen J., 1996. Fisheries co-management: a comparative analysis. *Marine Policy* 20, 405-418.

Siriweera W. I., 1994. *A study of the economic history of pre-modern Sri Lanka*. Vikas Publishing House Pvt Ltd., Jangpura, New Delhi.

Wijenayake W. M. H. K., Jayasinghe U. A. D., Amarasinghe U. S., Athula J. A., Pushpalatha K. B. C. & De Silva S.S., 2005. Culture-based fisheries in non-perennial reservoirs in Sri Lanka: Production and relative performance of stocked species. *Fish. Manage. Ecol.* 12, 249-258.

Department of Census and Statistics, 2002. *Headcount Index and Population below Poverty Line by DS Division–Sri Lanka*. Available from URL: http://www.ststistics.gov.lk. Accessed 27 May 2005.

Assessment of fishery resources after the impoundment of the Pasak Jolasid reservoir, Lopburi Province, Thailand

Veeravaitaya Narong[1], Suchart Ingthamjitr[2], Boonsong Srichroendham[1] and Pongpat Boonchuwong[1]

[1]Faculty of Fisheries, Kasetsart University, Bangkean, Chatujuk, Bangkok 10900 Thailand.
[2]Department of Fisheries, Kasetsart Campus, Bangkean, Chatujuk, Bangkok 10900 Thailand
[1]E-mail:ffisnrv@ku.ac.th

Abstract

The study of assessment of fishery resources in new Pasak Jolasid reservoir, Lopburi province was carried out to make available necessary information for technically sound management of the reservoir and it surrounding area. The 2-year study starting from 2003-2004 cover three main parts including ecological and socioeconomic assessment in order to enable comprehensive understanding of fisheries situation comparing before and after dam construction and recommend measures for management plan. The first part covered water quality analysis , phytoplankton, zooplankton, benthos and aquatic plants which collected field data 6 times a year. The second part concentrated on fisheries aspect including fish species, abundance, standing crop and catch per unit of fishing effort (CPUE). Collecting methodology were employed by electrofishing, series of gill nets setting in deeper areas as well as fish surveyed from fishermen and local markets. The last one studied on socio-economic of fishermen in the area. The results showed that existing water quality in the reservoir was suitable for aquatic habitat. The species and number of phytoplankton ,zooplankton, benthos and aquatic plants were increased at present comparing before dam construction. Total fish of 28 families , 97 species and 31 families , 130 species were recorded for before and after dam construction respectively. The range of standing crop were 0.59-1.69 kg/ha and 0.99-1.73 kg/ha for before and after dam construction respectively. The existing number of fishermen were 226 households.

The total catch in 1999-2000, 2000-2001, 2001-2002 and 2002-2003 were 577.593 tonnes per year 427.626 tonnes per year 612.706 tonnes per year 309.763 tonnes per year respectively. There is tendency of changing in fishery resources from riverine to standing water system.

Key words: assessment of fisheries resources, Pasak Jolasid reservoir

Introduction

Pasak River with total length of 513 kilometers flows in north-south direction. General elongate shape of the river causes rapid change in water level which usually causes flood and drought in every year. In accordingly with the king's perception, the Royal Irrigation Department conducted a study on the impact of Pasak reservoir project. The study indicated that Nong Bua sub-district in Pattananikhom district and Kamphran sub-district in Wangmuang district is the suitable place for dam construction. The construction started on 2nd December 1994 and completed on 30th September 1999. The king kindly named the reservoir as Pasak Jolasid since then.

The dam has general features of 4,380 meters in total length, maximum water level +43 meter above mean sea level (mmsl) but normal storage is at +42 mmsl with the volume of 960 million cubic meters cover area of 148.75 km^2 (Royal Irrigation Department, 1999)

The dam caused substantial change of hydrology of the river from *lentic* to *lotic* condition which impacts on aquatic ecology, aquatic fauna and people living around. Many authors and institutes, exemplify Chookajorn *et al.* (2001), Tangtrongpairoj and Sinchaipanich (2001)studied these impacts. The dam was constructed with main purpose for flood protection and irrigation but similar to many other dams in that it always included fisheries, tourism and some other activities in its multifunctional utilization.

This study has general objective to assess the impact of dam on fisheries aspect and generate measurement to sustain fish production and well being of local people. The study has following immediate objectives; to study the difference of ecology by focusing on water qualities, plankton, benthos and aquatic flora before and after impoundment ;to study the difference of fish and fisheries before and after impoundment; to study the socio-economic of local people after the impoundment and ;to propose mitigation plan, implementation plan, utilization plan and the proper fisheries management measures in the future.

Materials and methods

Study areas

Sampling stations were in and around the reservoir depending on each study aspect. Study on ecology and fisheries were generally carried out in 3 major areas in the reservoir; the inlet or upper part, reservoir or middle part and the outlet or lower part. Fish samples were collected from fish landing sites around the reservoir. Socio-economic information was collected from people in Pattananikhom, Tahluang and Chaibadan districts in Lopburi province and in Wangmuang district in Saraburi province (Figure 1).

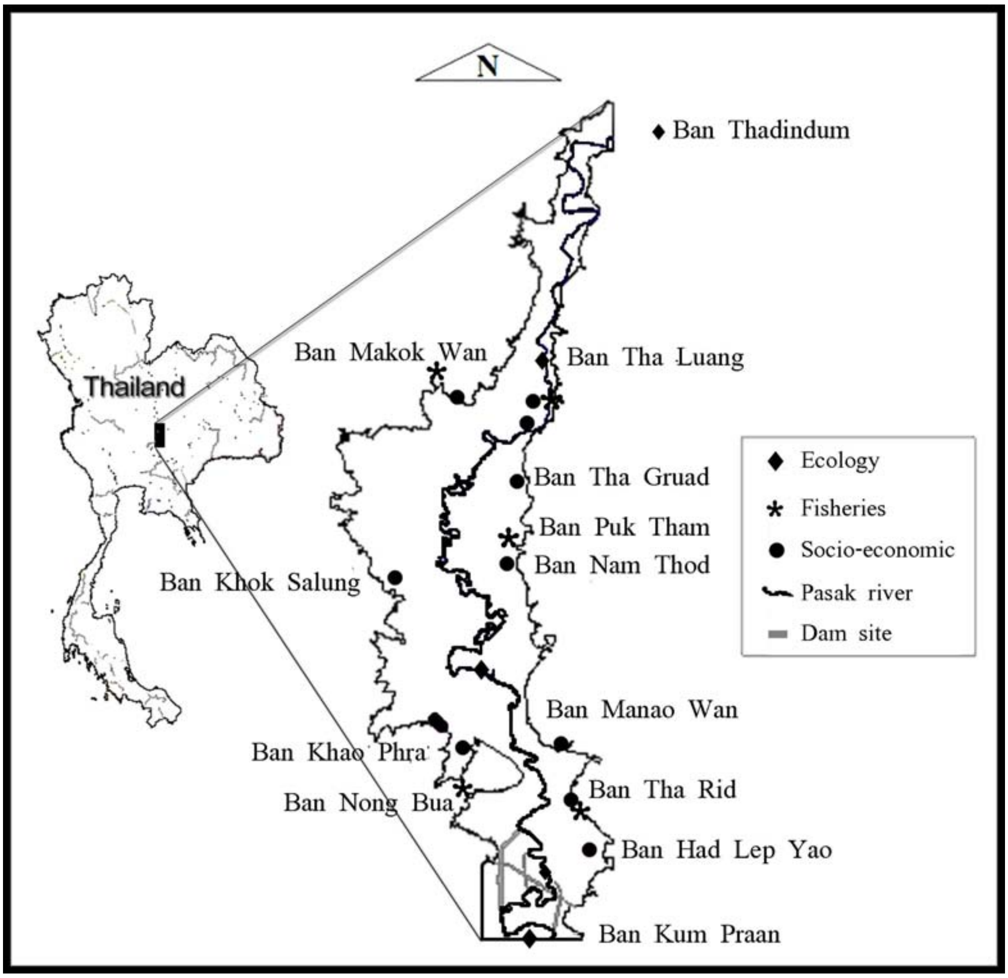


Figure 1. Map of Sampling area and stations.

Parameters, sampling technique and analysis

Aquatic ecology

Water quality

The monitoring parameters included temperature, transparency, pH, dissolved oxygen conductivity, alkalinity, hardness, ammonia, nitrate, total nitrogen, orthophosphate, total phosphorus and chlorophyll a. Used water sampling and analytical technique according to standard method (APHA, 1985)

Plankton

Use plankton net with mesh size 20 micrometer and 65 micrometer for phytoplankton and zooplankton collection, respective. Plankton was classified by using hand out of Wongrat (1995).

Benthos

Collect benthos at river base by grab and then primarily screen out the sample by sieve with mesh size 250 micrometer. Identify and classify benthos by using hand out of Bradt (1974) and Edmondson (1963).

Aquatic flora

Collect aquatic flora by surveyed, photographed from the marginal, littoral and pelagic area of the reservoir. Identified aquatic flora by using hand out of Department of fisheries (1995) and Sripen (1987)

Fisheries

Fish community and fish catch

Collect fish sample by purse seine, electro-fishing gear, beach seine and net. Identified and classified fish samples according to Rainboth (1996) Taki (1974) and Smith (1945). The primarily analyzed data will further be used to analyze for the biodiversity of fish, structure and ecological index of fish community.

The potential yield

Compile total length of 12 selected high economic fish species by sampling fishes from fishermen every month for 12 months and use the collected data to analyze for fish catch and fishing effort as well as fish production.

Socio-economic

Use Snow-ball sampling technique simultaneously with open and close questionnaires to collect data for socio-economic analysis.

Study period

The project scheduled for 18 months with the major period of 12 months for field data collection, started from 1 April 2003 to 30 September 2004 Each study aspect has different number of field data survey. Ecological study collected data every 2 month while fisheries study particularly fish community and fish catch had 3 field exercises and the potential of economic fish production study had collected fish sample every month for one year. Socio-economic conducted 4 field surveys.

Results and discussion

Aquatic ecology

Dam construction results a large reservoir which plays important role for fish production. The north-south direction elongate reservoir located at the utmost south of Pasak river basin, therefore, most of surface water occurring within the basin must flow through the reservoir and pass through the dam before confluence with Chaopraya River. Annual water exchange varies between 1,738.41-4,756.48 million cubic meters. It can be presumed that water in the reservoir is completely changed every year since the annual exchange is much higher than the capacity. Water level is normally maintained at minimum during rainy season with the aim to provide sufficient space for the coming inflow.

Some important physico-chemistry of water in the reservoir are; water temperature 29.1±2.1 degree Celsius, dissolved oxygen 6.7±1.7 mg/l hardness 112±22 mg/l, alkalinity 121±12 mg/l, pH 8.2±0.3, turbidity 88±117 FTU, ammonia-nitrogen 0.088±0.043 mg/l, total nitrogen 0.487±0.238 mg/l, nitrate nitrogen 0.060±0.024 mg/l, total phosphorus 0.096±0.086 mg/l, orthophosphate 0.019±0.009 mg/l and chlorophyll-a 25.8±26.8 μg/l.

Water quality after impoundment especially temperature and turbidity has a markedly smaller amplitude within a year comparing to that of before. Turbidity from Manao Wan to dam site is considerably stable across the year while the inlet station which covers the area from Thadindum bridge to Tah Luang district has remarkable seasonal variation of turbidity. Decreasing in turbidity in the reservoir markedly indicated the occurrence of sedimentation. It was found from the study that most of sediment settled down between Thadindum Bridge and Manao Wan village.

Sediment in the reservoir is nutrients enriched which plays crucial role on aquatic ecosystem. More than 60% of Pasak watershed is agricultural land, therefore a lot of nutrients were flushed into the reservoir annually either in solution or suspended solids form.

Under nutrients enriched condition coupled with high hardness and alkalinity and the shallowness of the reservoir caused Pasak reservoir the high potential eutrophication. It was found from the study that the reservoir can be classified as Eutrophic to Hypertrophic according to OECD (1982). The reservoir has annual average chlorophyll-a 38.4 μg/l, 74 μg/l total phosphorus and 54 cm of transparency. It should be noted that phytoplankton bloom in the reservoir may related to water level since the level during the study period was maintained extremely low.

Over-saturation of dissolved oxygen measured in January 2004 reaching to 9.6-13.1 mg/l was the signal of highly abundance of phytoplankton. Diversity of phytoplankton increased to 59 species with the dominant species of *Oscillatoria,* It should be noted that *Microcystis, Cylindrospermopsis* and *Anabaena* are toxic algae. Zooplankton also increased to 35 species with the dominant species of *B. facaltus* and *Brachionus angularis*. Benthos increased to 23 species with the dominant species of *Melanoides* sp. and *Scaphula* sp. There are 38 species of aquatic flora. The dominant species are Narrow leaved cattail *(Typha angustifolia) and Giant mimosa (Mimosa pigra)*.

Gastropod was the highest abundant among those benthos found. *Pila* sp. and *Melanoides* sp. are predominant gastropod while *Scabies* sp. is dominant bivalve. Exotic species like apple snail was found extensively by evidence of its adult shell and pink color eggs. This should be closely monitored since it may impact on population of local gastropods in long term basis.

Fisheries

Fish diversity

Only 97 fish species in 28 families were found in Pasak River (before impoundment) while recent study found increased to 130 species in 31 families. There are 80 identical species found either before or after impoundment, 24 species found only before impoundment and 52 species found only after impoundment. Total record of fish species found during 1981-2004 is 154 species in 35 families.

Cyprinidae is the most numerous family, comprises up to 40 species followed by Bagridae and Siluridae. The number of 45 species accounted for 45.45% of the 97 species found was found in every sampling station.

Fish abundance

The recent study found fish abundance in the reservoir tended to decrease from the previous studies in 2000 and 2002. The standing crop was 3.78 kg/ha, 2.05 kg/ha and 1.05 kg/ha in, 2000, 2002 and 2004 respectively. When comparing standing crop before and after dam construction it found that the range of standing crop were 0.59-1.69 kg/ha and 0.99-1.73 kg/ha for respectively.

Fish structure

The number of 13 predominant species accounted for 80% of the total production (by weight) could be categorized into Cyprinidae 69.33%, carnivorous 6.06%, catfish 4.86% and others 19.74%. F:C ratio of 1.8:1 explained the considerable large of carnivorous fish group. Fish diversity index is also high up to 2.718 ± 0.165.

Fisherman census

The number of fisher households living around the reservoir surveyed in 2000-2002 was 493 while the current study found only about 226 households. The more populated was on the eastern part particularly at Manao Wan is the most populated area.

Fishing gear

There were 14 types of fishing gears commonly used in the reservoir. The widely used gears are gillnet, hook long line, shrimp trap and cast net. The fishing gears developed by exotic fishermen are large drop net, small shrimp trap, larger shrimp trap, shrimp hook long line and beach seine.

Fish catch

The total catch in 1999-2000, 2000-2001, 2001-2002 and 2002-2003 were 577.593 tonnes per year 427.626 tonnes per year 612.706 tonnes per year 309.763 tonnes per year respectively. In 2003-2004 , fish catch varies between 337.8-445.6 tonnes/year or average 0.71-1.05 kg/ha/year or 4.21-61.51 kg/fisherman/day (average 18.37 kg/fisherman/day). Therefore, the reservoir could be classified as medium-high fish yield. There is tendency of changing in fishery resources from riverine to standing water system. Income generated from fish catch accounted to 12-20 million baht/year.

Economic fish

Out of 40 cached species, 12 species are regarded as high economic value species. These included Silver barb *(Barbodes gonionotus),* Smith's barb (*Puntioplites protozysron),* Nile tilapia *(Oreochromis niloticus),* Jillian's mud carp *(Henicorhynchus siamensis),* Red-tail tinfoil barb*(Barbodes altus),*Beardless barb *(Cyclocheilichthys armatus),* Black shark Minnow*(Morulius chrysophygadian),* Barb*(Labiobarbus siamensis),* Yellow catfish*(Hemibagrus nemurus)* ,Butter catfish *(Ompok bimaculatus),* Grey featherback *(Notopterus notopterus*), Smokey glass catfish *(Kryptopterus crytopterus).* Only three species are being under-exploited or exploited at the same level as its potential yield. These included Black shark minnow (*Morulius chrysophykadian)*, Whisker sheatfish (*Micronema bleekeri)* and Yellow catfish (*Hemibagrus nemurus)*. The other 9 species are being over-exploited.

Socio-economic

Structure

Local people operate mainly on agriculture both before and at the beginning of the reservoir impoundment. At that time, fisheries played an insignificant role since small fish catch was used only for household consumption. However, after impoundment for a couple of years and the skillful northeastern migratory fishermen became catch a lot of fish, many fisheries related activities operated by local people living around the reservoir became emerging. Exemplification of these activities is fish marketing, fish processing and many other related businesses. Further more the number of local people living around the reservoir switching to operate fishing was becoming increase.

Fish marketing

Fishermen sale their fish to fish traders at landing sites located around the reservoir. Because the reservoir is close to important fish markets in Bangkok, Angthong and Nakhon Pathom province, therefore fish from the reservoir can fetch considerably good price from these markets.

Over 40 harvested species could be categorized into 3 groups; low value small fish, high value larger fish (flesh fish) and expensive live fish e.g. Giant freshwater prawn *(Macrobrachium rocenbergii)* and Sand goby *(Oxyeleotris marmorata).* Fishermen market their fish through its marketing channel and the consumer may utilize fish as fresh, transformed products or live by the proportion of 63.3%, 34.4% and 2.0%, respectively.

Attitude of fisherman

Perception on the measurements for fisheries management of local and migratory fishermen is not much difference. The only difference is concerning with the use of engine boat for fishing. While the local fishermen support the measure of prohibiting use of engine boat for fishing but the exotic fishermen oppose the measure due to the reason of safety particularly during monsoon season. All fishermen have the same perception on the decreasing trend of fish production. In addition, majority of fisherman want the reservoir to be used for multi-purposes including fisheries, water utilization, tourisms and flood protection.

Conclusions

1. Water quality in Pasak Jolasid reservoir after impoundment is generally suitable for fish habitat. But eutrophication during dry season must be closely monitored.
2. The benefit derived after impoundment of Pasak Jolasid reservoir in term of fish production has been satisfying the local fishermen. However, continuous declining trend of fish yield was found.

Recommendations

1. Establish a regular ecological monitoring scheme. Variations of aquatic ecology in the reservoir are greatly affected by water level. Long term data collection on both ecological aspects and water level will suggest a clear relationship between water level and aquatic ecosystem. Hence, optimal water level could be regulated to maintain good ecological health which favors fish production.

2. Fish stock enhancement:

- Announce more fish conservation areas by the cooperation of local people. The announcement should focus on brood fish refuge areas during dry season as well as spawning and nursing areas for fish larvae during rainy season.
- Strictly control fishing method and fishing gears particularly during the early spawning season when brood fish migrate from refuge areas to spawning areas.
- Stocking indigenous fish that feed low to the low level of food chain as well as giant freshwater prawn. The latter already proved high potential yield.

Maintain optimal water level to provide suitable area for brood fish to refuge during dry season and facilitate brood fish to migrate to spawning ground during spawning season.

4. Maintain fish diversity by discourage use of the over-exploited species and encourage use of those under-exploitation for example Black shark minnow (*Molurius chrysophykadion*) and Thai river sprat (*Clupeichthys aesarnensis*).
5. Encourage pond aquaculture instead of cage culture because water variation in the reservoir both in term of quantity and quality may cause trouble to cage culture.
6. Increase value of fish by organizing fish transforming technique training programme to local people.
7. Government should have proper regulations comply for migratory fishermen during their stay and operate fishing.

Acknowledgements

This research was supported by Office of The Royal Development Projects Board (RDPB) and The Thailand Research Fund (TRF). The authors gratefully acknowledged Faculty of Fisheries and the Kasetsart University Research and Development Institute (KURDI) from Kasetsart University for provided funding to participate in the 11th World Lake Conference.

References

APHA (American Public Health Association). 1985. *Standard methods for the examination of water and wastewater.* Washington, D.C.

Brandt, Rolf A.M. 1974. The Non Marine Aquatic Mollusca of Thailand. *Archiv for Mollus-ke.nkunde no. 105*, Frankfurl am Main. 423 p.

Chookajorn, T., Ingkasuwan, O., Pachunchai, P., Somchan, W. and Promkroun, N. 2001. Ecology and Fish Population in the Pasak Jolasid Reservoir. *Technical Paper No. 24/2001.* Fisheries Environmental Division. Department of Fisheries. (*in Thai*)

Department of Fisheries, 1995. *Aquatic Plants in Thailand.* Inland Fisheries Division. Department of Fisheries. (*in Thai*)

Duangsawasdi, M., 1987. Water Quality Criteria for the Protection of Freshwater Aquatic Organism. *Technical Paper No.75/1987.* National Inland Fisheries Institute. Department of Fisheries. (in Thai)

Edmondson, W.T. ed. 1963. *Fresh water Biology*. John Willey & Sons Inc., New York. 1,248 p.

OECD (Organization for Economic Cooperation and Development). 1982. *Eutrophication of Waters. Monitoring, Assessment and Control.* Final Report. OECD Cooperative Programme on Monitoring of Inland Waters (Eutrophication Control). Environment Directorate, OECD. Paris.

Rainboth, W.J., 1996. *Fishes of the Cambodian Mekong.* FAO, Rome.

Royal Irrigation Department, 1999. *The Pasak Jolasid Dam, Lop Buri and Sara Buri Provinces.* Contribution on the opening ceremony of the Pasak Jolasid Dam by His Majesty the King Bhumipol. Bangkok. (*in Thai*)

Sripen, S., 1987. *Aquatic Plants.* Kasetsart University, Bangkok. (*in Thai*)

Smith,M. H., 1945*. The Fresh-water Fishes of Siam, or Thailand.* Smithsonian Institute. United State National Museum.

Taki, Yasuhiko, 1974. *Fishes of the Mekong Basin.* United State consultants, Inc. contract no.AID 439-699.

Tangtrongpairoj, M. and Sinchaipanich, C., 2001. Caysal Factors Affecting Attitude Towards Occupational Change of People Living in The Area of Pasak Watershed Royal Development Project. *Technical Paper No. 3/2001.* Fisheries Extension Division. Department of Fisheries. (*in Thai*)

Wongrat, L., 1995. *Phytoplankton.* Kasetsart University, Bangkok. (*in Thai*)

Wongrat, L., 1995. *Zooplankton.* Kasetsart University, Bangkok. (*in Thai*)

Relationship between productivity and transparency fall discovered in water quality monitoring and plankton analysis in Lake Kasumigaura ecosystem

Atsushi Numazawa

Kasumigaura Citizens' Association 2-13-6 Kawaguchi, Tsuchiura, Ibaraki, Japan 300-0033

E-mail: kca@cg.mbn.or.jp

Abstract

Lake Kasumigaura, the second largest (220 km^2) lake in Japan is very shallow (mean depth 4m). It is located on the edge of the Tokyo metropolitan area. It was an inlet sea lagoon about 400 years ago, but is now surrounded by a breakwater bank which is effectively managed. The water is desalinated by the barrier construction for use in agricultural irrigation, industrial water and as a city water supply. The water quality has gradually deteriorated, below the domestic environmental standard, and fisheries have declined, despite various countermeasures enacted by national and local government and the efforts of the residents in the catchment area.

The Kasumigaura Citizens' Association has investigated the water quality and plankton in the last four years. In this period, we noted situations such as a remarkable reduced transparency, cool summer, low water temperature, intense heat, mass death of cultured carp, outbreak of a kind of Diatoms, Skeletonema potamos and flooding by a typhoon.

White (milky) water of unidentified cause occurred from 2002 to 2004. During this time, mass death of cultured carp infected by koi herpes virus occurred and Skeletonema potamos simultaneously piled up, but the cause for these incidents remains unknown. The same period experienced a cool summer, when water temperature and transparency reduced. A small outbreak of phytoplankton was also noted. The levels of nitrate nitrogen and phosphate phosphorous were also seen to be high.

Removing the defunct materials from the local fisheries and waterweed proved necessary in improving water quality, creating material circulation through a food chain of phytoplankton, zooplankton and fish in the ecosystem. This was verified from water quality monitoring and plankton analysis. Useful data was obtained on the upkeep of the lake.

Key words: ecosystem, eutrophication, monitoring

Introduction

Lake water monitoring by the Kasumigaura Citizens' Association performed since 2001, has provided the opportunity for the citizens to participate in the scientific observation of lake Kasumigaura water quality and productivity. The valuable data obtained over the last four years, when we experienced white water (milky water or cloudiness), mass death of cultured carp by an infection of Koi (carp) herpes virus(KHV), an outbreak of Diatoms, *Skeletonema potamos*, low water temperature in a cool summer, an outbreak of opossum shrimp, *Neomysis* and an intensely hot summer reflects the associations' efforts.

Materials and methods

Monitoring was carried out monthly on an investigation ship scheduled from 9:00 a.m. to 3:00 p.m. at six monitoring points positioned by Global Positioning System as follows:Okijyuku-offshore (36° 03'00" N,140° 15'23"E), Ushiwata-offshore (36° 02'34"N 140° 19'30"E), Edosaki inlet (35° 59'13"N, 140° 22'07"E), Tennouzaki-offshore (35° 58'37"N 140° 27'57"E), Mitsumata-Lake Center (36° 00'48"N 140° 25'12"E), Takahama Inlet(36° 07'22"N 140° 22'28"E).

At each monitoring point, we measured the water temperature, dissolved oxygen, pH, transparency and electric conductivity of the surface water (30cm depth). At the same time we recorded the water color. A 3m high plankton net (NXX13, made by Rigosha Co' Ltd. Cat. No. 5513) was used to collect zooplankton at each pond. For the phytoplankton count, the surface water was sampled and examined under the microscope. The water samples were placed in a cooling box and transported to the laboratory where the suspended solid (SS), chloride ions and chemical oxygen demand (COD) were measured along with the inorganic nitrogen (ammonium nitrogen, nitrite nitrogen, nitrate nitrogen) and phosphate phosphorous, on a HACK colorimeter DR/890.

Results and discussion

2001

In a mesotrophic lake located in a temperate zone, Diatom blooms take place in spring and autumn. It is believed that water temperature in spring and autumn is optimal for the proliferation of phytoplankton. In summer, zooplankton become active as a predator. In Lake Kasumigaura, a typical eutrophic lake, an algal bloom (Microcystis) occurred every summer from 1970 to 1980, but fibrous blue-green algae became dominant in the 1990's. In the summer of 2001, precipitation was very low and the water level dropped about 40cm below normal. Consequently, SS was high due to bottom sludge raised by waves, This resulted in reduced transparency and a little blue-green algae. Diatoms bloom appeared in spring and autumn. In this summer, the level of phosphate phosphorous was high. White water was not observed during the year.

2002

Phytoplankton bloom was observed only in the spring. Muddy white water appeared in the summer.

The transparency changed greatly, showing an average of 80-90cm, while the nitrate nitrogen value also changed after the occurrence of white (milky) water. *Neomesis* was observed from April to June.

2003

White water was observed in most of the water body throughout the year. The transparency dropped to 60cm while COD value was almost 6mg/l. The maximum phosphate phosphorous level rose to 0.2 mg/l during this year. The maximum nitrate nitrogen level was 1.5 mg/l.

A slight spring bloom of phytoplankton was observed. Moreover, the low water temperature and weak sunlight due to the cool summer, resulted in a greatly reduced amount of phytoplankton during summer and autumn. Diatoms, *Skeletonema potamos* was dominant from October to December. Coincidentally, the mass death of cultured carp (about 2000 ton) occurred that October with KHV detected in the fish bodies by the national fishery research institute. The causal relationship between *Skeletonema* and Carp death has not yet been clarified. In the summer, *Neomysis* broke out. The winter catch of Japanese ice fish, *Salangichthys microdon,* was good.

2004

White water phenomenon persisted that year. Transparency was 50-60cm and COD value was 6-7mg/l. In summer, a record heat continued. Weak waves did not raise bottom sludge. In some inlets, algal blooms (Dominant species: *Microcystis* and *Anabaena*) were observed.

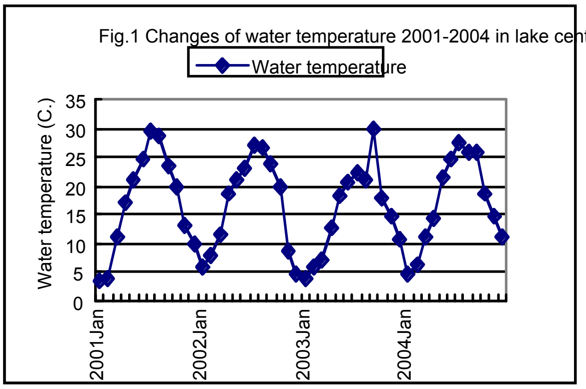
Two typhoons hit the area in autumn and a total precipitation of 400mm, while 1.25 m raise in water level was recorded. The dyke minimised water damage. Most of the white water consequently drained via an outflowing river into the sea. Due to the dilution from these floods, transparency was improved to 80-100 cm.

During the year, *Neomysis* proliferated while water fleas did not This showed high predating pressure on the water flea community. After June, *Neomysis* disappeared and the water fleas increased drastically. Phytoplankton was scarce throughout the year. Transparency improved in the latter half of the year. This was a favorable factor that lead to the estoration of Diatom blooms in the spring of 2005.

Parameters monitored were:

Water temperature

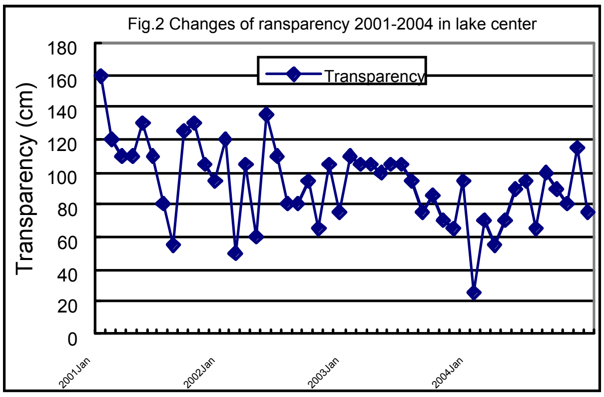
As a typical lake located in a temperate zone, the water temperatures of Lake Kasumigaura fluctuate annually, reflecting seasonal changes (Figure 1). However, in the summer 2003, it was very cold except for few days in September. The low water temperature continued during the summer of 2003 and the accumulated solar energy reaching the water body was significantly low. Therefore, it is speculated that productivity was low during this cool summer.



Transparency

About 20 years ago, the transparency of Lake Kasumigaura was high in winter, and low in summer, due to the outbreak of algal bloom . From about 10 years ago, transparency reduced in winter, because of the fibrous blue-green algae like *Oscillatoria* and *Phormidium* that are dominant throughout the year.

Nevertheless in 2001, above one meter of transparency was frequently recorded at each monitoring station. White milky water occurred in 2002 accompanied by bottom sludge raised by strong winds and waves. Transparency eventually dropped to 50-60cm (Figure 2). The period of low transparency continued until the summer in 2004. This abnormal transparency fall for 2 years created serious problems in water quality, thus resulting in reduced productivity.



Suspended Solids (SS)

The values of SS fluctuation reflect the density of phytoplankton, floating micro particles emanating from disturbed muddy inflowing water from rivers. At two monitoring stations, Okijyuku-offshore and Edosaki inlet, it was observed that SS values frequently increased after heavy precipitation. This was attributed to their proximity to river mouths. Even at the lake center monitoring station, when strong winds blew, the SS values increased rapidly, because sludge rose due to the shallowness of the water. However, during the occurrence of white water, concentrations of SS was frequently low, suggesting that SS was not the cause of the white water.

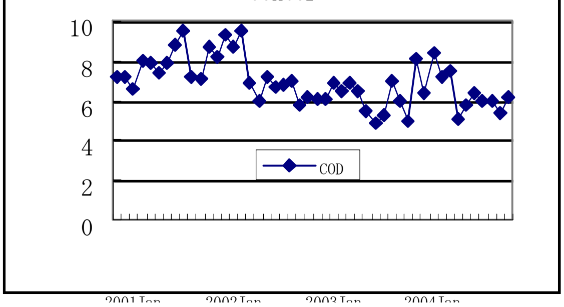
Correlation between transparency and SS

Generally, there is a negative correlation between transparency and SS concentration. However, there was a spot where transparency was significantly reduced as a result of cloudiness while SS was exceptionaly low. Readings for cloudiness due to white water and SS did not always tally. This is because SS are mainly made up of particulate substances that settle eventually in tranquil water. The substances that cause the white water, do not sink immediately in calm conditions but remain suspended for a long period, like a colloidal solution.

Chemical oxygen demand (COD) values

In Lake Kasumigaura, the COD values used to increase abruptly in a short time and this phenomenon was thought to occur as a result of diffusion of bottom sludge during strong winds and waves. It was observed that during the period of white water (from spring 2002 to summer 2004), the COD showed comparatively low values suggesting that sludge raising was not the direct cause of the white water (Figure 3). The possibility that colloidal components of stired bottom sludge were indirectly the cause of white water could not be excluded. The exact cause remains elusive, despite efforts made by researchers.

Fig.3 Changes of COD 2001-2004 in center



Phosphate phosphorous (PP)

The concentration of PP fluctuated intensely over the four years, but showed a tendency to increase during the white water or low COD values. This period correlated well with the time of few phytoplanktons as later mentioned.

Inorganic nitrogen

The concentration of nitrate nitrogen (NN) also increased when white water was at its worst. This tendency was observed at all the monitoring points.

Phytoplankton

About two decades ago, Lake Kasumigaura was notorious for a heavy bloom of blue green algae every summer, in particular *Microcystis*. Fibrous blue-green algae then became dominant for several years. But from 2000, Diatoms, like *Synedra* and *Melosira* were observed to increase.

Until spring 2002, Diatom blooms were observed periodically in spring and autumn of each ordinary year. However, since the white water emerged, no blooms were observed. In this period, low COD, high PP and NN concentration were characteristic, as already mentioned.

Occurrence of phytoplankton depends on the transparency among other factors. Solar energy generally reaches about two times the Secchi depth that is compensation depth. During the white water period, the worst Secchi depth was only about 50 cm. This meant that most of the lake water below one meter was a dark environment and primary production was extremely inhibited.

Zooplankton

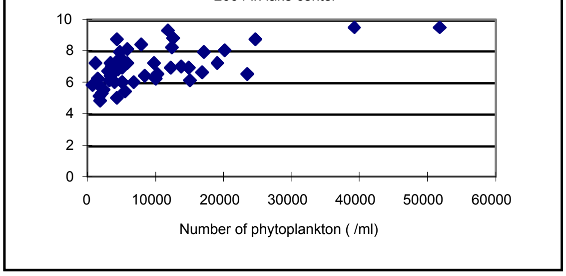
The number of Zooplankton in Lake Kasumigaura reaches a maximum every summer. *Diaphanosoma* and *Bosmina* increase mainly in the summer while *Copepodas* are common throughout the year. These water fleas are preyed on by *Neomysis.* Once *Neomysis* breaks out, zooplankton almost disappear in Lake Kasumigaura.

Despite *Neomysis* being few, the number of Zooplankton became less in the cool summer of 2003. White water was prevalent from summer to autumn tduring the year. A low water temperature and short daylight hours characterized this cool summer. Common phytoplankton did not proliferate, but rare Diatom, *Skeletonema potamos* dominated for a while only in autumn. Mass death of cultured carp (about 2500 tons, one-third of the total) coincided with this period. Ecologically this period was the worst ever in the Lake Kasumigaura ecosystem.

Correlation between COD and phytoplankton

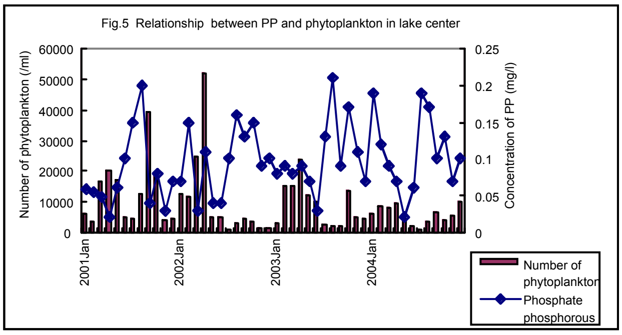
COD values become high when there is a large number of phyto-plankton. Interestingly, COD values did not fall under about 5 mg/l, even though phytoplankton was almost zero, suggesting that there were dissolved low molecular organic substances such as organic acids in the lake water (Figure 4).

Fig.4 Correlation between COD and phytoplankton 2001-2004 in lake center



Correlation between phosphate phosphorus (PP) and phytoplankton

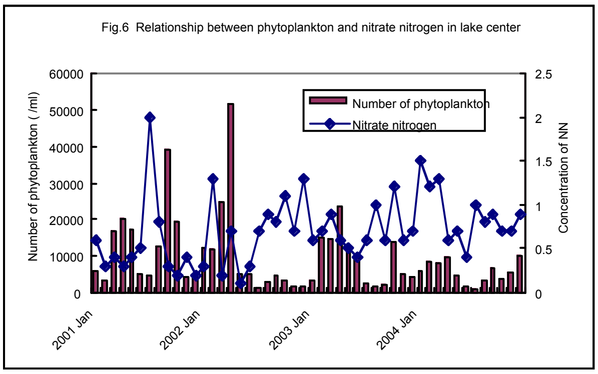
A negative correlation was observed between the PP band the number of the phytoplankton (Figure 5). When there was explosion of phytoplankton population, they absorbed the PP in the lake water considerably thus reducing its concentration.



The density of phytoplankton seems to control PP concentration in Lake Kasumigaura. If phytoplankton die suddenly, they decompose and PP concentration rises as expected. On the other hand, if phytoplankton is predated appropriately by zooplankton, energy flows via the food chain, nitrogen and phosphorous are finally removed through fishing, while water quality improvement is realized.

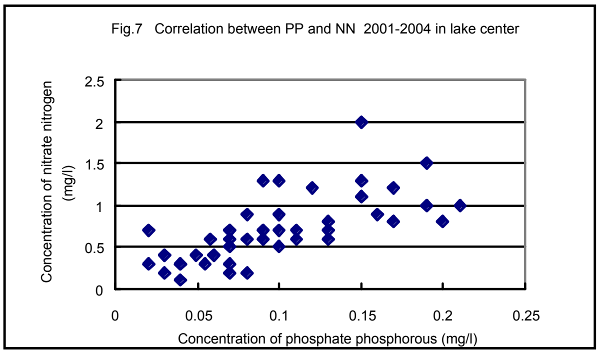
Correlation between nitrate nitrogen (NN) and phytoplankton

There was negative correlation between nitrate nitrogen (NN) and phytoplankton.. When the NN was low there was a great number of phytoplankton (Figure 6). This was similar to the case for PP



Correlation between PP and NN

As indicated in previous discussion, there was a positive correlation between PP and NN (Figure 7). However it is difficult to predict the changes of both components by the behavior of plankton only. T



Conclusions

In Lake Kasumigaura, white milky water and poor transparency were observed from 2002 to 2004. During this period, primary productivity was considerably inhibited, while levels of PP and NN increased. This suggested that the energy flow was inhibited. Negative correlation between the density of phytoplankton and concentration of trophic salts such as PP or NN indicated an effective way for the improvement of water quality. Improvement in transparency is very difficult in a eutrophicated wide, shallow lake like Kasumigaura, located in a plain and polluted by various loads. The white milky water was temporarily replaced by water flowing in from inflowing rivers after the heavy precipitation of typhoons in autumn 2004. However, weak white water has been observed again in the summer of 2005.

Towards the management of pollution loads into lake Victoria: Treatment of industrial effluent discharged into river Nzoia, Western Kenya

Orori, B. O., L. Etiégni, and K. Senelwa

Department of Forestry and Wood Science, Moi University, Box 1125, Eldoret - 30100, Kenya.
Email:benorori@yahoo.com

Abstract

River Nzoia, which drains into Lake Victoria, is one of the most abused rivers in western Kenya. Major industries including pulp and paper, sugar and chemical industries discharge their effluent into this river. The effluent discharged is highly variable in nature with BOD ranging from 125.3-520.8 mg/l; COD, 410-800 mg/l; colour, 1628-3200^{o}H; turbidity, 331.33-752.5 NTU; and total solids, 457.5-1145.8 mg/l. The water drains into the lake with disastrous effects on the lake ecosystems and the associated uses.

The ELCAS method, a low cost new technique developed at Moi University for the treatment of a pulp and paper mill effluent reduces the effluent BOD, COD, total suspended solids colour and turbidity by over 80%, making it possible to recycle the treated wastewater. The treatment and the potential to recycle the water reduces the pollution loads discharged into the river eventually safeguarding the ecosystem integrity of Lake Victoria and its environs. Continuing research is applying the technique to other industrial effluents in the tea and coffee industries around the country.

Key words: pollution loads; ELCAS and lake Victoria.

Introduction

Industrial establishment in western Kenya is on the rise due to favorable climatic conditions for raw material production. Pulp and paper production, sugar production, chemical processing and other agricultural related industries are among the many industries in the region. These industries have been known to discharge their effluent with little or without treatment at all into the few streams and rivers in the area. These streams and rivers empty their waters into Lake Victoria, the second largest fresh-water Lake in the world. The sugar, an integrated pulp and paper, chemical processing and other agro-based industries are situated along the banks of River Nzoia that drains its water into the lake. These industries are a significant point source of highly polluted discharges into the few water resources in the western part of Kenya.

River Nzoia the largest and longest (334 km) river in western Kenya receives its waters from the cherengani hills and Mount Elgon and is one of the rivers highly abused by these industries in the region. Figure 1 shows the main effluent characteristics of the three main industries that discharge their effluent into River Nzoia. The three main industries include the Pulp and paper (PPM), Mumias sugar (MMS) and Nzoia sugar (NZA) industry.

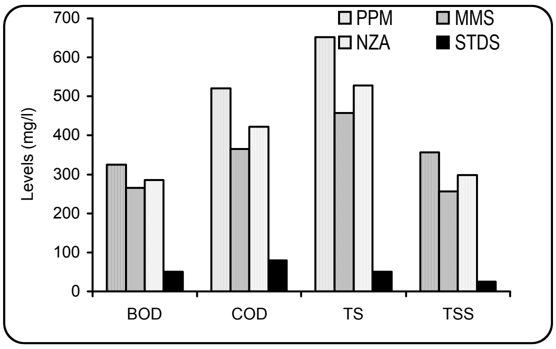


Figure 1: Effluent characteristics of major industries discharging their effluent into river Nzoia.

These pollution loads have a wide upward variation depending on production schedules, resulting into drastic effects on water resource and ecosystem in general. High pollution loads do not only affect the Lake Victoria and its environs, but all countries utilizing water from the Nile River; the longest river in the world. Even though the pulp and paper (8.01%) and the sugar industries contribute a greater share of the GDP to the economy of the country their effluent discharges are far below the international standards (Situma, 2003; Orori, 2003; Achoka, 1998; GOK, 2000).

Unlike the sugar industries, pulp and paper production effluent is known by its high pollution strength into streams (UNEP, 1981). The industry has been ranked the twentieth highest environment polluting industry in the world (Mall *et al.,* 1989). The presence of a pulp and paper if not well managed can be a point environmental degradation (UNEP, 1981).

High pollution loads discharged into water bodies, results into contamination and toxicity; excessive BOD and siltation; loss of aquatic resources and biodiversity. Furthermore, it results in increased water treatment cost for industrial and domestic use and reduced aesthetic value of the water bodies as well as reduces light penetration of the receiving waters and potentially affects benthic plant growth and habitat. (Springer *et al.,* 1995; Fifiled and Kealay, 1990; Mahida, 1991)

There has been extensive investigation on appropriate methods for industrial effluent treatment method. Currently, industries use physical, chemical or biological treatment methods; depending on the quantity of wastewater they generated and intended final use of the effluent. Biological treatment method is widely used because of its low operational cost. However, it is unable to achieve the required international effluent discharge standards for receiving water bodies in a simple, efficient and sustainable manner (Archibald and Roy-Arcand, 1995; Hilleke, 1991).

Effluent treatment remains to be a major drawback to industrial establishment despite the fact that most of the industries have to pass through an environmental impact assessment test before they are allowed to produce. Electrochemical techniques can achieve reduction of pollution loads into water bodies, however, electrical potential within electrodes normally causes extra voltage, which wastes energy (Springer *et al.*, 1995; Ahonen, 2001). Kenya has one of the highest electrical energy costs in the World. This method can only be applied in Kenya, if there is a way of reducing power consumption. This will make it simple, efficient and economical and help meet the new set standards for effluent discharged into rivers and streams in Kenya.

Description of the study area

The study was conducted at Pan African Paper Mill (PPM) Webuye, which discharges an average combined effluent of 40,000 m^3 per day into River Nzoia. (Achoka, 1998; PPM Annual Report, 2001). The mill has an annual production capacity of 120,000 tons of paper. Pulping of three species of trees (*Pinus patula*, *Cupressus lusitanica* and *Eucalyptus saligna*) is achieved through Kraft cooking and stone grinding process. Bleaching sequences used are CEHH or CEHP, which contribute a large quantity of coloured compounds into the River Nzoia. PPM employs a biological wastewater treatment system, which includes two clarifiers, followed by four lagoons and the effluent is finally discharged into the river. Effluent treatment is a factor of major concern not only to PPM but also to the surrounding community, the Lake Victoria environmental management and the country as whole (Dilek and Bese, 2001; Kakai, 2000).

Materials and methods

Five sampling points were established at the exit of each of the components of the wastewater treatment system, which consists of a primary clarifier (SP1), first aeration pond (SP2), second aeration pond (SP3), first stabilisation pond (SP4) and second stabilisation pond (SP5). Wastewater samples were collected two days, every week in the morning and late in the afternoon for a period of three months. Samples obtained from these points were subjected to ELCAS treatment. Effluent characteristics were measured before and after ELCAS application. Figure 2 shows the process description for ELCAS. Effluent from each of the above sampling point was collected in collector tank (tank 1), it was then allowed to flow to the ash-mixing tank. The effluent was mixed with ash and allowed to settle. It was then allowed to flow by gravity into a treatment tank (tank 2) where iron electrodes connected to electrical power source were used to clarify the effluent. The clarified effluent was allowed to flow by gravity into another tank (tank 3) for effluent characteristics testing. Samples were tested for temperature, colour, turbidity, pH, biochemical oxygen demand (BOD), chemical oxygen demand (COD), total solids (TS), total suspended solids (TSS) and electrical conductivity (ELC) according to standards methods. Data collected were analyzed using Statistical Package for Social Scientists (SPSS) computer programme (version 12.0). The data were subjected to univariate analyses to determine the differences in measured parameters at $P \leq 0.05$. Where necessary Tukey statistical post hoc test was employed to separate means of those parameters.

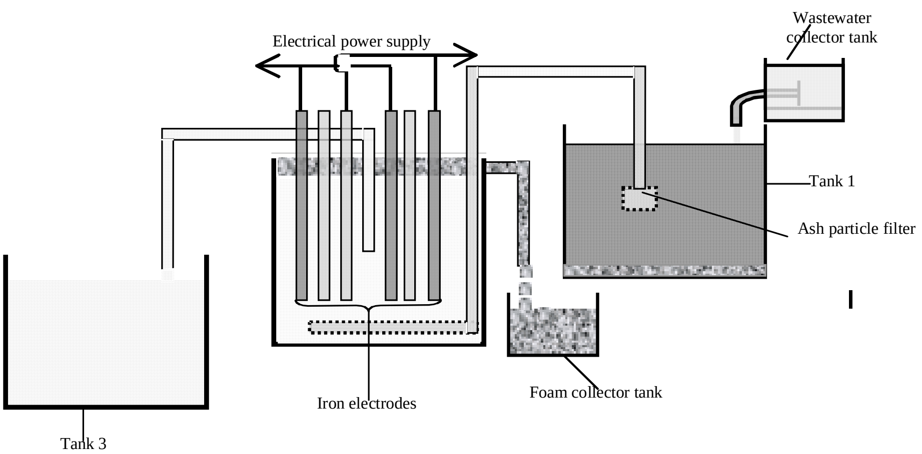


Figure 2: Process Description of ELCAS.

Results and discussion

Figure 3 shows the comparison of effluent characteristics between biological treatment process and ELCAS technique. The effluent from all sampling point showed a decrease in most of the parameters far below the required international effluent discharge standards into river and streams. COD was reduced by 80.66% compared to the biological treatment process of 49.5%. ELCAS technique reduced BOD by 81.25% compared with the biological treatment of 65.8%. Both BOD and COD were significantly different in all the sampling points. Generally, the big difference between COD and BOD in the biological treatment process indicates that there was more of non biodegradable material in the pulp and paper mill effluent that ends up into River Nzoia. High molecular compounds in a pulp and paper mill are not easily biodegradable, resulting into high pollution loads into the receiving waters. This depicts that biological treatment process is inadequate for a pulp and paper mill effluent. But application of ELCAS most of the hard-to-oxidize material were removed because of low final COD.

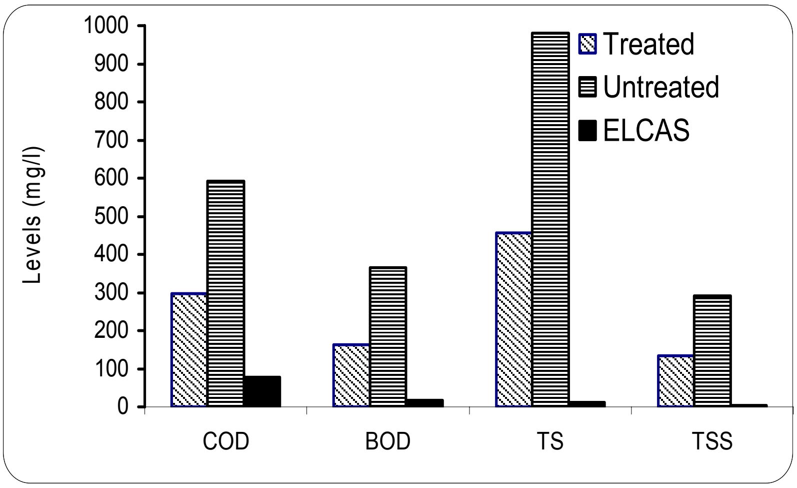


Figure 3: COD, BOD, TS and TSS for ELCAS, Treated and Untreated Effluent.

Greater percent (97.3%) of total solids (TS) were removed by ELCAS as compared with biological treatment method (53.3%). There was a significant difference in TS solids removed in all sampling points. This implies that the ELCAS receiving waters would not experience deposition and banking material as result of solid accumulation. Furthermore, total suspended solids (TSS) were highly reduced by ELCAS (94.9%) as compared to the biological treatment process (54.4%) employed by the pulp and paper. It was also noted that TSS removed by ELCAS was significantly different in all the five sampling point, which was not the case for the biological treatment method.

Figure 4 shows effluent colour, turbidity and electrical conductivity. The colour of the effluent increased in the biological treatment process by almost 95.2% whereas for ELCAS it was totally removed. Colour values in all the sampling point were significantly different in both ELCAS and biological treatment process. Increased effluent colour in the biological treatment system was probably due to the polymerization of wood by-products into phenolic metabolites of higher molecular size. It would therefore appear that more oxidation products were being formed as the effluent went through the treatment process, which is in agreement with the findings by Hasalam (1987). However, electrical conductivity increased significantly in the case of ELCAS but decreased through biological treatment process. The probable reason for increase in electrical conductivity was the introduction of some mineral elements from the ash. Ash was found to contain a number of elements which were not available in the effluent before treatment, but finally exhibited themselves in the treated effluent (Orori, 2003).

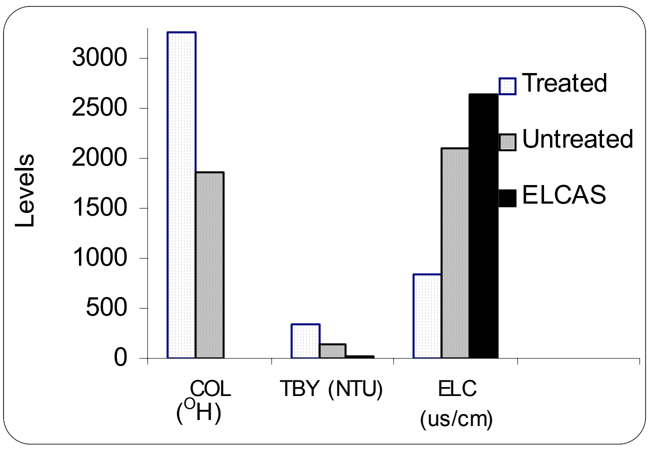


Figure 4 Colour, Turbidity and electrical conductivity of ELCAS, treated and untreated effluent.

Turbidity of wastewater was significantly reduced by ELCAS (96.4%) as compared with the biological treatment process (56.7%). A significant difference between values for ELCAS technique of all the sampling points and the biological treatment process was noted.

It was noted that pH of ELCAS treated water increased in all the points. ELCAS treated wastewater pH change was not significantly different from the sampling points. The increase in ELCAS treated effluent pH may be attributed to the addition of ash, which resulted in increased hydrogen ions. The increase in temperature of ELCAS treated water was not significant in all the sampling points.

Table 2: Major Elements in ELCAS, treated and untreated effluent.

Elements	(μg/l)		
	Untreated	ELCAS	Standard water quality guidelines*
Aluminum	400000.00	2250.00	200000
Arsenic	<10	0	50000
Barium	139.21	87.36	-
Bismuth	219.35	124.00	-
Cadmium	10	5	5000
Calcium	24500.00	17500.00	-
Chromium	250.55	100.00	50000
Cobalt	<10	0	100000
Copper	0	0	1000000
Iron	174.84	203.00	300000
Lead	0	0	500000
Magnesium	546.55	265.41	-
Manganese	805.81	451.80	10000
Potassium	250000.00	9000.00	-
Selenium	0	0	10000
Silicon	1050.68	564.41	-
Sodium	545600.00	440000.00	-
Titanium	0	0	-
Zinc	237.74	235.23	5000000

*Column was adopted from Chapman (1992)

Elemental analysis of ELCAS treated wastewater showed a significant increase and decrease in some elements concentration. Elements such as As, Cd, Co, Cu, Hg, Ti, Mn, Mo, Ni, Pb, Se, Sb, U, V, Zn found to be present in ash were noted in the ELCAS treated water, but in small quantities. However, electrical current flow coupled with wastewater coagulation reduced these metals far below their original wastewater concentration. Some metals increased in concentration above the level in the untreated wastewater such as chromium and iron. It should be noted that concentration of most metal elements in ELCAS treated wastewater were within the required limit for domestic water requirements (Chapman, 1992).

Recycling of ELCAS treated water

Table 3 shows the quality of ELCAS treated effluent compared with standard quality of water for fine paper grades. Analysis of ELCAS treated wastewater indicated that it is possible to recycle this water to various sections of a Kraft pulp and paper mill. All physico-chemical parameters of the treated wastewater were within the requirements for most of the paper grades, which finally will results, into low effluent discharge into water bodies (UNEP, 1981).

Table 3: ELCAS and Water Quality for Paper Making.

Properties	ELCAS	*Water Quality for paper making
Turbidity (NTU)	0	10-100
Colour (^{0}H)	0	5-30
Alkalinity (ppm as $CaCO_3$)	8.3	7.5-15
Chlorides (mg/l)	61	75-300
Iron (mg/l)	0.203	1.0 –0.1
Total Solids (mg/l)	26.87	200-300
Manganese (mg/l)	0.451	0.05-0.5
Magnesium hardness (ppm)	0.436	5
Calcium hardness (ppm)	1.090	5

**Column obtained:* TAPPI Standards: E 600 s-48, E 601 s-53, E 602 s-48, E 603 s-49

Mass of ash, electrical power consumption and electrodes mass loss

Mass of ash introduced, electrical power consumption and electrodes mass loss were the main aspects used to determine operational cost of ELCAS technique. These are shown in table 4. The quantity of ash introduced, increased to a maximum and started to decrease. The maximum point was indicated by the time necessary to pass current for total colour removal. Mass of ash was equal, but electrical power consumed was significantly different for all the sampling points. The increase in electrical power was determined by the change in time to pass current until total effluent colour removal was attained.

Power consumption increased significantly from SP1 to SP5. This indicated that the effluent was oxidized and hard-to-oxidize product, which might have been formed when the effluent passed through the biological treatment process. This has negative implication on the receiving waters because the biological treatment process may be increasing the toxicity of the effluent instead of decreasing. Electrode mass loss increased with time and the loss was significant in all sampling points. The increase in electrodes mass loss may have been caused by polymerization of compounds formed during the biological treatment process, which required longer time to eliminate them.

Table 1: Mass of ash and electrical power consumed by ELCAS technique.

Sampling point	Mass of ash (kg/m^3)	Power consumption (kw)	Electrodes mass loss (g/m^3)
SP1	0.833	1.121	0.133
SP2	0.833	1.466	0.234
SP3	0.833	1.696	0.300
SP4	0.833	1.955	0.380
SP5	0.833	2.156	0.488

Least power consumption was noted in ELCAS technique than the ones reported in electrochemical treatment experiments (Orori, 2003). This could be attributed to the catalytic properties of metal oxides such as MgO, Mn_2O_3, Cr_2O_3, PbO_2 in ash leachate. The catalytic properties of these metal oxides on the surface, or in the space between the anode and cathode has been recognized in past studies and might have assisted in the reduction of the time for current flow as reported by Ahonen (2001). Ash provided these metal oxides, which in solution changed to their respective hydroxides, resulting in the reduction of power consumed. Also the presence of several metal ions in the ash leachate probably reduced iron electrodes polarization and increased wastewater ionisation resulting into fast electrons transfer and lower power consumption with ELCAS.

Operational cost analysis

Operational cost per year was determined on the basis of ash addition, electrical power consumption and electrodes mass loss. These were found to be the major basis for determining operational cost. No cost was attached to ash leachate, since PPM produces on average 7 metric tones of ash per day. ELCAS recorded the least operation cost of US$2705.78, which increased to US$8505.93 per year.

Conclusion and recommendations

The best method to treat the effluent of an integrated pulp and paper mill was found to be ELCAS. Total operational cost of this system would average between US$2,705.78 to US$8,505.93 per year. The biological treatment method indicated higher operational cost in comparison with ELCAS. ELCAS will also allow the recycling of treated wastewater for use in some parts of the factory. Most of the physico-chemical parameters of the treated wastewater will fall within international standards and national standards. ELCAS reduced colour by 100% and BOD, COD, TSS, TS and turbidity were reduced by 81.25%, 80.66%, 94.90%, 97.26% and 92.14% respectively. The pH of the treated water by ELCAS was within recommended range, even though it appeared to increase. These break through will finally, conserve the water resources from River Nzoia and integrity of Lake Victoria and its environs.

It is recommended that PPM should set up an ELCAS full time plant and compare its cost with the present treatment costs. Large quantities of wastewater discharged to the river would need to be recycled if ELCAS is applied because it meets quality standards required for use in a Kraft pulp and paper mill. However, its necessary to investigate the long-term toxicity of ELCAS treated wastewater on aquatic organisms.

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References

Achoka, D. J., 1998. *Physico-chemical Parameters of Liquid Effluent from Pan African Paper Mills in River Nzoia*. Ph.D. Thesis Moi University.

Ahonen, H., 2001. *A Review of General Information on Electrochemical Process Wastewater Treatment.* Finland Laboratories 7p.

Alloway, B. J., and Ayres D.C., 1993. *Chemical Principles of Environmental Pollution*. Blackie Academic and Professional Pub., London, 29p.

Chapman, D., 1992. *Water Quality Assessment: A Guide to the Use of Biota, Sediments and Water in Environment Monitoring*. Chapman Hall Publishers. London. pp. 585.

Dilek, F. B. and Bese, S., 2001. Treatment of Pulping Effluents by using Alum and Clay–Colour Removal and sludge Characteristics in Water. *SA* Vol. 27 No. 3, pp. 361.

Fifield, F. W., and Kealey, D., 1990. *Analytical Chemistry 3rd edition*. Blackie Academic and Professional Press, London.

GOK, (2000). An Economic Survey of Kenya for 1999/2000, Government Printers, Nairobi, Kenya.

Halsalm, E., 1987. Twenty-second Procter Memorial Lecture: Vegetable Tannins - Renaissance and Reappraisal. *Journal of the Society of Leader Technologists and Chemists*, Vol. 72, pp. 45-64.

Hilleke, J., 1991. *Industrial Environmental Control.* In Springer, A. M., ed. 2nd edn. Tappi Press. Atlanta. pp. 305-343.

Mahida, U. N. ed., 1981. *Water pollution and disposal of wastewater on land.* New Delhi, 335 p.

Mall, I. D., Upadhya, S. N., Singh A. R., and Upadhya, Y. D., 1989 Environmental Pollution due to Pulp and paper industry: *In*. *Silver Jubilee International Seminar and Workshop Appropriate Technology for Pulp and Paper Manufacture in Developing Countries*, New Delhi, pp. 1 - 17.

Orori, O. B., 2003. Colour Removal from Wastewater of a Pulp and Paper Mill in Kenya by a Combination of Electrochemical and Coagulation Methods. MPhil. Thesis, Dept. of Wood Science and Technology, Moi University, Eldoret , Kenya, 146p.

Situma, L. M., 2003, *Modelling of Aerated Lagoons at PanAfrican Paper Mills Webuye*. MPhil. Thesis, Dept. of Wood Science and Technology, Moi University, Eldoret , Kenya, 126p.

PPM,2001. PPM Annual Report. (Unpublished)

Springer, A., M., Vincent, C. H., and Timothy, S. J., 1995. Electrochemical Removal of Colour and Toxicity from Bleached Kraft Effluents. *Tappi J.,* Vol. 78 No. 12, pp. 85-91.

Tappi Standards, 1992. W*eighing, Sampling and Testing Pulp Moisture Content: Tappi Test Methods.* Tappi press, Atlanta, Georgia USA. T210.

UNEP, 1981. *Environmental Management in the Pulp and Paper Industry*, Vol. 1, 234p.

Impact of agro-industrial activities on the water quality of River Nyando, Lake Victoria Basin, Kenya

Raburu P. O.[1] and Okeyo-Owuor J.B.[2]

[1]Department of Fisheries and Aquatic Sciences, Moi University. P.O. Box 3900, Eldoret, Kenya. Email:praburu2002@yahoo.com

[2]School of Environmental Studies, Moi University, Box 3900, Eldoret, Kenya.

Abstract

The impact of agro-industrial activities on the water quality within River Nyando, Lake Victoria Basin was studied at different hierarchical levels between August 1997 and June 1999. Triplicate water samples were collected on a monthly basis from various stations ranging from the source to the mouth of the river and analyzed for selected physical and chemical water quality parameters using standard analytical methods. Statistical analysis was performed using MINITAB and STATISTICA computer packages.

Agricultural land use was found to be the major factor contributing to changes in water quality. Salinity and pH varied at river basin scale, turbidity, TDS and conductivity at catchment scale, while DO, alkalinity and pH at sub-catchment and river reach scale. The nutrient loads increased downstream. Anthropogenic sources contribute to high levels of nutrients within the basin. The changes recorded in water quality along the river were comparable to the modified Index of Biotic Integrity (IBI) and Nyando Habitat Evaluation Index (NHEI) derived for the river during the same period of study.

Findings of this study can be used to design measures for mitigating and monitoring environmental impacts arising from agro-industrial activities within the Lake Victoria Basin. The study recommends a comprehensive Nyando River Basin Management Programme to address the multiple issues environmental within the basin.

Key words: water quality, agro-industrial activities, land use

Introduction

Rivers play a significant role within a landscape as they are at the receiving end of all the human activities within their catchments. To get a true reflection of what happens within the catchment of a river basin either through point- or non-point sources of pollution, studies of spatial and temporal changes in water quality are very important. Non-point source pollution arises from sources that are normally associated with agricultural and silvicultural, and human activities within the catchment. Non point sources such as nutrients, pesticides, heavy metals and sediments are transported from land by atmospheric, surface water and ground water pathways (Nikolaidis *et. al.,* 1998).

The character of streams and rivers reflect an integration of physical and biological processes occurring in the catchment. Landscape properties that contribute most directly to the structure and function of adequate systems include prevailing climate, catchment and riparian landuse or cover patterns, channel slope and aspect, quartenary and bedrock geology, and hydrography (Richards *et. al.,* 1977; Townsend *et. al.,* 1977). Catchment processes operate at different hierarchical levels and therefore analysis of catchment attributes should be studied at different hierarchical scales. The appropriate focal level or scale of observation is defined by boundaries of the system under study, where scale represents either temporal or spatial dimension (Johnson and Gage, 1977).

Landscape form and composition play a major role in regulating stream chemistry. Landscape-level processes define the overall supply of elements to a stream and provide the framework within which other processes operate on smaller spatial scales and shorter temporal scales to regulate supply and availability (Meyer *et. al*., 1988). The availability and cycling of nutrients within a riverine environment is very important in the functioning of the ecosystems. Analysis of the changes in river water quality often reveals the impact of different activities taking place within a river basin. The landscape influences its water bodies through multiple pathways and mechanisms, operating at different spatial scales. A spatial conceptualization of aquatic ecosystems suggests a hierarchical organization of physical units, perhaps most clearly captured for rivers in hierarchy: habitat - reach, - segment – sub-catchment - basin (Hawkins *et. al.,* 1993) and in the nested classification of stream order (Strahler, 1964).

Human activities are responsible for fundamental changes to riparian vegetation of stream catchments around the world. For instance conversion of natural forest to agriculture may influence stream water chemistry (Maasdam and Smith, 1994), discharge (Gustard and Wesselink, 1993), water temperature (Hanchet, 1990), characteristics of the channel (Sweeney, 1993) and inputs of radiant energy and organic matter. Deforestation can increase light levels, and thus enhance algal productivity (Ulrich *et. al.,* 1993) but decrease inputs of woody debris, leaves and other coarse particulate organic matter (Evans *et. al.,* 1993). Changes in land use that affect riparian vegetation can, therefore, be expected to have important consequences for the abundance of invertebrates and fish as a result of changes in physico-chemistry of stream water.

Studies within the Saginaw River Basin revealed that nitrogen concentrations, alkalinity and TDS were more sensitive to agricultural land use during the summer and to underlying geology during autumn (Johnson *et. al*., 1997). In the same catchment, land use within the riparian region and throughout the catchment was equally effective predictors of total nitrogen, nitrate, orthophosphate and alkalinity. However, total phosphorus and TSS were better explained by land use within the riparian region than by catchment-wide variables, indicating the dominance of local controlling mechanisms while Ammonia and TDS exhibited the opposite pattern indicating regional control. Strong relationships between land use and nutrient concentration or export have also been observed for phosphorus and nitrogen (Omernick, 1976; Peterjohn and Correll, 1984; Osborn and Wiley, 1988). Increased export of nitrite-nitrate and orthophosphate was observed in predominantly agricultural and urban catchments as compared to forested catchments. According to Osborn and Wiley (1988), soluble reactive phosphorus concentrations were elevated downstream of urban centers, particularly during low flow periods unlike nitrate concentrations which were elevated during late winter and early spring in response to agricultural activities and during summer and autumn as a result of urban runoff.

Nyando River is 153 Km long and its basin covers a catchment area of 3,450 Km^2 and drains into the Winam Gulf of Lake Victoria which lies between longitude 34^o 13' and 34^o 52' East and latitude 0^o 4' and 0^o 32' South of the equator. The river originates in the highlands at an altitude of 1,700 m.a.s.l and terminates at the lakeshore swamps at an elevation of 1,135 m.a.s.l. Studies on the impact of land use on the water quality of riverine environments within the Lake Victoria Basin are scarce. A knowledge gap exists, as water quality managers do not have adequate data on which to base their management practices. The paper describes a study carried out to determine the changes in the water quality of Nyando River in relation to different agro-industrial activities within the Basin.

Materials and methods

The study was carried out from August 1997 to June 1999. Changes in water quality were examined at various hierarchical levels The highest hierarchy, the river basin scale compared impact of environmental variables on water quality 40 kilometers from the river mouth along River Nyando and the lower reaches of Sondu-Miriu Basins.. This involved stations NRU, NRW, NRX and NRY within the Nyando and stations SMR, SMS and SMT within the Sondu- Miriu River Basins. Changes at the river catchment scale were examined in all sampling stations along River Nyando from the source (Station NRR) to the river mouth (Station NRY). The impact of non-point pollution sources was examined at the sub-catchment scale by comparing the water quality above and below coffee, tea and sugarcane land use practices within the basin. Point source pollution was investigated at the river reach level by sampling above and below effluent discharge points at stations NRS, NRT, NRU and NRK (Figure 1).

Three replicate measurements and water samples were collected on a monthly basis at every sampling station. The physical water parameters temperature, conductivity, total dissolved solids (TDS), turbidity, salinity and pH were measured *in situ* using conductivity / TDS meter, turbidimeter and pH meter respectively.

Standard water quality handling and analytical methods (APHA, 1994) were used to determine the chemical water quality parameters and nutrients. The nutrients determined included phosphate-phosphorus PO_4^{3-}-P, nitrate-nitrogen NO_3^--N, Ammonium-nitrogen NH_4^+-N and Silicates. Physico-chemical parameters determined include Dissolved oxygen (DO), alkalinity, hardness, pH, temperature, conductivity, total dissolved solids (TDS), turbidity and salinity.

Data analysis

Statistical data analysis was done using MINITAB release12.1 and Microsoft EXEL computer packages. In all the analyses, 95% level of significance was used as the critical point. General Linear Model Analysis of Variance (ANOVA) and two-way ANOVA was used to test for similarity of water quality between the river basins, different land use practices and above and below sources of pollution both at the catchment and sub-catchment levels. For temporal and spatial variation, Tukey's Multiple Range Test was performed to delineate which particular observations were significantly different from the others.

Figure 1. Map of Nyando and Sondu-Miriu Basins showing the location of sampling stations along the rivers.

Results

River basin scale

A comparison of conductivity shows that all stations in Sondu-Miriu are similar but differs significantly ($F=124.44$, df(11,518), $p<0.0001$) from all the other sampling stations in River Nyando except for station NRD. The concentration of PO_4^{3-}-P in all stations along River Nyando and the loads entering the lake were significantly higher in the Sondu – Miriu River ($F=9.48$, df(11,497), $p<0.0001$). Apart from stations NRR (5.07 $\pm$ 0.45 mgL^{-1}) and NRY (3.53 $\pm$ 0.29 mgL^{-1}), the mean concentration of nitrate-nitrogen along River Nyando was significantly higher than stations in the lower reaches of Sondu-Miriu ($F=11.27$, df(11,467), $p<0.0001$) but the load entering the lake was higher from Sondu-Miriu. No significant difference was noticed in the mean concentration of ammonium-nitrogen between Nyando and Sondu-Miriu River. The two river basins however differed in their silicate concentrations ($F=6.34$, df(11,434), $p<0.0001$).

River catchmnet scale

Figure 2 shows the spatial variation of water quality along River Nyando.

Conductivity ($F = 124.44$; df = 11,518; $P<0.0001$), TDS ($F = 95.82$, df =14,470), $P<0.0001$) and turbidity ($F=21.55$, df=(11,461), $P<0.0001$) of the water varied significantly at different sites along the river. Whereas TDS decreased, turbidity increased downstream. The variation in the concentration of nutrients during the study period at stations NRR, NRT, NRW and NRY (Figures 3 to 6) indicates that the amount of nutrients increases downstream.

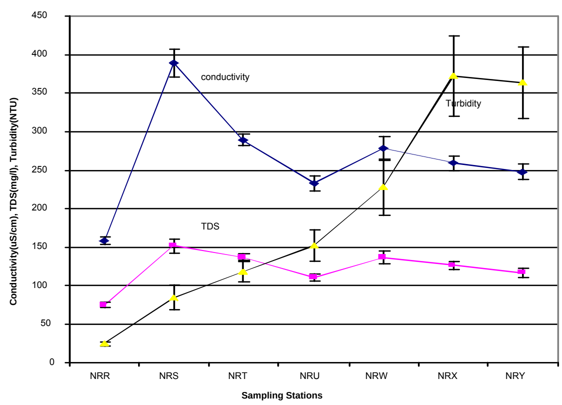

Figure 2. Variation in conductivity, turbidity and TDS in River Nyando from the source to the mouth.

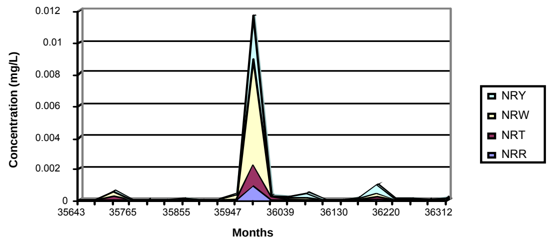

Figure 3. Variation in the concentration of NH_3-N along River Nyando.

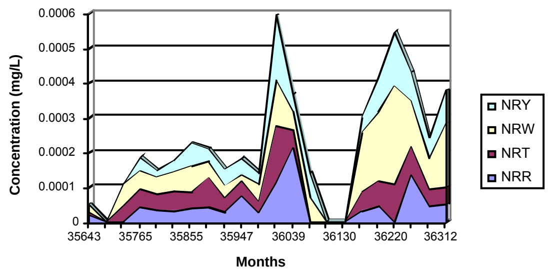

Figure 4. Variation in the concentration of NO_3-N along River Nyando.

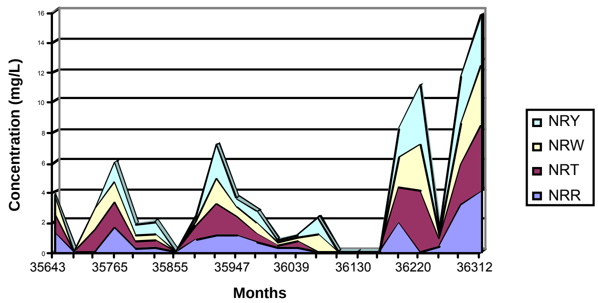

Figure 5: Variation in the concentration of Silicates along River Nyando.

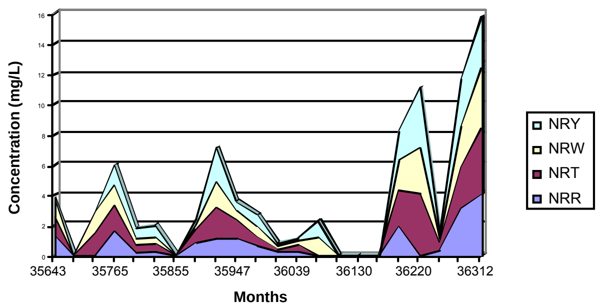

Figure 6. Variation in the concentration of PO_4-P along River Nyando.

River sub-catchmnet scale.

(i) Coffee Zone

Significantly poorer water quality occurred below the coffee-growing zone as compared to sites above. Phosphate-phosphorous concentration which showed no variation between the two sites, significantly varied with time (F = 74.67.df (15.95), P<0.001). The same was true with silicates (F = 52.74, df (13,83), P<0.001). The concentration of ammonium-nitrogen (F=153.33, df (14,89): P<0.001) and nitrate-nitrogen (F = 16.18, df (14,89); P<0.001) also showed a strong seasonal variation during the study period.

(ii) Tea Zone

Significant variation in alkalinity (F = 8253.47, df (1,29), P<0.001), turbidity (F = 9.21, df (1,17), P<0.01), conductivity (F = 15936.23, df (1,29), P<0.001), pH (F = 28.51, df (1,29), P<0.001) and BOD (F = 5.14, df (1,23), P<0.03) was recorded between the sub-catchments with and without tea plantations. It is important to note that most water quality parameters like conductivity, TDS, pH, turbidity and Ammonium-nitrogen were higher in the reference sites that in sub-catchments with tea plantations. The nutrients phosphate-phosphorous (F = 26.17, df (1,23), P<0.001), silicates (F = 56.72, df (1,17), P< 0.001) and nitrate-nitrogen (F = 32.18, df (1,17), P<0.001) were significantly higher in the tea zone..

(iii) The Sugarcane Zone

The values of water quality parameters below the sugarcane zone were significantly higher than areas without sugarcane except for the nitrates (F=112.73, df(1,35), p<0.0001) which was significantly higher in the latter station. There was however no significant difference in the pH between the two sites.

Impact of Industrial effluent

Table 1 gives the changes in water quality above and below Muhoroni Sugar Company (MSC) and Agro–Chemical and Foods Company (ACFC) The water quality was poorer below than above the factories. Of the variables analyzed only the alkalinity, nitrate-nitrogen and silicates had significantly higher concentrations in the station above the factories. Apart from phosphate-phosphorus the water quality variables varied significantly with time during the study period.

A comparison of the water quality below Chemelil Sugar Company (CSC) and MSC / ACFC (Table 2) revealed that the impact of the effluents on water quality was significantly different. All the nutrients had higher concentrations below Muhoroni than Chemelil Factory except for the silicates and all the water quality parameters analyzed varied significantly with time.

Table 1. Mean values of physico-chemical parameters above and below Muhoroni Sugar Company and Agrochemicals and Food Company

PARAMETER	Mean Above	Mean Below	P-Value
	MSC/ACFC	MSC/ACFC	
Conductivity (u/cm)	232.54	278.26	<0.0000
Alkalinity (mg/L)	136.02	130.23	<0.0000
Turbidity (NTU)	152.61	227.92	<0.0000

TDS (mg/L)	110.07	136.61	<0.0000
Salinity	0.11	0.14	<0.0000
DO (mg/L)	6.32	6.61	<0.01
BOD (mg/L)	3.12	2.60	<0.0000
Silicates (mg/L)	1.48	1.27	<0.0004
NH4- N (mg/L)	0.0003	0.0005	>0.42
NO3-N (mg/L)	6.06	5.17	<0.0002
PO4-P (mg/L)	0.000005	0.000005	>0.05

Table 2: Comparison of mean values of Physico-chemical parameters below Chemelil Sugar Company and MSC / ACFC during the study period

PARAMETER	Mean Below CSC	Mean Below MSC/ACFC	P-Value
Conductivity (u/cm)	276.85	285.08	<0.0000
Alkalinity (mg/L)	156.21	140.96	<0.0000
Turbidity (NTU)	213.79	190.35	>0.1
TDS (mg/L)	132	140	<0.0000
Salinity	0.12	0.14	<0.0000
DO (mg/L)	7.11	6.47	<0.0000
PH	7.55	7.43	<0.0000
Silicates (mg/L)	1.61	1.27	<0.0000
NH4-N (mg/L)	0.00014	0.0005	>0.1
NO3-N (mg/L)	4.66	5.08	<0.0000
PO4- P (mg/L)	0.00007	0.00009	<0.0000

Discussion

River basin scale

Most of the physico-chemical parameters were higher in the Nyando as compared to the Sondu-Miriu River Basin. High levels of salinity, alkalinity, TDS, conductivity and turbidity in Nyando could be attributed to poor agricultural practices and high use of agro-chemicals. The similarity of the water quality parameter at station SMR on the Sondu-Miriu to those of the upper reaches of River Nyando shows that the former is not as degraded as the latter.

The amount of nutrients in Nyando was also found to be higher than in Sondu-Miriu except for NO^{3-} -N. This could be attributed to several point and non - point agro-industrial activities in the Nyando Basin which pollute the river. However, higher concentrations of nitrates in Sondu-Miriu as compared to Nyando could be attributed to the high amounts of nitrate- based fertilizers used in maize plantations in the former catchment. Considering the nutrient loads, it emerges that Apart from PO_4-P, the nutrient loads of all the other nutrients are higher at the Sondu-Miriu Rivermouth than Nyando. This is due to a greater discharge as compared to Nyando. The higher concentration of nitrate could however be attributed to speciation of the nitrogen species that has been found to differ according to agricultural land use. Cooke and Prepas (1998) found nitrate-nitrogen to be the dominant nitrogen species in runoff from cropland while ammonium-nitrogen predominated in runoffs from mixed agricultural catchments. The situation in the study area is therefore not unique as the sugarcane zone in the Nyando catchment is an area with mixed agricultural practices as opposed to the Sondu-Miriu that is predominantly maize crop.

River catchment level

The variations of water quality within the Nyando River at the catchment level are closely related to the land use practices within the catchment. Conductivity and TDS for instance were lowest in the reference station at the source of the river. These parameters differed quite significantly in sub-catchments with intense agro-industrial activities along the river. Stations in less impacted sub-catchments had lower concentrations as compared to stations located in areas with high human population, agricultural and industrial activities. The waters of the river were also more turbid at stations in the middle reaches of the river and stations in the lower reaches. The land use within this steep valley is located in areas dominated by intensive large scale and subsistence agriculture where both cash and food crops are grown in the riparian zones to the riverbanks without a buffer zone. The same areas occur in the floodplains with very high human and livestock population density and a rather drier climate. This coupled with unstable riverbanks characteristic of this area exposes the soils to erosive forces thus heavy siltation.

The amount of dissolved oxygen was lowest in stations located downstream of industrial and municipal effluent discharge points. The decomposition of organic substances from the industries and urban centers contributes to the low levels of dissolved oxygen in the water. Low dissolved oxygen levels at station NRY, however could be attributed to reduced water speed or lack of turbulent water movements that could increase the oxygenation at this floodplain station.

The pH, salinity and alkalinity of the river water did not vary much throughout the catchment. This could be explained by the fact that the basement rocks within the entire catchment are basically the same contributing to similar acidity-alkalinity levels. Changes seen at the sub-catchment and river reach levels however are due to anthropogenic sources.

The concentration of nutrients within the catchment of the river basin originates more from diffuse as opposed to point sources. Although the levels of phosphates and nitrates are higher at stations, associated with industrial effluents, the land use upstream these stations is also composed of large and small-scale agricultural farms with huge inputs of agro-chemicals. Minimal levels in reference station and the general increase in the amount of nutrients downstream supports the argument on the predominant contribution of diffuse nutrient sources within the catchment.

River sub-catchment scale

Changes in water quality at the sub-catchment scale in the Nyando Basin are influenced by land use. Poorer water quality was recorded below the coffee-growing zone both temporally and spatially. It is important to note that whereas pH showed no variation at the catchment level, at sub-catchment the parameter was significantly higher. The concentration of phosphate- phosphorus and Silicates did not vary above and below the coffee zone but temporally the variation was significant. Nitrate- and Ammonium-nitrogen however differed both in space and time. This could be attributed to the predominant use of nitrogen-based agro-chemicals in coffee farms and organic wastes generated from the coffee industries within this sub-catchment. Temporal variation of all the nutrients within the sub-catchment indicates that these nutrients are continually generated from this zone throughout the year. Apart from agricultural activities, this is a transition zone between the upper and middle catchment with good vegetation cover. However deforestation rate is increasing and will enhance degradation in this area. The high level of nitrate-nitrogen concentration in this zone is comparable to the findings of Havel *et al.,* (1999) who recorded highest stream water nitrate concentrations in recently manipulated areas of the catchment. He attributed this to nitrification and decomposition of organic matter in deforested and transitional zones, a reason that could well apply to the study area.

Of all the land use practices in the basin tea growing appears to have minimal impacts in the river water quality. The occurrence of low TDS, conductivity and turbidity below the tea-growing zone could be ascribed to the provision of adequate vegetation cover around the farms that reduces the soil erosion within these sub-catchments. High levels of nitrate nitrogen and silicates can however be attributed to the use of agro-chemicals with nitrates and silicates as the major constituents. Silicates could also arise from non anthropogenic sources.

The high conductivity, TDS, turbidity and alkalinity in sub-catchments where sugarcane is grown is a direct consequence of high population density, poor agricultural practices and lack of buffer zone along the river. It is common in this sub-catchment to find farms cultivated right up to the riverbanks. This coupled with multiple tillage and huge input of agro-chemicals in the sugarcane farms contributes largely to elevated levels of these parameters. The high levels of nutrients in this sub-catchment can also be attributed to the same reasons above. However, high levels of nitrate- nitrogen in sub-catchments without sugarcane can be explained by the use of fertilizers where there is large-scale growing of maize.

Several spatial and temporal patterns observed in streamwater chemistry in most catchments are controlled by land use history and hydrology which controls the storage and transport of constituents (Havel *et al.*, 1999). The relative contribution of point or non-point sources depends on the source of the nutrients in respective catchments. Some studies have reported the predominance of point sources (Sundblad *et al.,* 1994) but in Nyando the situation appears different. The high contribution of nutrients from diffuse as opposed to point sources in the Nyando Basin however, parallels several other studies that have linked nutrient loadings to catchment land use, and particularly agriculture (Omernick *et al.*, 1991, Jaworski *et al.*, 1992, Johnson *et al.*, 1997, Moreau *et al.*, 1998). Agricultural land use within the Nyando River Basin is therefore a major contributing factor to the variation in the water chemistry and quality.

River reach scale

Industrial wastewater discharged into River Nyando has varied impacts on the river's water quality at the river reach level. Elevated levels of conductivity, TDS, and turbidity below ACFC / MSC can be closely linked to the industrial effluent from the factories, particularly to ACFC whose effluent has a characteristic persistent brown color. These changes can not wholly be ascribed to point sources as the area is within the sugarcane growing zone with potential diffuse pollution sources. Whereas pH, salinity, alkalinity and dissolved oxygen did not vary significantly at catchment scale, their variation seems to be more significant at the river reach level as can be seen in reaches below the industries.

The contribution of nutrients from ACFC/ MSC to the river system does not seem to be significant at this river reach possibly due to high contribution of nutrients from Muhoroni Township in the vicinity of the factories upstream. However, poor water quality between this reach and below Chemelil Sugar Company could be attributed to the efficiency of the treatment facility at Chemelil. During the study period, the treatment facility at ACFC was prone to frequent breakdowns and that of Muhoroni Sugar Factory was non-functional. The treatment facility at Chemelil was however well maintained with a provision of wetlands within the system, which was more efficient in the uptake of nutrients.

Pollution of inland water bodies by industrial and municipal wastewaters is a common phenomenon in developing countries (Gosh and MCbean, 1996). Whereas the effect of industries was not significant at catchment scale, it was however very significant at river reach scale. Point sources of pollution affect water quality due to high organic and nutrient pollution. High levels are generally recorded below effluent discharge points and reduces downstream due to the rivers self cleansing capacity (Markantonatos *et al.*, 1995). In the Nyando Basin, there are several point pollution sources spread along the river, particularly at Lelu, Fort Ternan, Muhoroni, Chemelil, and Ahero. These do not allow the river to recover adequately from industrial effluents. Reduction of pollutants from River Nyando to Lake Victoria therefore relies greatly on the effectiveness of the rivermouth wetlands.

Seasonality

The water chemistry in the Nyando basin varied temporally even in cases where no significant variation was noticed among sub-catchments or river reaches. Seasonal variation is however not a universal phenomenon. Reporting lack of distinct seasonal variation in the water chemistry, Topalian *et al.*, (1999) ascribed this to regular and intermittent pollution pulses. Seasonality in the concentration of nutrients, particularly in agricultural areas is however common with peaks being recorded during high flow periods (Probst, 1995, Sundblad *et al.*, 1994, Markantonatos *et a.l*, 1995, Moreau *et al.*, 1998). High flows usually occur when fields lack vegetation cover and nitrate-nitrogen for instance easily leached from the soil. Inorganic nitrogen forms are subject to biological transformations that increase with increasing temperature (Davies and Keller, 1983). Seasonality in nutrient levels can be attributed to the deposition in the sediments during low flow periods and resuspension and transportation during high flow periods.

References

APHA, 1992. *Standard Methods for the Examination of Water and Wastewater, 18th Edition*. American Public Health Association, Washington D.C.

Cooke, S.E. and E.E. Prepas 1998. Stream Phosphorus and Nitrogen Export from Agricultural and Forested Watersheds on Boreal Plain. *Canadian Journal of Aquatic Sciences*, 55:2292-2299.

Davis, J.S. and H.M. Keller, 1993. Dissolved Loads in Streams and Rivers - Discharge and Seasonally Related Variations. *In*. B.W. Webb ed. *Dissolved Loads of Rivers and Surface Water Quantity/Quality Relationships.* IAHs Publication No. 141, International Association of Hydrobiological Sciences, Washington.

Garcia-Ruiz, R., S.N. Pattinson and B.A. Whitton (1998). Dnitrification in River Sediments: Relationship Between Process Rate and Properties of Water and Sediment. Freshwater Biology 39:467-476.

Ghosh, N. C. and E. A. Mcbean, 1996. Water Quality Modelling of the Keli River, India. *Water, Air and Soil Pollution,* 102:91-103.

Gustard, A. and A.J. Wasselink, 1993. Impact of Land-use Change on Water Resources: Balquhidder Catchments. *Journal of Hydrology*, 145:389-401.

Hanchet, S.M., 1990. Effect of Landuse on the distribution and Abundance of Native Fish in Tributaries of the Waikato River in the Hakarimata Range, North Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 24:159-171.

Havel, M., N.E. Peters and J. Carny, 1999. Longitudinal Patterns of Stream Chemistry in a Catchment with Forested Dieback, Czech Republic. *Environmental Pollution*, 104:157-157.

Hawkins, C.J., J.L. Kerschner, P.A. Bisson, M.D. Bryant, L.M. Decker, S.V. Gregory, D.A McCoullogh, C.K.Overton, G.H. Reeves, R.J. Steedmand and M.K. Young, 1993. A Hierarchical Approach to Classifying Stream Habitat Features. *Fisheries* 18:3-11.

Jaworski, N.A., P.M. Groffonan, A.A. Kellar and J.C. Prayer, 1992. A Watershed Nitrogen and Phosphorus Balance: The Upper Protomac River Basin. *Estuaries* 15:83-85.

Johnson, L.B., C. Richards, G.E. Host and J.W. Arthur, 1997. Landscape Influences on Water Chemistry in Midwestern Stream Ecosystems. *Freshwater Ecology* 37:193-208.

Johnson, L.B. and S.H. Gage, 1997. Landscape Approach to the Analysis of Aquatic Ecosystems. *Freshwater Biology* 37:113-132

Jonardan, K.G. and D.J. Schaeffer, 1975. *Illinois Water Quality Inventory Report.* Illinois Environmental Protection Agency, Springfield, Illinois, U.S.A.

Markantonates, P.G., N.C. Bocalis and M.O. Angelidis, 1995. Pollution Control in the Catchment Basin of River Evrotas, Greece. *Water Science and Technology*, 32 (9-10):247-255.

Meyers, L.S., T.F. Thuemler and G.K. Kornely, 1992. Seasonal Movements of Browntrout in Northern Wisconsin. *North American Journal of Fisheries Management* 12:433-441.

Moreau, S.G. Bertru and C. Buson, 1998. Seasonal and Spatial Trends in Nitrogen and Phosphorus Loads to the Upper Catchment of the River Valaine (Brittany): Relationships with Land Use. *Hydrobiologia* 373/374:247-258.

Newbold, J.D. 1992. 'Cycles and Spirals of Nutrients;, in Calows, P. and Petts, G.E., eds. T*he River Handbook. Hydrological and Ecological Principles*, Vol. 1. Blackwell Scientific Publications, Oxford. pp. 379-408.

Omernick, J. M., 1976. *The Influence of Land Use on Stream Nutrient Levels.* EPA-600/3-76-014. US Environmental Protection Agency, Washington DC.

Omernik, J.M., A.R. Abernathy and L.M. Male, 1991. Stream Nutrient Levels and Proximity of Agricultural Forest Land to Stream: Some Relationships. *Journal of Soil and Water,* 36:227-231.

Peterjohn, W.T. & D.L. Correll, 1984. Nutrient Dynamics in an Agricultural Watershed: Observations on the Role of a Riparian Forest. *Ecology*, 65:1466-1475.

Probst, J. L., 1985. Nitrogen and Phosphorous Exportation in the Garonne Basin (France). *Journal of Hydrology* 76:281-305.

Richards, C., R. J. Havo, L.B. Johnson and G.E. Host, 1997. Catchment and Reach Scale Properties as Indicated of Macroinvertebrate Species Traits. *Freshwater Biology* 37:219-230.

Sundblad, K., A. Tonderski and J. Rulewski, 1994. *Nitrogen and Phosphorus in the Vistula River, Poland - Changes from Source to Mouth Water Science and Technology* 30(5):177-186.

Sweeney, B.W. 1992. Streamside Forests and Physical, Chemical and Trophic Characteristics of Predmont Streams in Eastern North America. *Water Sciences and Technology.* 26:2653-2673.

Sweeney, B.W., 1993. *Effect of Streamside Vegetation on Macroinvertebrate communities of White Clay Creek in Eastern North America.* Proceedings of the Academy of Natural Sciences of Philadelphia 144:291-340.

Topalian, M.L., M.G. Rovedatti, P.N. Castane and A. Salibian, 1999. Pollution in Lowland River System. *A Case Study: The Reconquista River (Buenos Aires, Argentina)*. Water, Air and Soil Pollution 114:287-302.

Townsend, C.R., C.J. Arbuckle, T.A. Crowl and M.R. Scarsbrook, 1997. The Relationship between Landuse, Physicochemistry, Food Resources and Macroinvertebrate Communities in Tributaries of the Taieri River, New Zealand: A Hierarchically Scaled Approach. *Freshwater Biology* ,37:177-191.

Ulrich, K.E., T.M. Burton and M.P. Oemka, 1993. Effect of Whole-tree Harvest on Epilithic Algal Communities in Heedwater Streams. *Journal of Freshwater Ecology 8:83-92.*

Integrated water pollution assessment in data and resource poor situations

Peter Scheren[1]

[1]Eindhoven University of Technology, Eindhoven, the Netherlands. Address for correspondence: UNEP/ROA, Room A-130, P.O.Box 47074 Gigiri, Nairobi, Kenya

E-mail: peter.scheren@unep.org,

Abstract

It is a generally acknowledged fact that problems related to the pollution of water systems are complex and, consequently, the assessment and development of strategies for abatement of such problems requires an integrated approach. In practice, however, there are surprisingly few cases where such approach is effectively implemented: Many research activities, including policy-support research, still focus on isolated disciplines, on specific aspects of a problem, or on parts of a water system only. This is particularly so in the case of international waters, where political obstacles often hamper the implementation of transboundary assessment and policies.

This research proposes and tests an integrated approach to water pollution assessment and policy development, based upon the Driving Forces – Pressure – State – Impact – Response (DPSIR) framework. While the DPSIR framework, and related frameworks, is and has been used in policy supporting research, such as the Transboundary Diagnostic Analysis (TDA) studies in many International Waters projects undertaken by the Global Environmental Facility (GEF), the framework is mostly used as a static tool for the identification of environment cause-effect relationships. The innovative part of this study is the application of this framework as a basis for a dynamic decision support model. The emphasis therein is on quantifying and modeling the cause-effect relations in order to assess the implications of policy scenarios on the water system, including its socio-economic implications. The system has been tested on a number of international water systems, including Lake Victoria in East Africa and the Ebrié Lagoon in West Africa. The approach has proven to be very useful in situations of relative data and resource poorness, in particular when combined with an uncertainty analysis.

Key words: Ebrié Lagoon, Lake Victoria, Integrated Environmental Pollution Assessment

Context and objectives

Water resources are under increasing threat from environmental pollution directly or indirectly caused by human presence and activities worldwide. In turn, the deteriorating state of water resources strongly impacts the same human existence. Because of the complex interrelationships between anthropogenic and natural factors of water pollution problems, finding solutions for these problems requires the integration of knowledge from various disciplines, including physical/chemical, biological, socio-economic and technological aspects. It is this acknowledgement of interrelated natural and socio-economic aspects that leads to the challenge taken up by this research: to bridge the gap between disciplinary environmental sciences and governance through an Integrated Approach. Within the context of this paper this approach is referred to as Integrated Water Pollution Assessment (IWPA).

The focus of this research is on situations of elative data- and resource-poorness, as is often the case in developing countries. Two practical cases, Lake Victoria in East Africa, and the Gulf of Guinea in West Africa, were used as case studies. The objective of these studies was to develop a decision support tool for water pollution control measures. Aspects of these studies are used for illustration in this paper. Detailed results of the case studies are presented by Scheren et al. (1995, 2000a&b and 2002a&b) and Scheren (2003).

Theoretical and methodological framework

The basic features of the applied IWPA framework are: (i) a systems approach; (ii) a problem solving (decision support) oriented assessment process, and (iii) inter-disciplinarity. The various methodological tools used in this study are described in the following sections.

The DPSIR framework (Walmsley, 2002), as presented in Figure 1, was used as the overall framework for assessment of the cause-effect chain of the water pollution problems. The driving forces represent the socio-economic factors, such as population growth and industrial/agricultural activities, and natural factors such as erosion and atmospheric deposition that cause pressure on the environment through for example domestic and industrial effluents and pollutant runoff from lands. These pressures may in turn affect the state of the water system, for instance in terms of water quality and eutrophication, resulting in adverse impacts, such as water-borne diseases, reduced fish and drinking water resources. Such impacts provide the basis for corrective responses/actions taken at the different levels (a-d in Figure 1), for example: (a) land-use planning; (b) wastewater treatment; (c) hydrological modifications, or; (d) improving health care.

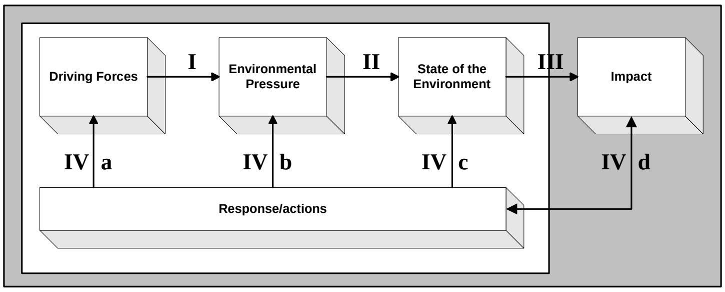

Figure 1. Schematic representation of the DPSIR framework.

The DPSIR framework was used as a basis for development of a decision-support model, involving modelling and assessment of the respective environmental cause-effect chain relations. The methods applied for this analysis concern first of all a pollution load analysis based upon the **Rapid Assessment** method (Economopoulus, 1993) for quantitative assessment of the 'driving forces' – 'pressures' relation (Step I in Figure 1), for which the following general formula was applied:

Equation 1

$$[\text{Waste Load}] = [\text{Functional Variable}] \times [\text{Pollution Intensity}] \times [\text{Penetration Factor}]$$

where the *waste load* represents the actual input of pollutants into the lagoon, the *pollution intensity* represents the characteristic amount of waste produced per unit of a certain *functional variable*, and the *penetration factor* enables incorporation of possible pollution reduction effects (e.g. wastewater treatment). A suitable selection of parameters allows estimation of difficult to measure waste loads on the basis of more easily available data, such as population, industrial production and land-use figures. Table 1 presents functional variables and pollution intensities as applied in this research.

Table 1 Functional variables and pollution intensities for the principle pollution sources.

Pollution source	Functional variable	Pollution intensity
Industries	Annual production [t/y; m^3/y]	Pollutant output per unit product [kg/t; kg/m^3]
Households	Population number [-]	Pollutant output per person [kg/y]
Land runoff	Area of (non) cultivated land [m^2]	Export coefficients of pollutants [kg/m^2y]
Atmospheric Deposition	Surface area [m^2]	Areal pollutant deposition rate [kg/m^2y]

Step II in Figure 1, 'pressure' – 'state' relation, was assessed using a 2-dimensional **substance balance model**. The models used for this purpose are based upon the following general formula:

Equation 2 $$V^i \frac{dC_A^i}{dt} = W_A^i + \sum\left[C_A^{s,r,a,i\pm1} Q_{in}^{s,r,a,i\pm1}\right] - R_A^i - Q_{out}^i C_A^i$$

where *i* represents a given segment of the system, $V^i, W_A^i, Q_{out}^i, C_A^i$ and R_A^i represent respectively volume, load of pollutant A, outflow, concentration of pollutant A and net rate of loss processes in segment $i, C_A^{s,r,a,i\pm1}$ [g/m^3] is the concentration of pollutant A from inflowing water streams, which could be from sea (s), rivers (r), atmospheric deposition (a) and/or segments neighbouring segment *i* (i±1), and $Q_{in}^{s,r,a,i\pm1}$ [m^3/y] is the actual water inflow from the same sources. The parameter R_A^i in Equation 2 represents the net balance of pollutant loss processes in the system. In the case of nutrients this involves the net balance of sedimentation and resuspension processes. For nitrogen, moreover, denitrification may play a key role, particularly in case of long retention times. Finally, in certain cases, fixation of gaseous nitrogen by blue-green algae, such as reported for Lake Victoria, may form an important factor of influx of nitrogen into the system.

It should be noted that attempts were made to use more complicated models for Step II in the analysis. However, due to the limited data base, the increased complexity of the models provided limited additional value, and in fact, introduced system uncertainties that were difficult to quantify.

The assessment of Step III ('state' – 'impact') involves primarily the **analysis of socio-economic impacts** of the state, or quality, of the water body. The proposed framework for such analysis is a function-value assessment, based upon a framework proposed by, among others, Turner et al. (1998), Turner (2000) and de Groot et al. (2002), as schematically presented in Figure 2.

The final step in the assessment (Step IV) concerns an **evaluation of possible response measures**. The technique used for this analysis is based upon an evaluation of the consequences of specific measures, under prior defined scenarios. A framework for the definition of possible response measures is presented in Figure 3.

Inherent to the considerable uncertainties in data, information about the system and its processes and relations, and the translation of such processes and relations into mathematical equations, the results of the case studies entail important uncertainties in (i) the form of assessment tools (structure and internal relations of models) and (ii) variables and parameters. An **uncertainty analysis** based upon internal (model calibration and sensitivity analysis) and external (comparison with data, expert reviews) analysis methods was executed. Details of this uncertainty analysis are presented by Scheren (2003).

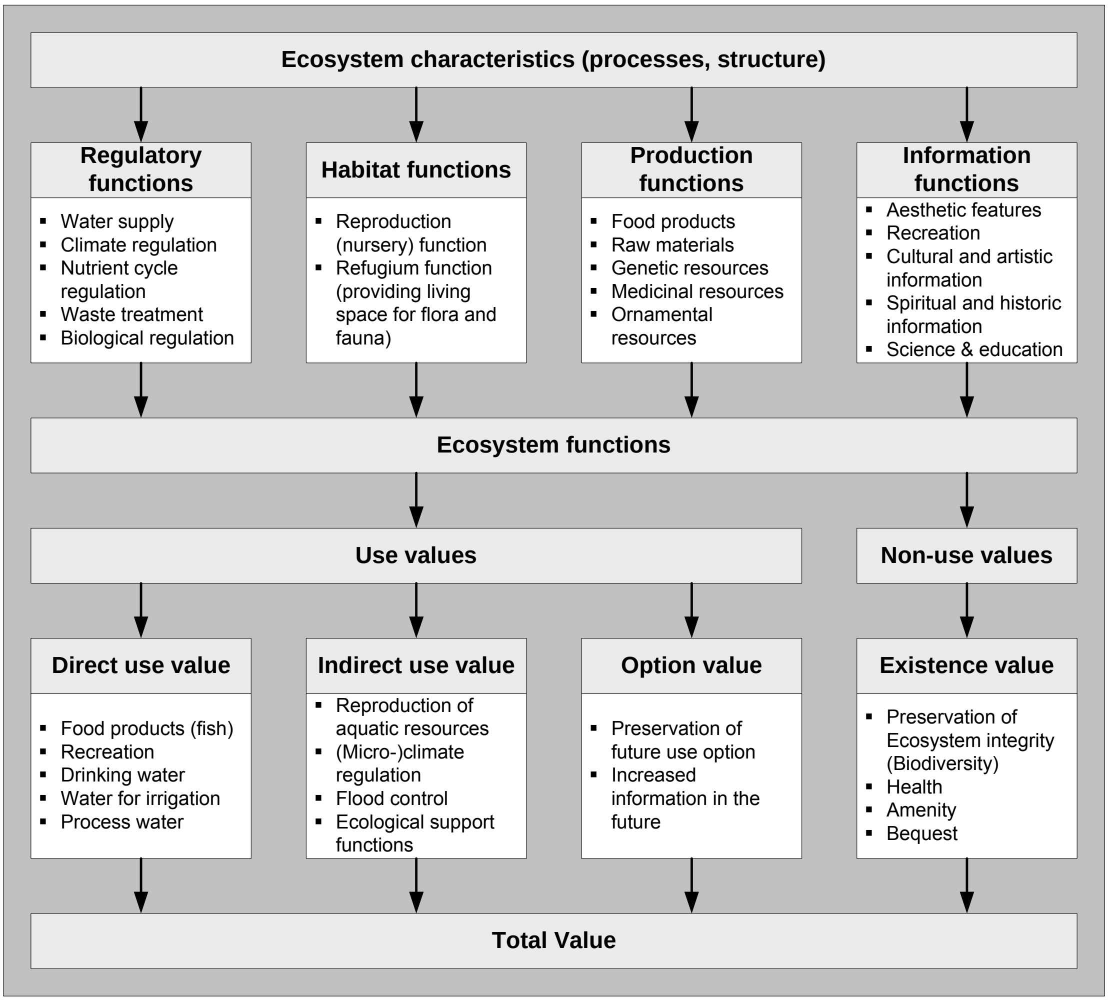

Figure 2: Overall framework for state - impact assessment related to water systems.

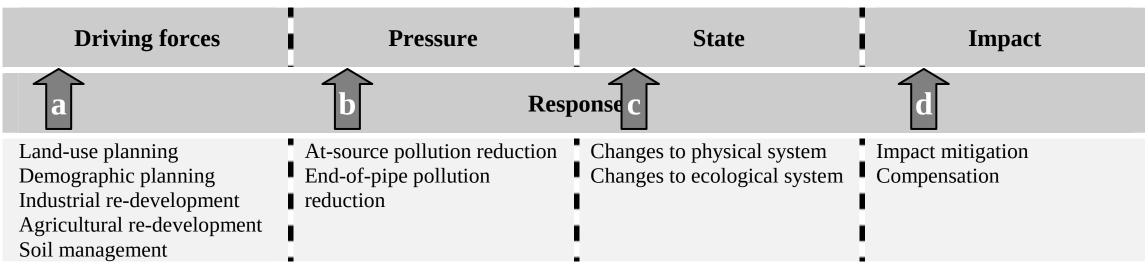

Figure 3: Framework for potential responses to water pollution problems.

Results of the case studies

The previously-described IWPA framework was applied to the cases of Lake Victoria and the Gulf of Guinea. On the basis of the DPSIR framework, the environmental cause-effect chain of each case was assessed, the main features of which are similar for both cases:

- The principal anthropogenic driving forces are rapid population growth, industrial and agricultural development, forest destruction and biomass burning, reinforced by changes in the natural environment changes, such as in meteorological conditions and ecosystem composition (e.g. aquatic vegetation and fish species);
- The main environmental pressures resulting from these driving forces are point-source pollution loads (domestic and industrial), land runoff and leaching of nutrients and atmospheric deposition,

strengthened by pressures resulting from natural driving forces;

- Under such pressures, the state of the water system is affected through changes in the ecosystem, including increased nutrient concentrations, abundant algal growth resulting from eutrophication, deoxygenation, hydrogen sulphide formation and fish mortality;
- The eventual impacts manifest in terms of decreased fish resources, loss of amenities, tourism/recreational value and biodiversity, as well as decreased usability of water for domestic, industrial and agricultural use, and the occurrence of water-borne and insect-borne diseases.

A complete overview of the DPSIR-based cause-effect chain for lake Victoria is presented in Figure 4. Differences between the cause-effect chains of the two research cases relate mainly to the higher level of urbanisation and industrial development in the Ebrié Lagoon case. As a consequence, organic pollution loads from urban sources are relatively more important than in the case of Lake Victoria, where land runoff and atmospheric deposition are the primary sources of nutrients that cause eutrophication of the lake.

On the basis of this cause-effect chain, the key forcing functions through which water pollution may be managed were determined, resulting in various alternative response measures. These options were subsequently compared on the basis of the established quantitative model equations and qualitative interrelationships within the established DPSIR framework. Finally, the results of the IWPA and recommended response measures were considered within the light of uncertainties in data and results. In both cases, the most effective solution for the pollution problem relates to a reduction of non-point source pollution, by means of control measures for demographic developments, sustainable land-use, more efficient agricultural methods, alternative sources of household biomass, and the preservation and rehabilitation of forests and shoreline vegetation. The abatement of urban point-source pollution has, in both cases, only localized effects.

For illustrative purposes, some of the results of the IWPA exercise for Lake Victoria are presented below. Figure 5 presents a scenario analysis for nutrient and BOD loads into the lake, as well as concentrations of nutrients in its waters, for the period 1990-2050, based upon the IPCC B2 scenario (IPCC, 2000) for East Africa. The results of the Lake Victoria case are particularly illustrative in view of the fact that the analysis, which was largely performed over the period 1993-1997, has been superseded by a much more detailed modelling and assessment exercise undertaken as part of the Lake Victoria Environmental management Programme (LVEMP), as reported among others by COWI (2002) and Twong'o *et al.* (2002). Despite the fact that the assessment performed under this study was based upon limited data (both in terms of quality and quantity), its results are largely in agreement with the results of the later detailed study. The differences are well within error margins. The exception in this is the much higher than initially thought phosphorous load from atmospheric deposition. The general conclusions of both levels of assessment are, however, fully in line.

More details of the lake Victoria IWPA were published by Scheren et al. (1995 and 2000a&b). Further details on the general approach, including other case studies, are provided by Scheren (2003).

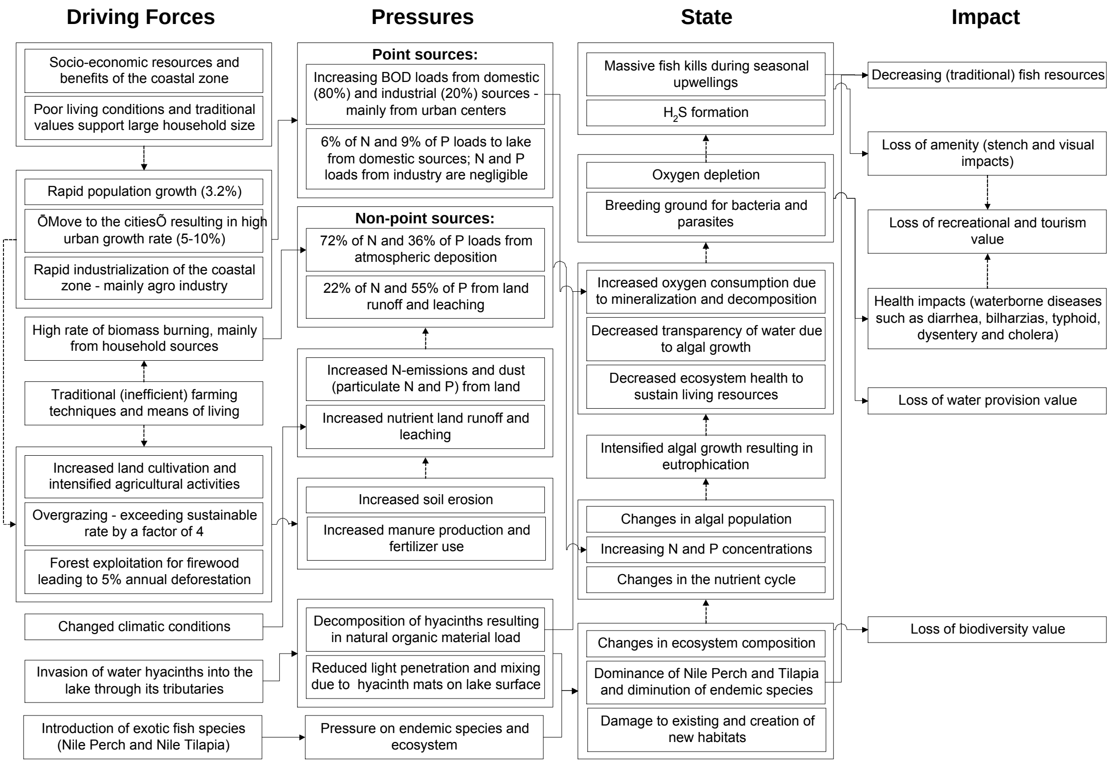

Figure 4.Schematic representation of the Lake Victoria environmental pollution cause-effect chain.

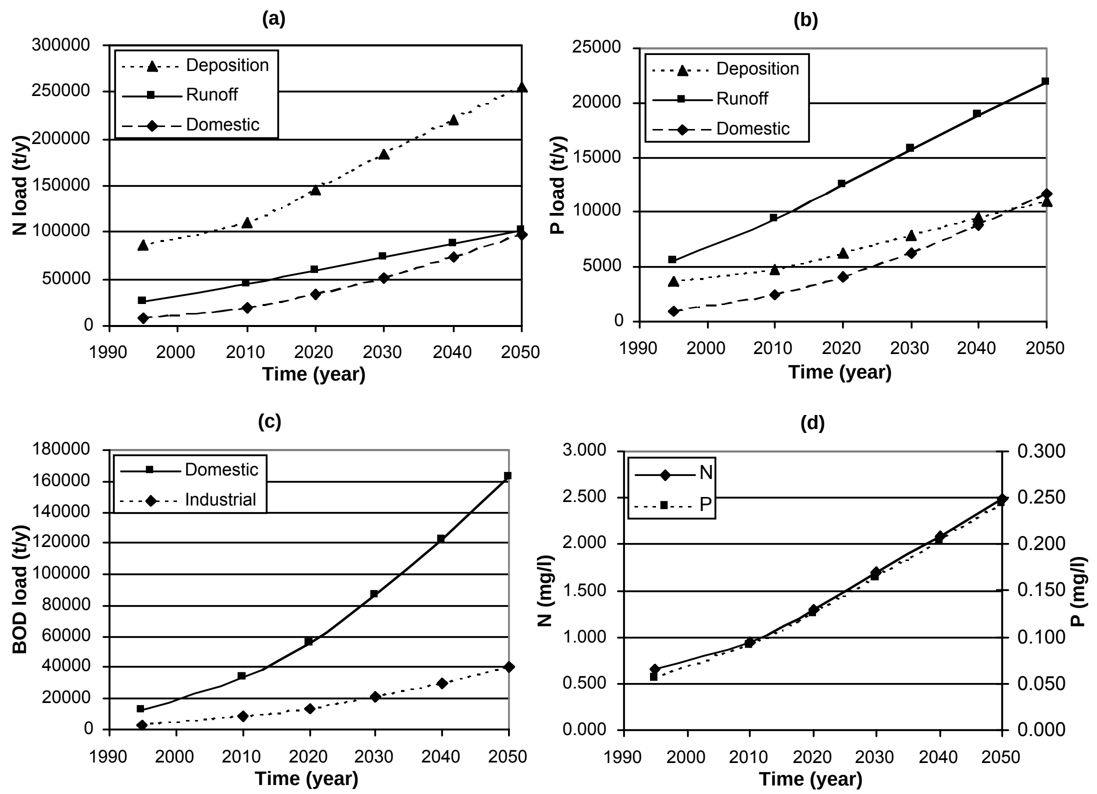

Figure 5. The do-nothing scenario for Lake Victoria for the period 1990 - 2050: (a) N load (t/y) from domestic point sources, land runoff and atmospheric deposition; (b) P load (t/y) from domestic point sources, land runoff and atmospheric deposition; (c) BOD_5 load (t/y) from domestic and industrial sources.

Conclusions

The proposed approach for IWPA has resulted in a clear insight into the causes and effects of the water pollution problems of Lake Victoria and the Ebrié Lagoon. The environmental cause-effect chain, established on the basis of the DPSIR framework, has enabled the identification and evaluation of possible response measures. It may therefore be concluded that the case study level objective defined for this research — to develop a decision support tool for the identification and evaluation of water pollution control measures — has been achieved.

The success of these cases has provided confidence in the general applicability and effectiveness of the proposed IWPA approach for this type of water pollution problems. However, there are also certain weaknesses and restrictions in its applicability, in particular as it concerns (i) uncertainties related to particular elements of the assessment, and (ii) the minimum amount of data required for the assessment, which determines the depth and the spatial resolution of the assessment.

Nevertheless, it is concluded that the established IWPA framework has many merits, which, in its application to the two research cases, has allowed for conclusions to be drawn which would not have been possible through a less systematic and comprehensive assessment. As such, the framework has proven to be useful and robust under the data- and resource poor situations of the two research cases.

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References

COWI Consulting Engineers 2002. *Integrated Water Quality / Limnology Study for Lake Victoria. Lake Victoria Environmental Management Project,* Part II Technical Report.

De Groot, R.S., Wilson, M.A. & Boumans, R. M. J., 2002. A Typology for the Classification, Description and Valuation of Ecosystem Functions, Goods and Services, *Ecological Economics*, 41, 393-408.

Economopoulos, A. P., 1993. *Assessment of Sources of Air, Water and Land Pollution.* Part one: Rapid Inventory Techniques in Environmental Pollution, WHO Geneva.

IPCC - Intergovernmental Panel on Climate Change (2000). *Emission Scenarios*, Cambridge University Press, Cambridge.

Scheren, P.A., Bosboom, J.C., Njau, K.N. & Lemmens, A. M., 1995. Assessment of Water Pollution in the Catchment Area of Lake Victoria, Tanzania, *Journal of Eastern African Research and Development*, 25, 129-143.

Scheren, P.A., Zanting, H.A. & Lemmens, A. M., 2000a. Estimation of Water Pollution in Lake Victoria: Application and Elaboration of the Rapid Assessment Methodology, *Journal of Environmental Management*, 58, 235-248.

Scheren, P.A., Mirambo, V., Lemmens, A.M., Janssen, F.J. & Katima, J. H., 2000b. Assessment of Pollution Sources and Socio-economic Circumstances related to the Eutrophication of Lake Victoria, In: *Proceedings of the Lake Victoria 2000 Symposium in Kampala*, Uganda, May 2000.

Scheren, P.A., Ibe, A.C., Janssen, F.J. & Lemmens, A. M., 2002a. Environmental Pollution in the Gulf of Guinea - A Regional Approach, *Marine Pollution Bulletin*, 44(7), 633-641.

Scheren, P.A., Kroeze, C., Janssen, F.J., Hordijk, L. & Ptasinski, K. J., 2002b. Integrated Water Pollution Assessment of the Ebrié Lagoon, Côte d'Ivoire, West Africa, *Journal of Marine Systems*, 44, 1-17.

Scheren, P.A., 2003. *Integrated Water Pollution Assessment in data- and Resource-poor Situations*. PhD Thesis, Eindhoven University of Technology, Eindhoven (freely available from http://alexandria.tue.nl/extra2/200311844.pdf).

Turner, R.K., Lorenzoni, I., Beaumont, N., Bateman, I.J., Langford, I.H. & McDonnald, A.L., 1998. Coastal Management for Sustainable Development: Analysing Environmental and Socio-economic Changes on the UK Coast, *The Geographical Journal*, 164(3), 269-281.

Turner, R. K., 2000. Integrating Natural and Socio-economic Science in Coastal Management, *Journal of Marine Systems*, 25, 447-460.

Twong'o, T.K., Sikoyo, G.M. & Wakhungu, J. W., 2002. *Shared aquatic ecosystems of East Africa: status and trends*. African Centre for Technology Studies (ACTS), Nairobi.

Walmsley, J. J., 2002. Framework for Measuring Sustainable Development in Catchment Systems, *Environmental Management*, 29(2), 195-206.

Drainage Basin Security of Hazardous Chemical Flux in The Yodo River Basin

Saburo Matsui

Graduate School of Global Environmental Studies, Kyoto University,
Yoshida honmachi, Sakyou-ku, Kyoto City, Japan,606-8501
matsui@eden.env.kyoto-u.ac.jp

Abstract

The Yodo River basin consists of three major tributary basins and other small river basins, namely Uji, Katsura and Kizu, which overlap respectively Shiga, Kyoto and Nara prefecture's administrative areas.

Lake Biwa the largest lake in Japan drains water through the Uji river. The water quality of the lake in terms of BOD, continuously improved for the last decade. However, the quality in terms of COD did not show any improvement in spite of large amount of infrastructure finance being introduced. Eutrophication of the lake still continues, showing nitrogen concentration level being no improvement. Non-point as well as point source control is not strong enough. There is a gap between BOD and COD evaluations of the lake water quality. Hazardous chemical fluxes are estimated based upon PRTR reports of Japan 2001. PCBs are still discharged into the lake, although the report of Shiga Prefecture showed zero discharge. Dace fish monitoring clearly showed that PCBs contamination with the fishes was no changed since '80s in spite ban of use and production of PCBs in '70s. There is still leakage of PCBs into the lake. The major exposure of Dioxins to Japanese is fish diet rather than meat and eggs. The risk of water contamination must take it consideration of not only drinking water safety but also ecological magnification of food chains in water.

Ecological health aspect of hazardous chemicals is also important such as organotins with imposex of sea snails. Finally, public participation of hazardous chemical management shall be very important using the way of risk communication based upon annual report of PRTR in Japan.

Key words: Lake Biwa, The Yodo River, PRTR, Dioxins, PCBs, organotins, hazardous chemicals

Background of Lake Biwa-The Yodo River Drainage Basin

The Yodo River basin consists of three major tributary basins and other small river basins(Figure 1). The Uji River flows from Lake Biwa of Shiga Prefecture, which is the largest lake in Japan. Other major rivers are the Kizu River that flows through Mie and Nara Prefectures, and the Katsura River through Kyoto Prefecture. After the confluence of three major rivers, the Yodo River flows to Osaka Bay connected to Pacific Ocean. Lake Biwa and The Yodo River basin is highly developed in urban, industrial and agricultural activities. Lake Biwa and other rivers hold commercial fishery and sport fishing as Osaka Bay. The population that drinks the lake and river water is estimated to be 14 million, and the lowest intake of the river water provides water supply to Osaka City whose population about 3 million, where a portion of the water has passed through 4 times of human consumption in the cascade use (Figure 2).

Lake Biwa and The Yodo River Watershed

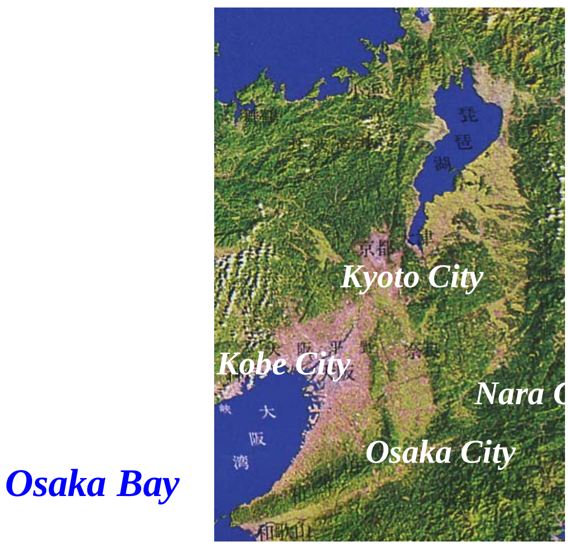

Figure 1. Lake Biwa and the Yodo River watershed

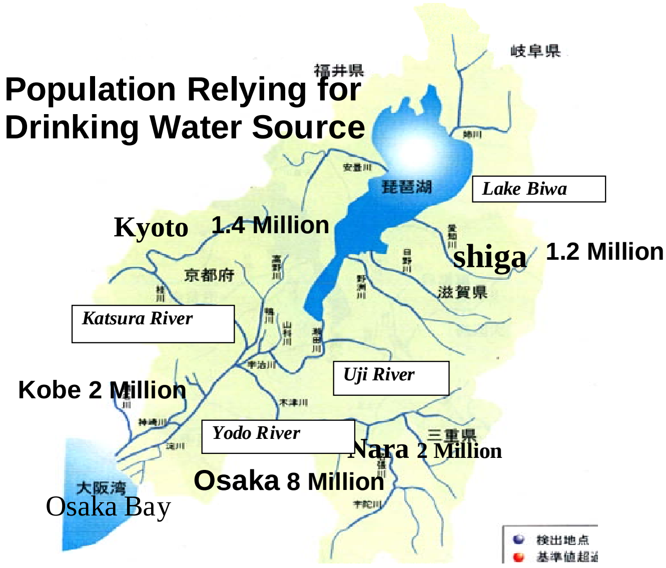

Figure 2. Population relying for drinking water source.

Contamination fluxes of BOD and COD, and their gap

Japanese government regulates river water quality in terms of BOD_5, and lake and coastal water quality in terms of COD_{Mn}. Lake Biwa water quality standard is set for 1.0 mg/L COD_{Mn}, but the current levels are about 2.5 mg/L for Northern Lake Biwa and about 3.5 mg/L for Southern Lake Biwa, exceeding the environmental standard (Figure 3). Although we do not apply BOD standard for lakes, as to the BOD_5 indication, it is shown constantly improved such as about 0.5 mg/L for Northern Lake Biwa in 2000, and about 1.0 mg/L for Southern Lake Biwa in 2000(Figure 4). The improvement indicates successful measures against point source pollution. In spite of the improvement of BOD, the lake has still problems of COD improvement. The pollutant loads of COD can be seen in figure 5, where different types of pollutant loads are depicted. It looks improved water quality in term of BOD, but every summer, the lake develops partly a red tide and water bloom of blue green algae which show still eutrophication problems. Nitrogen and phosphorus control against point and non-point source pollution is not enough yet. The stream standard of the Yodo River at lower reaches is set 3.0mg/L BOD_5, but the current levels show between 1.5 and 1.9 mg/L, which show relatively good quality (Fig 6). However, the end of the river flow reaches to Osaka Bay whose water quality is regulated by COD_{Mn} that show three levels of environmental standards namely, Zone A, 2.0 mg/L, Zone B 3.0 mg/L, and Zone C 8.0 mg/L (Figure 7). The current levels are respectively 2.7 mg/L for Zone A, 3.1 mg/L for Zone B, and 3.5 mg/L for Zone C in 1998 (Figure 8). The Zone A, outer part of Osaka Bay does not meet the environmental standard yet. Here is a gap between water indexes of BOD and COD, the latter includes non-biodegradable chemicals such as humic substances and numerous hazardous chemicals compared to the index of BOD. Japanese government set a target of COD discharge amounts for concerning prefectures with Osaka Bay, which is shown in figure 9. The approach of reducing total COD discharge amounts from prefecture level is an interesting control method.

However, scientists of water quality are facing a big question what are COD compositions, and what are origin of COD substances? The dead cells of plankton including green algae, diatoms, blue green alage, etc can contribute to COD concentration as consequences of eutrophication. Humic substances derived from dead plants of agriculture and forest, and even humic substance like products of activated sludge treating sewage may contribute COD concentrations in river, lake and bay. The gap between BOD and COD seems growing so that investigation of the reason of the gap becomes important.

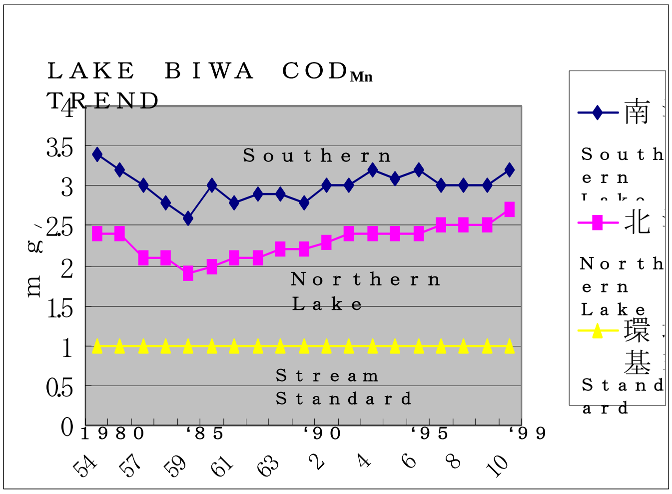

Figure 3. The trend of COD$_{Mn}$ with Lake Biwa against the stream standard.

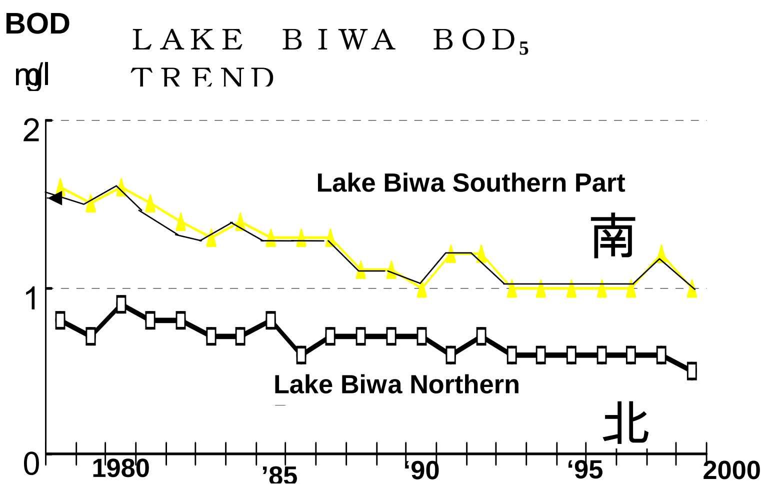

Figure 4. The trend of BOD$_5$ with Lake Biwa.

Contribution of COD from Lake Biwa Drainage in 2000
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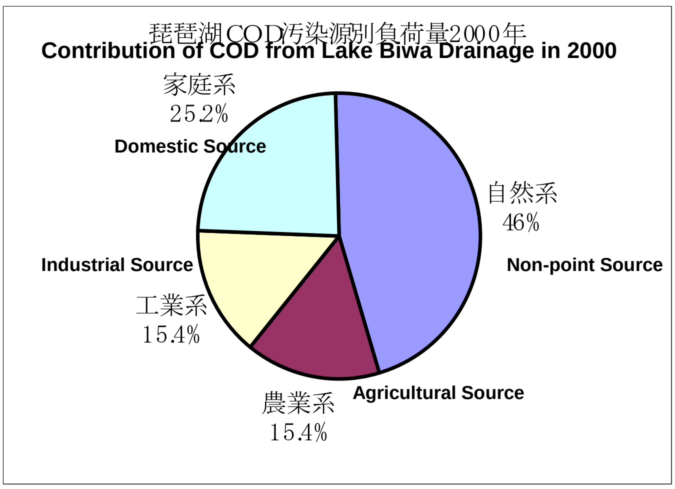

Figure 5. Different pollutant loads of COD to Lake Biwa in 2000.

____5 of The Yodo River at Lower Reaches

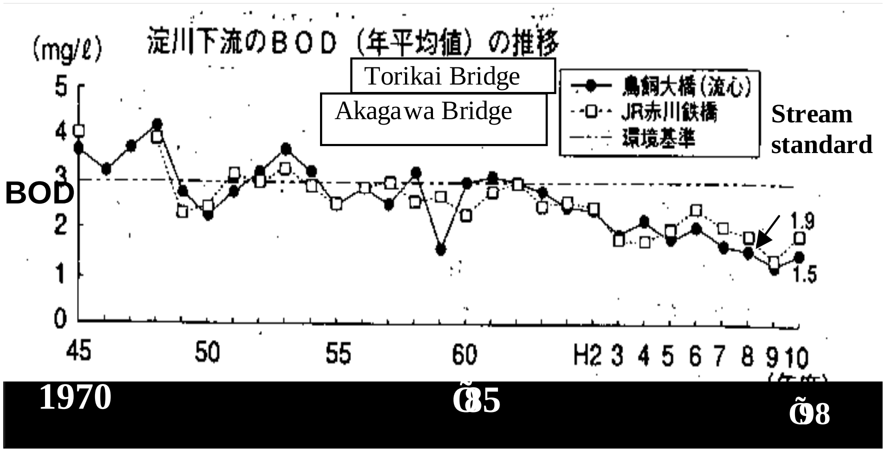

Figure 6. of The Yodo River at Lower Reaches

____ Zone Standard of Osaka Bay

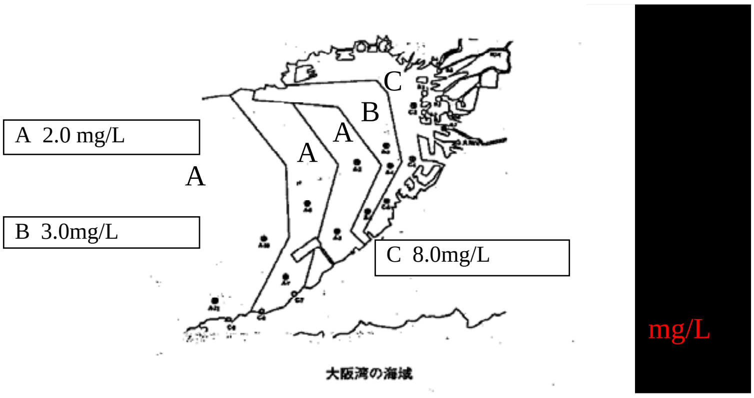

Figure 7 COD Zone standard of Osaka Bay

C O D$_{Mn}$ of Osaka Bay

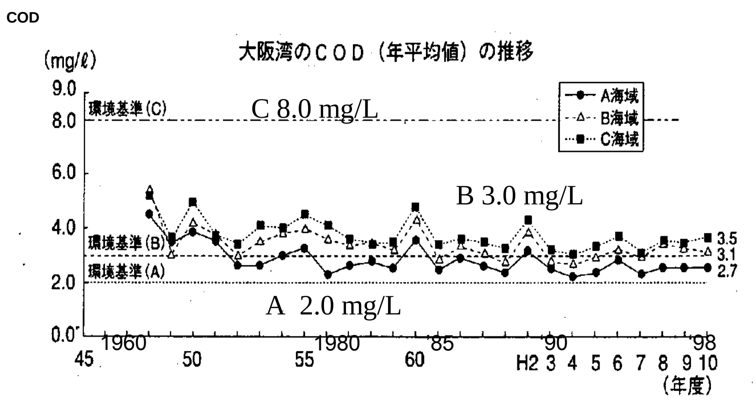

Fig 8 The trend COD$_{Mn}$ of Osaka Bay

Allocation of COD_{Mn} Discharge for Prefectures

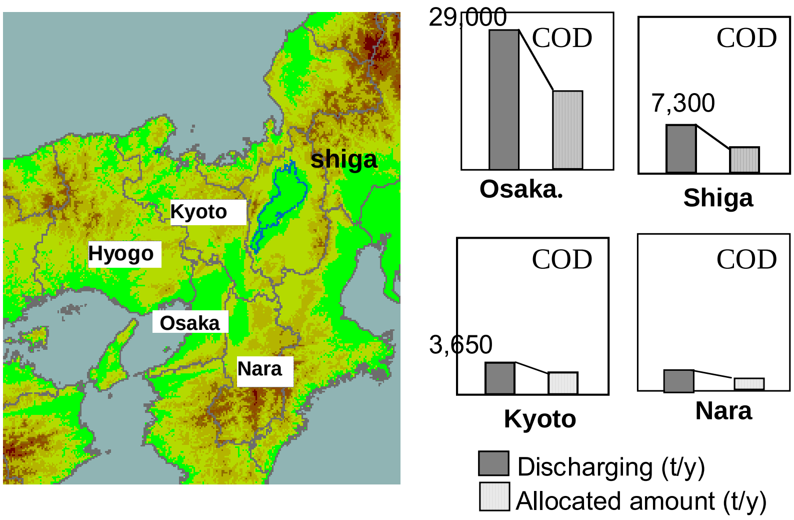

Figure 9 Allocation of COD_{Mn} discharging loads for concerning prefectures with Osaka bay

Hazardous Chemical Fluxes

Due to complex human activities, numerous chemicals flow into the river systems, which form the complex hazardous chemical flux. It is an enormously complicate business to manage those hazardous chemicals in the river systems. Hazardous chemicals are categorized such as industrial chemicals, agricultural chemicals, unintentionally produced chemicals and toxic metals. Japanese government brought the regulation of Pollutant Release and Transfer Register (PRTR) into operation in 2001. Three hundred thirty-five species of hazardous chemicals including Dioxins, PCB, heavy metals, etc. were designated as target substance for control. The designated industries and organizations that were to submit annual inventory reports, were 34,830 in number. The total amount of release of the chemicals reported was about 310,000 tons/year. The total amount of transfer of the chemicals reported was about 220,000 tons/year. In addition to the report, there were estimated reports from prefecture governments on release of the chemicals other than designated industries, which was about 580,000 tons/year. The sum of the release and transfer reached 1,110,000 tons/year. An interesting result of the exercise of PRTR with Shiga Prefecture that holds the watershed of Lake Biwa, is among top ten hazardous chemicals discharged in 2001, nine chemicals were volatile organic chemicals that escape industries in air, and became non-point pollutants after photochemical, physical and biological degradation(Figure 10). Ethylenglycol was only major discharge into wastewater stream, which was understood that wastewater treatment can decompose well. The PRTR report for all Japan was very interesting to understand the situation of Dioxins in 2001.

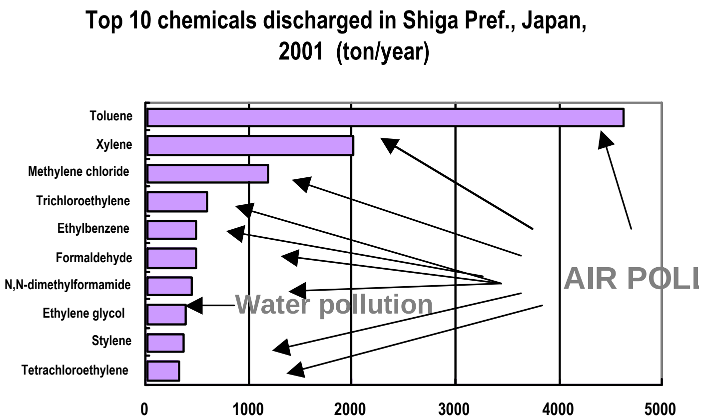

Figure 10 Top ten hazardous chemicals discharged in Shiga Prefecture which holds Lake Biwa.

Figure 11 shows Dioxin discharge amount at the prefecture level in the 5 group of classification in terms of discharging rate. Shiga Prefecture was in the second lowest group discharging 23,712 mg – TEQ/year, while Chiba and Shizuoka Prefectures discharged more than 93,796 mg-TEQ/year. As to the PCBs discharge, there was also five groups of classification in terms of amount. Shiga Prefecture reported zero discharge that was in the lowest group (Figure.12). This report did not well estimate the leakage of PCBs from storing places where PCBs contained in various containers including transformers. Shiga Prefecture Institute of Public Health and Environmental Science has been monitoring concentrations of PCBs and DDT, DDE,etc with daces, freshwater fishes. Figure 13 shows contamination of daces with PCBs and DDT, DDEs in Lake Biwa, which clearly indicate that in spite of ban of PCBs production and use in the early ‘ 70s in Japan, the contamination level of PCBs has not declined well, while DDT, and DDEs show gradual declining which were ban during early ‘ 70s. The life span of daces in Lake Biwa is short, but exposure of PCBs to the fish continues. There must be significant leakage of PCBs continued. Japanese Government started activities of destruction of PCBs, which is not fast enough to stop the leakage.

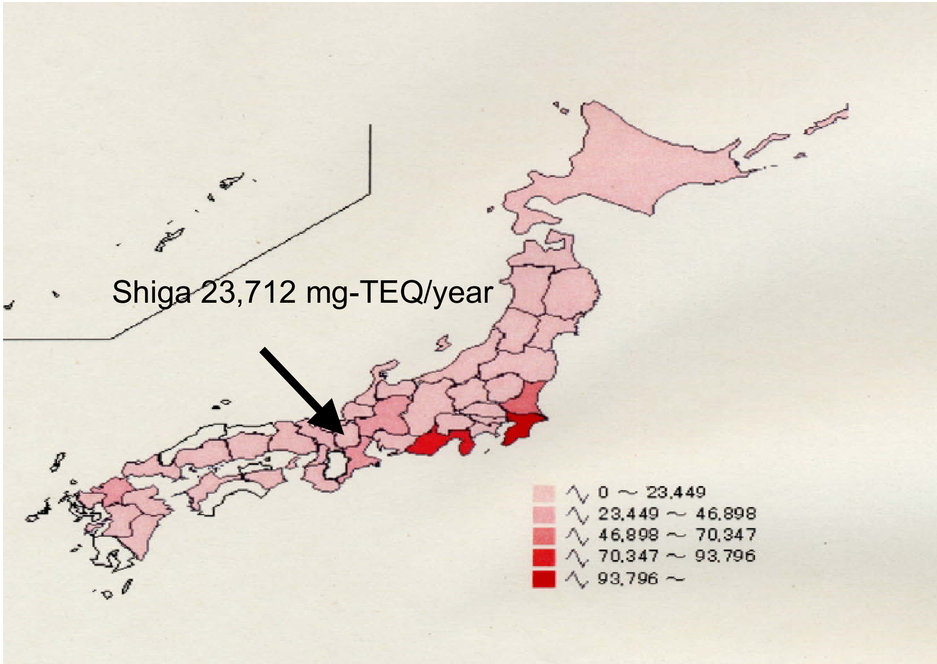

Figure 11. PRTR Prefecture Report on Dioxins 2001 (mg-TEQ/year)

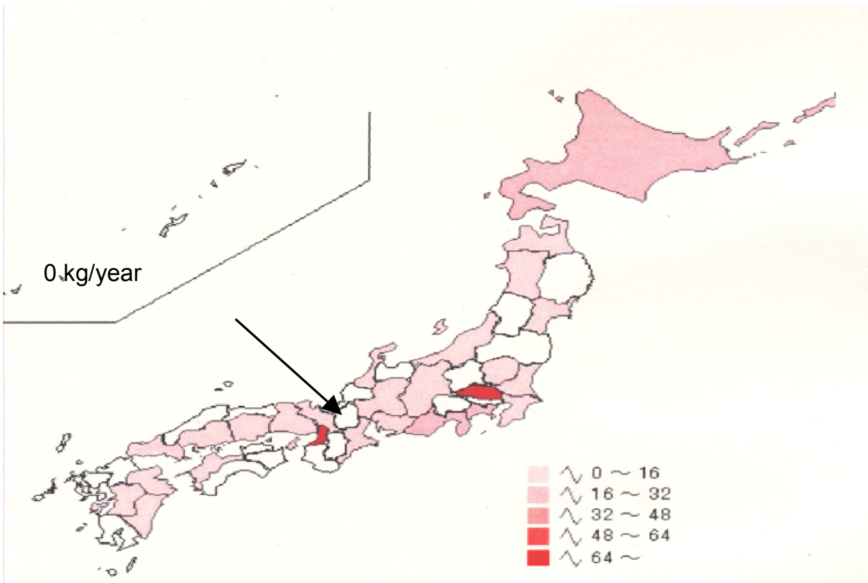

Figure 12. Prefecture Report on (kg/year).

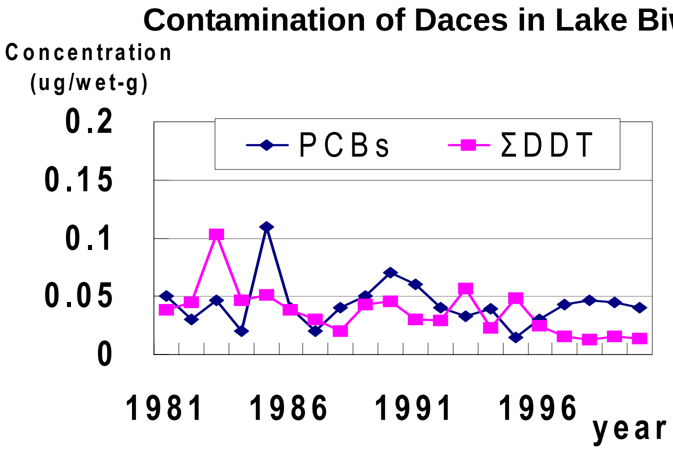

Figure 13 Contamination of Daces with PCBs and DDT, DDEs in Lake Biwa.

Water quality management tends to focus only on the security of drinking water, so that if the monitoring of drinking water quality shows below drinking water standards, the efforts of cleaning river water quality are stopped. The latest development of environmental science is warning that more attention and efforts should be made to look at river water quality in terms of ecosystem health and food chain safety with new views of endocrine disrupting chemicals in addition to carcinogenic chemicals. Japanese fish consumption is much larger compared to other people so that most hazardous chemicals are introduced through more food than water. Figure 14 shows Dioxins intake by Japanese through average diet (pg TEQ /kg body weight /day). The largest portion of Dioxins intake is through fishes, followed by meat and eggs, dairy products, vegetable and grains. Intakes through rice, sugar/sweet, oil, beans, etc are relatively minor. Fish contamination through food chain processes with hazardous chemicals is very much concerned by Japanese.

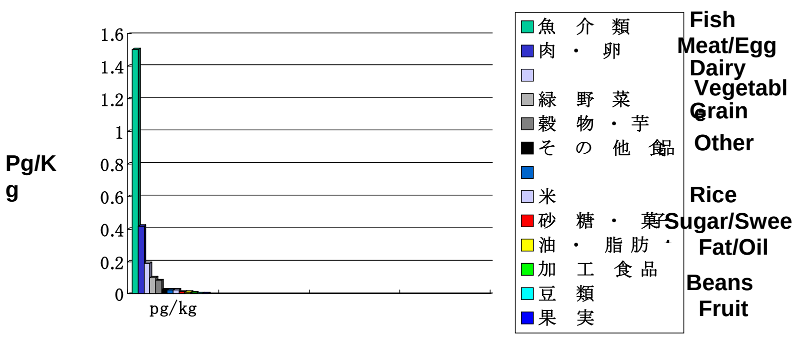

Figure 14 Dioxins intake by Japanese through average diet (pg TEQ /kg body weight /day).

As to ecosystem health, Japan faces serious problems of imposex with sea snails by organotins that were used for anti-fouling paints for boats/ aquaculture nets, which were prohibited in most of developed countries but developing countries (Figure 15). The imposex by oranotins are understood that cytochrom P450 aromatase is prohibited by oragnotins when male hormone is converted into a female hormone, estoron so that female sea snails, gastropods show extended penis and inhibition of breeding, namely reproductive abnormalities (Figure 16). Due to the imposex, the fishery of sea snails in Japan showed strange sex ratio seen in Figure 17, where the sex ratio among male, false-male and female is abnormal. Causative organotins are used in many ways including timber preservation and antifouling (tri-butyl tin), PVC stabilizer and catalysts of plastic industries (di-butyl tin),and catalysts of plastic industries (mono-butyl tin). They are degraded from tri- to di- and mono-butyl tins by microbial activities. Di-and mono-butyl tins are still in use in industries as well as PVC stabilizer.

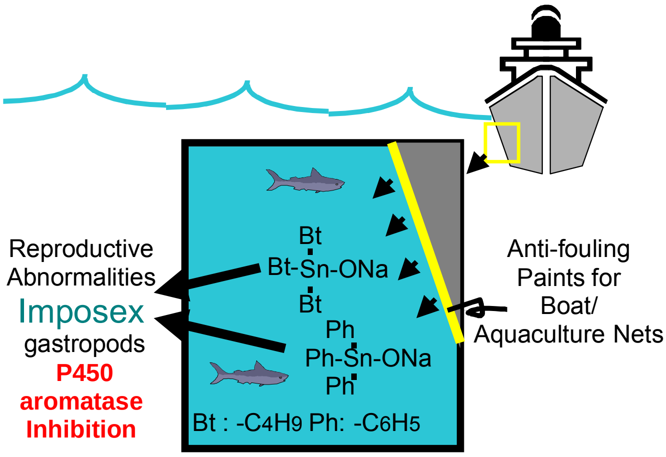

Figure 15 Organotin contamination in the water environment.

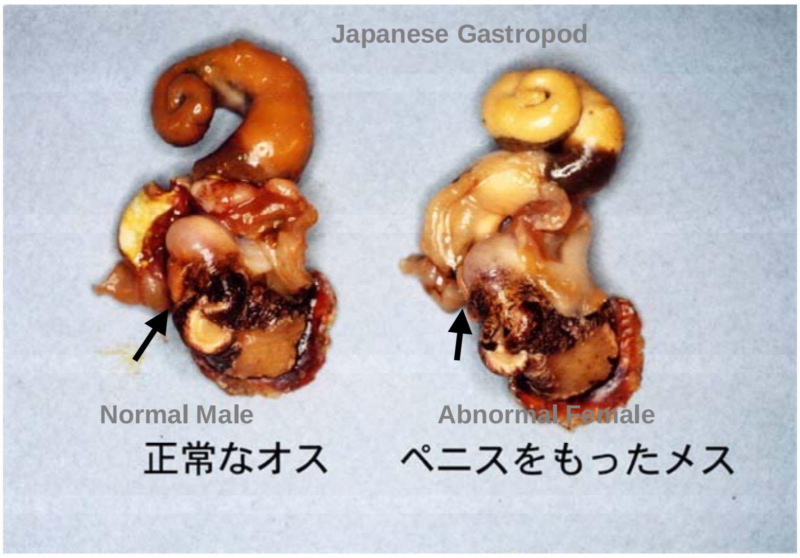

Figure 16. Normal male and its penis in the left, and abnormal female with it penis in the right.

Fishery of A popular Type of Sea Snail at a Market

Figure 17. The sex ratio among male, false-male and female is abnormal at a market in the area of Japan Sea side.

Figure 18 shows hazardous chemical discharge by four prefectures along Lake Biwa and the Yodo River system. Osaka Prefecture that is a highly industrial area, discharge largest amount of chemicals as waste treatment and directly into air as volatile substances. Relatively small amount of chemicals is discharged into sewage partly due to water quality regulation of pre-treatment. It is relatively important to further control hazardous chemicals in the form of direct discharge into air, which end up in the form of non-point pollutants in the watershed. It is also important industrial solid and liquid treatment and disposal. The leakage of hazardous chemicals in water and air from industrial disposal sites is another important point in view of chemical fluxes. The Japanese PRTR exercise has just started from 2001, it can be seen that public participation to hazardous chemical management of industries becomes more important in the way of risk communication. Environmental activities of voluntary citizens against hazardous chemicals shall be a key for sustainable industrial society of 21st century.

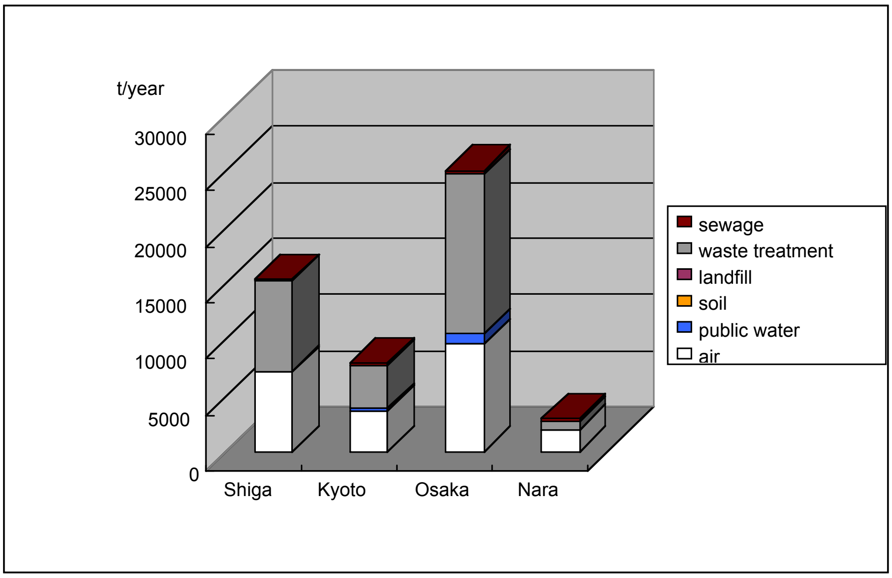

Figure 18. Hazardous chemical flux from four prefectures along Lake Biwa and the Yodo River system.

Conclusion

Japanese Government regulates river water with BOD index and lake and coastal water with COD. There is a gap between them that river water quality has been significantly improved with BOD index, while almost no improvement of lake and coastal water with COD index. This gap issue must be investigated for improving public water quality in the near future. Eutrophication of lakes and coastal water with nitrogen and phosphorus is still an unimproved problem in Japan.

Hazardous chemical management shall be improved by the introduction of the PRTR regulation in Japan. However, risk communication between industry and public is a key to regulate hazardous chemicals in addition to voluntary improvement of industry. Growing production of new species of chemicals is continuing which may cause a new consequence of pollution in global scale. Consumer and environmental activities need focus on more hazardous chemicals.

Acknowledgement

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References

Shiga Prefecture Environmental Report, 2001. (in Japanese)
The PRTR report Japan, 2001. (in Japanese).
Reports of Shiga Prefecture Institute of Public Health and Environmental Science, 21,1986.

Occurrence of aquatic macrophytes in relation to biotic and abiotic factors in a medium sized tropical lake in the Nigerian savanna

Bako, S. P. and M. L. Balarabe

Department of Biological Sciences, Faculty of Science
Ahmadu Bello University, Zaria, Nigeria.
E-mail: spbako2002@yahoo.com or paulbako@witty.com

Abstract

Kangimi Lake is a medium sized man-made lake ecosystem located near Kaduna town in the Nigerian Northern Guinea Savanna. A survey of this tropical eutrophic freshwater ecosystem was conducted to evaluate the pattern of occurrence of aquatic macrophytes in relation to some biotic and abiotic components of the lake, over a period of two seasons (rainy and dry seasons). A 20% decline in effective lake volume during the dry season was accompanied by decline in total dissolved solids, nutrient content, phytoplankton biomass (chlorophyll-a content) and morpho-edaphic index. On the other hand, transparency, electrical conductivity, temperature, pH and dissolved oxygen increased. Eleven species of aquatic macrophytes (mainly emergent forms) were recorded. These declined during the dry season, and this decline was accompanied by an increase in submerged macrophyte populations. The dominant emergent forms were *Echinochloa pyramidalis* and *Oryza longiostaminata*, while the submerged form was *Najas pectinata* (this genus being reported only for the second time in Nigerian freshwaters). Ten species of zooplankton were recorded. Species richness increased and was dominated by rotifers and copepods. Physico-chemical properties of the lake had important effects on the occurrence and distribution of aquatic macrophyte populations, as well as, phytoplankton biomass, and these in turn had important effects on zooplankton populations and their relative abundance.

Key words: Macrophytes, Physico-chemical parameters, Zooplankton.

Introduction

Biological and physicochemical properties of aquatic ecosystems are important factors that interact to determine the effective use and management of aquatic resources. The properties include abiotic water quality parameters, as well as biotic factors, such as phytoplankton, zooplankton, and aquatic macrophyte populations, which all play important roles in the potential and actual yield profiles of the ecosystem (Kemdirim, 2000).

The distribution of emergent and floating macrophyte species is correlated with water electrical conductivity, K^+ ions and total phosphorus content, as well as, increases in canal width (Khedr and El-Demerdesh, 1997). Environmental and biotic interactions involved in macrophyte community structure and dynamics are highly complex, especially with regards to levels of nutrient availability, relative to phytoplankton and periphyton growth (Hough, *et. al*., 1989). Increased nutrient loading shifts the trophic status of shallow lakes from clear water dominated macrophyte to a turbid water state where the main producers are planktonic algae (Kufel and Kufel, 1997). Nutrient induced dominance of phytoplankton and periphyton leads to reduced growth of submerged macrophytes, with eventual dominance of emergent macrophytes and periphytic and eulittoral algae. On the other hand, there is a tendency for allelopathic suppression of phytoplankton by macrophyte organic secretions (Hough *et al*., 1989). There is, generally, a marked seasonality in macrophyte populations, and where these occur in high proportions (especially under poor management), they constitute hindrances to navigation and fishing (Gbem *et al*., 1999).

Water transparency in temperate and sub-tropical lakes is generally high in lakes with a macrophyte coverage exceeding 30%, and this is irrespective of nutrient contents (Jeppensen, *et al.,* 1990). In addition, increased submerged macrophyte coverage often leads to increase in the number of predatory fishes (Grimm and Backx, 1990). Furthermore, phytoplankton biomass could decline as a result of predation by zooplankton (Kemdirim, 2000). Predation pressure, however, may be reduced partly because the submerged macrophytes concomitantly protect the zooplankton against predation by fish fry foraging within the vegetation (Dehl, 1988). Submerged macrophytes can be a direct food source for herbivorous and detritivorous invertebrates (Newmann, 1991; Lodge, 1991) or an indirect food source through the periphyton they support (Bronmark, 1989) or the material they trap (Rooke, 1986). They can also provide macro-invertebrates with surfaces to live on, material for building cases and sites for over-wintering or laying eggs (Soszka, 1975).

There have been very few attempts to evaluate interactions between aquatic macrophyte populations and other biotic and abiotic components of freshwater ecosystems in Africa and particularly in Nigeria. This study therefore is an attempt to evaluate the possibility of these interactions in a eutrophic freshwater man-made ecosystem (Kangimi Lake) in the Nigerian Savanna.

Materials and methods

Study area

Kangimi Lake is a medium sized man-made ecosystem located off the Kaduna – Jos (Trunk A16) highway, and is reached by a dirt track running 12.88km South-East from Katabu at 1.61km, on the high way in Kaduna State, Nigeria; on Longitude $10^o45'$ and Latitude $07^o36'$E (approximately).

The lake was created as a result of an impoundment of the Kangimi River, at approximately 3.22km upstream of its confluence with Kaduna River. It's main purpose was to supply potable water to Kaduna and environs, but it also serves as a source of irrigation water and a fishing ground to the surrounding rural communities. It has a minimum depth of 2.56m and a maximum depth of 15.82m (during the rainy season) and 1.82m (minimum) in the dry season. Its maximum length is about 9.64km. It has a surface area of 567 hectares, and its capable of holding a total volume of 59095 million litres of water. Its catchment area is approximately 226.92km^2 mainly covering the Kangimi River basin. The land around the lake is mainly a peneplain underlain by pre-cambrian rocks. The basement complex consists of granite and gneiss. The top surface of these rocks has been decomposed to give a non-uniform thickness of laterite soil, which is variable in grading, ranging from silty to coarse clay.

Sampling/data collection and analyses

Field visits were conducted over a twelve- month period spanning the rainy and dry seasons. The lake area was partitioned into four transects and four quadrats (4m^2 each) were laid along each transect, thus establishing sixteen sampling stations. Measurements of water depth, temperature, Secchi disc transparency, and pH as well as aquatic macrophyte counts were recorded for each sampling station. Water samples were collected and analyzed for conductivity, dissolved oxygen (Lind, 1979), NO_3-N (APHA, 1980), phosphate-phosphorous (APHA, 1980), chlorophyll a (Lind, 1979). Zooplankton were collected with a70-mesh size plankton net, fixed in 4% formalin prior to sorting and identification according to Ward and Whipple (1959) and Pennak (1978). Aquatic macrophytes were sampled by biomass estimation methods according to Miles *et al.,* (1970); Higgins *et al*., (1994) and identified with reference to field guides such as Miles *et al*., (1970) and Cook *et al.,* (1974), Obot and Ayeni (1987).

Results

Physico-chemical characteristics of the lake are presented in Table 1. There was about 20% decrease in water volume (as implied by cahnges in mean depth) of the lake during the dry season, as compared with the (volume during) the rainy season. Secchi transparency, water temperature, pH, conductivity and dissolved oxygen all increased during the dry season. In contrast, dissolved solids, nutrient content (NO_2^-N and PO_4^-P), morpho-edaphic index (MEI) as well as chlorophyll-a (phytoplankton biomass) contents decreased during the dry season as compared with the rainy season.

Table 1. Physicochemical characteristics of Kangimi Reservoir tn the rainy and dry seasons.

Parameter	**Rainy Season**	**Dry Season**	**Mean**
Mean depth (m)	6.73	5.38	6.06
Secchi Transparency (m)	1.67	2.35	2.01
Temperature (^oC)	24.00	29.50	26.75
pH	7.60	8.20	7.90
Conductivity (ohms)	41.00	69.00	55.00
Total dissolved solids (TDS) mg$^{1-1}$	44.80	26.60	35.70
Dissolved Oxygen (D.O) mgl^{-1}	6.30	7.50	6.90
Nitrate-Nitrogen (NO_3^+N) mgl^{-1}	1.74	1.02	1.38
Phosphate-Phosphorus (PO_4^-P)	32.00	28.00	30.00
ugl^{-1}	18.00	15.00	16.50
Chorophyll-a ugl^{-1}	6.66	4.94	5.80
Morpho-edaphic index (MEI)			

Table 2 presents the aquatic macrophyte profile of the reservoir during the sampling periods. A total of eleven species representing seven families were recorded. More plant species (9) were recorded during the rainy than in the dry season (7). There was a preponderance of emergent and marginal species. The gramineae (Poaceae) generally dominated the vegetation of the lake, both in terms of species richness (4 spp) and relative cover (%). During the rainy season, *Orya longiostaminata* A. Chev. & Roehr., and *Echinochloa pyramidalis* Hitchc. & Chase together made up over 60% of the emergent vegetation of the lake. During the dry season however, *E. pyramidalis* alone made up over 60% of the emergent vegetation in the lake. In both seasons, *Polygonum lanigerum* R. Br. was next in abundance among the emergents, and this was followed by a strong presence of *Phragmites karka* (Retz) Trin. & ex Steud., and *Typha australis* Schum. & Thonn. (during the rainy season), and the submergent, *Najas pectinata* (Parl.) Magnus. and the emergent *T. australis* during the dry season.

Floating species such as *Vossia cuspidata* Griff. and *Nymphaea maculata* Schum. & Thonn. (*N. micrantha* Guill. & Perr.) were of minor occurrence, with the former occurring only during the dry season.

Three species of flood tolerant marginal plants were recorded. *Sesbania bispinosa* (Jacq.) W.F.Wight was the most common during the rainy season. The other two species (*Mimosa pigra* L. and *Neptunia* *oleracea* Laur.) appeared only during the dry season around the draw down zone of the lake.

Table 2:Aquatic macrophyte community of Kangimi Reservoir in the rainy and dry seasons.

Serial No.	Family/Species	Occurrence (%) Rainy season	Dry Season	Growth Habit
1.0	**Gramineae (Poaceae)**			
1.1	*Oryza longiostaminata*	20.84	-	Emergent
1.2	*Echinochloa pyramydalis*	41.69	63.74	,,
1.3	*Phragmites karka*	06.50	-	,,
1.4	*Vossia cuspidata*	01.64	02.96	Floating
	Polygonaceae			
2.0	*Polygonum lanigerum*	15.15	10.20	Emergent
2.1	**Nymphaeaceae**			
3.0	*Nymphaea micrantha*	01.90	-	Floating
3.1	***Najadaceae***			leaved
4.0	*Najas pectinata†*	0.74	9.48	
4.1	**Papilionaceae**			
5.0	*Sesbania bispinosa††*	04.21	-	Submergent
5.1	**Fabaceae**			
6.0	*Mimosa pigra*	-	0.36	Marginal
6.1	*Neptunia oleracea*	-	0.24	
6.2	**Typhaceae**			,,
7.0	*Typha australis*	06.00	07.70	,,
7.1				
				,,
Total No. of Species present		09	07	

Notes : Previous Nigerian records only in Kainji and Jebba Reservoirs. †† Not strictly aquatic species, but tolerant to seasonal flooding.

The zooplankton profile of the lake during the sampling periods is presented in Table 3. A total of 10 species representing 3 sub-families of crustaceans were recorded in the lake. Number of species recorded was higher during the dry season than in the rainy season. The cladocerans were more abundant during the rainy season, than during the dry season. Rotifers and then the copepods followed a similar parttern of abandance. In contrast, the rotifers were more abundant during the dry season, than in the rainy season. The copepods and then the cladocerans followed these. Nauplii larvae, *Daphnia* sp., *Polyphenus* sp. and *Kellicortia* sp. were more abundant only in the dry season, while others such as *Diaphanosoma* sp. was recorded only during the rainy season.

Table 3:Zooplankton Community Of Kangimi Reservoir In The Rainy And Dry Seasons.

Serial No.	Group/Species	Occurrence (%) Rainy Season	Dry Season
1.0	Copepoda	25.3	37.0
1.1	*Diaptomus* sp.	+	+
1.2	*Nauplii* larvae	-	+
2.0	Cladocera	41.2	22.0
2.1	*Ceriodaphnia* sp.	+	+
2.2	*Daphnia* sp.	-	+
2.3	*Macrothrix* sp.	+	+
2.4	*Polyphenus* sp.	-	+
2.5	*Diaphanosoma* sp.	+	-
3.0	Rotifera	33.5	41.0
3.1	*Keratella* sp.	+	+
3.2	*Chromogaster* sp.	+	+
3.3	*Kellicortia* sp.	-	+
3.4	*Brachionus* sp.	+	+
Total No. of species present		07	10

+ present: - represents absent.

Discussions

The decline in effective lake volume in the dry season can attributed to enhanced evaporation, offtake for irrigation and natural seepage, which results in the annual drawdown flacuation commonly observed in tropical lakes. This decline corresponded with declines in other variables such as total dissolved solids, nutrient content, chlorophyll-a and MEI. Total dissolved solids in a water body declines as the turbulence, induced by physical events such as silt-bearing floods and wind action decreases. This is also associated with lowered volume of water. Similar results in the Kainji Lake, Nigeria indicated that suspended particulate matter was higher at the peak of the rainy season (Imevbore and Boszormenyi, 1975). In this case, lower chlorophyll-a content acted in combination with lower water movements to give rise to increased transparency during the dry season. Low chlorophyll-a content, in spite of increased transparency during this period, could be attributed in part to the reduced vegetation in the lake, which meant that the water surface was more exposed to direct effects of intense sunlight, thus resulting in higher water temperatures. High water temperature may alter algal physiology, leading to lowered chlorophyll content. (Barica, 1987). Furthermore, it has been shown that when pH values are above 8.0, the proportion of free CO_2 becomes very low (Beadle, 1981). Only few algal species and some aquatic macrophytes, which can utilize the bicarbonate ion as a source of CO_2 can photosynthesize (Beadle, 1981). Decline in phytoplankton populations could also have resulted from increased populations of zooplankton, which also occurred during this period. A previous report by Kemdirim (2000) has linked diurnal declines in phytoplankton to increased predation by zooplankton. Higher electrical conductivity during the dry season could be due to the increased concentration of ions as the lake volume decreased. Comen *et al.,* (1983) had observed that high balance of evaporation leads to reduction in water volume with an increase in salt concentration. Increase in salt concentration may also have had concomitant effects on relative abundance of aquatic macrophytes, although the relative response of these plants to electrical conductivity varies between species. In a similar study, Khedr and El-Demerdash (1997) found that while the distribution of emergent species such as *Phragmites australis* (Cav.) Trin. & ex Steud. and *Typha* sp. L. were positively correlated with increasing electrical conductivity; floating plants such as *Eichornia crassipes* (Mart) Solms-Laub and emergents, such as *Echinochloa stagnina* Beauv. showed no significant correlation with any of the environmental variables studied in the Nile Delta area. Lower aquatic macrophyte species composition during the dry season could also have resulted from the lowered nutrient content in the water, particularly NO_3^- and PO_4^- observed in this study. Nutrient availability is a crucial factor in aquatic macrophyte, phytoplankton and periphyton growth (Hough *et al.*, 1989). Decreased nutrient content could be attributed to low decomposition from autochthonous and allocthonous sources; lack of nutrient influx from the catchment, or reduced sediment-water interactions with the corresponding physico-chemical variables, such as pH and Cl^- content. Low decomposition rates could have resulted in the observed increase in dissolved oxygen content during the dry season. Decomposition of organic matter is a high O_2 consuming process. In addition, high O_2 content during the rains could be a result of increased rate photosynthesis, as reflected in the higher chlorophyll-a content and increased macrophyte populations during this period. Maitland (1978) reported that the amount of oxygen in a water body depends on the balance between the extent of contact with the atmosphere, as well as the amount produced (through photosynthesis by phytoplankton and aquatic macrophytes) and the amount consumed within the system.

The only submerged macrophyte observed in this reservoir was *N. pectinata* and this finding is of significance because, this species has only previously been recorded in the Kainji and Jebba Lakes in the country (Ita, 1993). The population of this macrophyte was greatly boosted (over 1000 times) in the dry season as compared to the rainy season. This boost occurring at a period during which many of the emergent species recorded a decline in populations, and during which phytoplankton (chlorophyll-a content) biomass also declined. This demonstrates some of the complex interactions described by Hough *et al.* (1989), that nutrient induced dominance of emergent macrophytes, periphytic and eulittoral algae lead to reduced submerged macrophyte growth, probably due to a shading effect. Reduced nutrient content lowers phytoplankton and emergent macrophyte growth thereby resulting in increased submergent macrophyte growth. These interactions have resultant impacts on zooplankton/fish interactions, mainly due to the beneficial effects of submerged macrophytes to zooplankton populations (Lodge, 1991; Newman, 1991; Oertli and Lachavanne, 1995).

In this study, the species richness of zooplankton was found to be higher during the dry season, thus coinciding with the boost in submerged macrophyte populations. Further, there was a relative dominance of the submerged macrophyte population, with the cladoceran, and rotifer populations being higher in the instances of low submerged macrophyte populations, while copepods and rotifers dominated during periods of high submerged macrophyte populations. Similar interactions have been reported by Schriver *et al.* (1995), in a shallow eutrophic lake. Their findings showed that in the absence of submerged macrophytes, zooplankton biomass was low and dominated by cyclopoid copepods, while phytoplankton bio-volume was high. When the population of submerged macrophytes increased, planktonic cladoceran biomass was high and dominated by relatively large-sized specimens, while the phytoplankton biovolume was low and dominated by small fast-growing flagellates.

This study has shown that declines in nutrient content of a medium-sized tropical freshwater ecosystem is accompanied by decline in emergent macrophyte populations and a resultant boost in the population of submerged macrophytes. Improvements in submerged macrophyte abundance may have substantial positive impact on the zooplankton, leading to lower phytoplankton biovolume and higher water transparency.

References

American Public Health Association, 1980. *Standard methods for the examination of water and wastewater*. (APHA) Washington, DC. 15th edition. 1076 pp.

Barica, J., 1987. Water Quality Problems Associated with High Productivity of Praire Lakes in Canada: A Review. *Water Qual. Bull.* 12, (3) pp 107-115.

Beadle, L. C., 1981. *The inland waters of tropical Africa: an introduction to tropical limnology* (2nd ed.). Longman (1981), London, New York.

Bronmark, C., 1989. Interactions between epiphytes, macrophytes and freshwater snails: a review. *Journal of Mollusc Stud.* 55: 299-311.

Comin, F.A., Alonso, M., Lopez, and Comelles, M., 1983. Limnology of Gallocanta Lake, Aragon, Northeastern Spain. *Hydrobiol.* 105: 20-221.

Cook, D.K.C., Gut, B.J., Rix, E.M., Schneller, J. and Seitz, M., 1974. *Water plants of the world: A manual for the identification of the genera of freshwater macrophytes*. Dr. W. Junk. Bv. (publ.). The Hague (1974) p. 33.

Dehl, S., 1988. Foraging efficiency of three freshwater fishes: effects of structural complexity and light. *Oikos* 1988: 209-214.

Gbem, T.T., M.L. Balarabe, S.P. Bako, J. Auta and J.K. Balogun, 1999. *A study of the changes in physical characteristics of a small man-made freshwater ecosystem in relation to aquatic vegetation (a case study of Bomo Lake, Zaria) 1987-1997).* Proceedings of the International Seminar on Performance of Large and Small-scale Irrigation Schemes in Africa. ICID/NINCID Workshop, Abuja, Nigeria, 15th-19th November, 1999, pp. 307-315).

Grimm, M.P. and Backx, 1990. The restoration of shallow euthrophic lakesand the role of northern pike, aquatic vegetation and nutrient concentration. *Hydrobiol.* 200/201: 205-218.

Hough, R.A., Fornwall, M .D. Negele, B.J., Thompson, R.L. and Putt, D.A., 1989. Plant community dynamics in a chain of lakes: principal factors in the decline of rooted macrophytes with eutrophication. *Hydrobiol*. 173: 199-217.

Higgins, K.F., Oldemeyer, J.L., Jenkins, K.J., Clambey, G.K. and Harlow, R.F., 1994. Vegetation sampling and measurement. *In.* T.A. Bookhout, ed. *Research and Management Techniques for Wildlife and Habitats* (5th ed.) pp. 567-591.

The Wildlife Society Bethseda Md.

Imevbore, A.M.A. and Boszormenyi, Z., 1975. Preliminary measurements of photosynthetic productivity in Lake Kainji. *In.* A.M.A. Imevbore and O.S. Adegoke eds. *The Ecology of Lake Kainji: The Transition from River to Lake*. (1975) pp. 136-145. University of Ife Press, Ile-Ife, Nigeria.

Ita, E.O., 1993. *Aquatic plants and wetland wildlife resources of Nigeria*. CIFA Occasional Paper No. 21: FAO, Rome (1993).

Jeppensen, E., Sondergaard, M., Mortensen, E., Kristensen, P., Riemann, B.Jensen, H.J., Muller, J.P., Sortkjaer, O., Jensen, J.P., Christoffersen, K., Bosselmann, S. and Dall, E., 1990. Fish manipulation as lake restoration tool in shallow eutrophic temperate lakes 1: Cross-analysis of three Danish case-studies. *Hydrobiol*. 200/201: 205-218.

Kemdirim, E.C., 2000. Diel rhythm of plankton and physico-chemical parameters in Kangimi Reservoir, Kaduna State, Nigeria. *J. Aquat. Sci.*. 15: 35-39.

Khedr, A.A. and El-Demerdash, M.A., 1997. Distribution of aquatic plants in relation to environmental factors in the Nile Delta. *Aquat. Bot*. 56: 75-86.

Kufel, I . and L. Kufel, 1997. Eutrophication processes in a shallow, macrophyte-dominated lake-nutrient loading to and flow through lake Luknajno (Poland). *Hydrobiol.*. 342/343; 387-394

Lind, O.T., 1979. *A handbook of limnological methods*. C.V. Mosby Co., St.Louis. 199 pp.

Lodge, D.M., 1991. Herbivory on freshwater macrophytes. *Aquat. Bot.*.41:195-224.

Maitland, P.S., 1978. *Biology of freshwaters*. R.M. and Company, Great Britain. 315 pp

Miles, P.M., Miles, H.B. and Graham, V.E., 1970 *Tropical Biological Field Studies*. Hulton Educational Publications Limited, U.K. 136 pp.

Newmann, R.M., 1991. Herbivory and detritivory of freshwater macrophytes by invertebrates: a review. *J. North Amer. Benth.. Soc.*. 10**:** 89-114.

Obot, E.A. and Ayeni, S. O. J., 1987. *A handbook of common aquatic plantsof the Kainji lake Basin, Nigeria*. Kainji Lake Research Institution. 51ppSavlog printing Publications, Ilorin.

Oertli, B. and Lachavanne, D. (1995) The effects of shoot age on colonization of an emergent macrophyte (*Typha latifolia*) by macro-invertebrates. *Freshwater Biol.* 34: 421-431.

Pennak, R.W., 1978. *Limnology and trace elements studies in relation to primary production in two mine lakes*. M.Sc. thesis. University of Jos, Jos. pp. 201.

Rooke, B., 1986. Macroinvertebrates associated with macrophytes and plastic limitations in the Eramosa River, Ontario, Canada. *Archiv fur Hydrobiol.* 106: 307-325.

Schriveer, P. Borgestrand, J., Joppesen, E., and Sondergaard, M., 1995 Impact of submerged macrophytes on fish - zooplankton - phytoplankton interaction: large-scale enclosure experiments in a shallow eutrophic lake. *Freshwater Biol.* 33 : 255 - 270.
Soszka, G. J., 1975. Ecological relations between invertebrates and submerged macrophytes in the lake littoral. *Ekol. Pols.* 23: 393-415.
Ward, H.B. and Whipple, G.C., 1959. *Freshwater biology*. Edmonson, W.J.(ed.) John Wiley and Sons, Inc., New York.

Limnology in Nigeria: problems and prospects

Balarabe M. L.

Department of biological sciences, Ahmadu Bello University, Zaria, Nigeria
Email:mlbalarabe2001@yahoo.com

Abstract

Nigeria is blessed with abundant inland freshwater resources accounting for about 12.4% of its total land mass. These resources are faced with changing climatic cycles, population pressure and increased human activities. Generally with the exception of few large man-made lakes that were studied, there has been a death of information on limnology/water quality status and the economic potentials of these aquatic resources. Basic information is important for effective conservation, management and policy issues related to their rational utilization and development. The paper will attempt to highlight these problems and possible solutions.

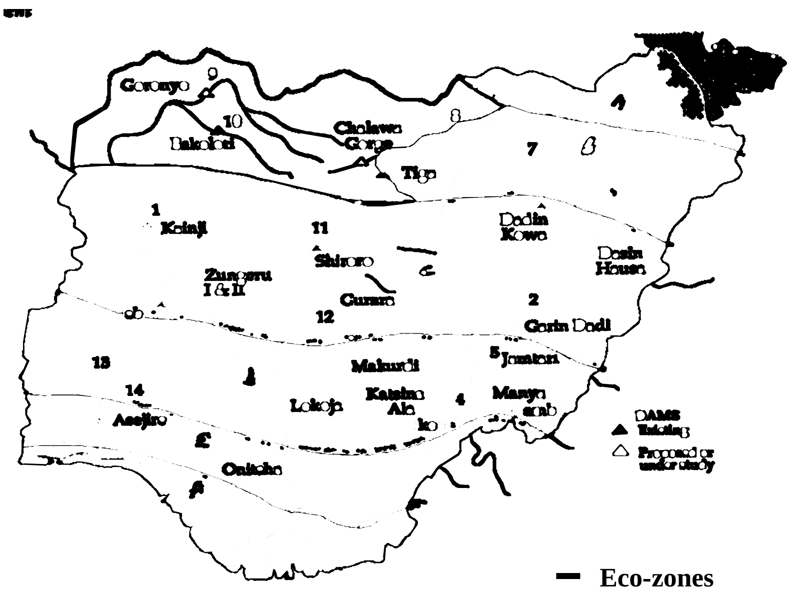
Key word: Limnology

Introduction

Nigeria is situated within the West Coast of Africa, it lies approximately between latitudes 4^0N and 14^0N and longitudes 3^0N and 15^0E. The country has a land mass of approximately 923,00Km^2 encompassing a vast geographical area of contrasting land forms, climatic conditions and vegetation belts. It has a population of over 120 million people. The ever increasing demand of energy necessitated the creation of Kainji lake, the largest man-made lake in Nigeria for Hydro Electric Power generation (H.E.P) which was commissioned in 1968. The aftermath effect of the 1969-1973 droughts led to the creation of several additional man-made lakes especially in Northern Nigeria for the purpose of domestic water supply, irrigation, flood control and fisheries production.

The quality of a given water body is governed by its physical, chemical and biological processes all of which interact with one another and greatly influence its quality. Most early water quality monitoring activities in Nigeria exclusively concerned with the protection of municipal water sources for domestic use. Fisheries productivity of most reservoirs in Nigeria has not been optimized and information on available data is scanty (Ita and Sado, 1985). Therefore both quantitative and qualitative assessment along with background strategy for appropriate potential for use and exploitation of water resources and fisheries development is essential (Bankole and Omorinkoba, 1992). Marshall and Maes (1994) reported that fish production in Africa is the lowest than in any other continent of the world.

Over the last few years, there has been a considerable population growth, with increased urbanization, industrial growth, intensive agricultural land use and climatic changes. These entailed tremendous pressure on the aquatic ecosystems. Conservation and planned utilization of natural resources represents one of the major challenges facing man kind today. Limnology in Nigeria is still developing, it is an essential tool required for proper utilization and adequate management of water resources for sustainable development.



Legend

Some major Rivers in Nigeria.
1.-Niger,
2-Benue
3-Cross River
4-Katsina–Ala
5-Donga
6-Taraba
7-Gongola
8-Hadejia
9-Rima
10-Sokoto
11-Kaduna
12-Gurara
13-Ogun
14-Oshun

Figure 1. Map of Nigeria showing ecological zones and major.(Modified from NEST 1991).

Nigeria's water resources

The water resources of Nigeria are extensive and diverse. They cut across all the major ecological zones (Figure 1) from the tropical rainforest of the Southern coastal zone to the arid Sahel Savanna of the extreme NorthIta *et al*., (1985) identified over 12.5 million hectares of inland water bodies in Nigeria (Table 1).

Table 1: Surface Areas of Lakes, Reservoirs, Ponds and Major Rivers in Nigeria.

Water Body	Surface Area ('000 ha)
Lake Chad (Nigerian sector)	550.0
Kainji Lake	127.0
Anambra River	1,401.0
Benue River	129.0
Cross River	3,900.0
Imo River	910.0
Kwa Iboe river	500.0
Niger river (less Kainji and Jebba Lakes)	169.0
Ogun River	2,237.0
Oshun River	1,565.4
Fish Pond	5.48
Cattle ponds	0.64
Flood Ponds	61.65
Undifferentiated stagnant pools of seasonal rivers	200.0
Reservoirs	275.53
Floodplains	0.002
Burrow-pits	0.002
Mining paddock	0.10
Total:	12,487.82

Source: *Ita et al.*, (1985)

Over 80% of all the major reservoirs are located in Northern Nigeria, a region tht is prone to drought and advancing desertification. Small and medium sized reservoirs are constructed across a large number of seasonal and perennial streams and rivers for the purpose of irrigation (Mbagwu and Ita, 1994), water supply and fisheries development.

Development of limnology in Nigeria

The development of limnology in Nigeria can be grouped into three hiatorical phases : 1960s, 1980s and 1990s

1. **Earlier Works:** Limnological studies in Nigeria have been on the increase since the pioneering work of Onabamiro(1952). Holden and Green (1960) studied the plankton periodicity of Sokoto River. Imevbore (1965 and 1967) provided a preliminary checklist of planktonic organisms of Eleiyele reservoir in Ibadan.

2. **Pre and post Kainji Lake:** Kainji Lake is the largest man-made lake in Nigeria which was formed by damming River Niger. This major river cuts across many countries of West Africa. The purpose of creating the lake was for Hydro-Electric Power (HEP). The Federal Government in collaboration with Food and Agriculture Organization (FAO) of the United Nations established the Kainji Lake Research Project (KLRP) in 1968 to undertake post impoundment research studies on the newly formed man-made lake. Ayeni and Olatunde (1992) stated the long term purpose of the project as to curry out research and surveys on the effects of man in the watershade of Kainji lake and to develop the best management system of the watershade for both present and future generations. In 1975, it was changed to Kainji Lake Research Institute (KLRI) and by 1988, it was changed again to National Institute for Freshwater Fisheries Research (NIFFR). Imevbore and Adegoke (1975) compiled the first integrated comprehensive limnological work in Nigeria which provided baseline information on the transition of River Niger into a lacustrine environment (Kainji Lake). Ayeni and Olatunde (1992) compiled all the research work and activities of Kainji Lake in its two decades of existence.

3. **Resent limnological works:** Some important limnological works carried out in Nigeria include studies on the chemical and phytoplankton composition of Oshun River along with the seasonal variation and distribution of zooplankton of Lake Asejire (Egborge, 1971, 1973 and 1977). Some aspects of freshwater ecology in two man-made lakes in Zaria (Anyam, 1980). Other works on zooplankton studies were reported by Ovie et al, (1992), and Balarabe (2001). Plankton productivity, nutrient dynamics and heavy metal concentrations were also carried out in some water bodies in Plateau State of Nigeria as reported by Wade and Anadu (1987), and Kemdirim (1987).

Problems of limnology in Nigeria

Limnology is not properly integrated in the national planning with respect to water resources development programme. This is so despite the existance of eleven river basin development authorities established since 1976, whose mandates include among others things, the development, management and control of water resources. There is has been a gradual increase of population pressure around water sources coupled with lack of water catchment protection measures. Most reservoirs that were built in the last thirty years are faced with intense human activities resulting in threats of pollution, proliferation of aquatic weeds, siltation, drought and desertification. All these lead to accelerated deterioration in water qualityas well as reduction in size and volume of water. Major rivers like Kaduna and Chalawa that pass through major cities now serve as sewers for domestic wastes and industrial effluents. Lack of experts, non-implementation of environmental monitoring and compliance by the regulatory agencies and problems associated with pollution from all sources are major problems affecting development of limnology in Nigeria (Adeniji,1980). Obsolete equipment, high cost of analysis and poor or absence of reference laboratories for taxonomic identification are some of the current problems facing limnological research in the universities. Azionu (2001) noted that poor funding, lack of equipment and other research materials as well as fragmented short term uncoordinated individual approaches also hinder growth of limnological research in Nigeria.

Prospects of limnological development

The Nigerian Society for Applied Fisheries and Hydrobiology was founded in 1984. The main objective of the society was to pursue scientific research and dissemination of research findings in the area of fisheries, hydrobiology and other related aquatic sciences. The maiden journal of the society was launched in April 1986. The name of the society was later changed to Nigerian Association for Aquatic Sciences (NAAS) and consequently the journal too was also changed to Journal of Aquatic Sciences in 1988.

Nigerian Fisheries and Aquatic Information Centre was established by the information and documentation unit of the National Institute of Freshwater Fisheries Research (NIFFR) in December 1986. It performs the following functions:-

i. Acquire retrospectively and index literature on Nigerian fisheries and other freshwater related disciplines.

ii. Produce Nigerian fisheries and Aquatic Sciences abstract

iii. Compilation of specialized bibliographies

iv. Carryout reference services

In 2003, it became the input center for Nigeria in Aquatic science and fisheries information system (ASFIS) of the FAO which publish the Aquatic sciences and fisheries abstract (ASFA) database. Thereby promoting international awareness for Nigerian journals, titles, articles and their authors on the websites of ASFA .

The future of limnology in Nigeria

In order to promote limnology in developing countries like Nigeria, Tundisi (1980) suggested the establishment of integrated studies in lakes and reservoirs in which long term projects should be used as a model of research and approach which will give emphasis to the development of techniques and methodology. In addition to these there is the need to:-

- Initiate policies aimed at protection and conservation of aquatic resources of the country. The threats of pollution and climatic impacts due to droughts and desertification should be seriously considered along with socioeconomic components in water resources planning, utilization and management.
- Characterize rivers, lakes and reservoirs on the basis of size, ecological location and peculiarities.
- Establish an Institute of Limnology with a database and information centre in the Savanna region of Nigeria where most of the lakes and reservoirs are located.
- Collaborative linkage with foreign research Institutions, International organizations such as International Lake Environment Committee (ILEC) living lakes initiatives and other similar institutions in promoting Limnology.
- Look into the issues of shallow lakes in the field of limnology with respect to their conservation, protection, fisheries development and control of aquatic weeds through community based organisations.
- Integration of socio-economic measures in limnological studies.
- Training of personnel and provision of simple equipment by international organisations for the promotion of limnology in developing countries like Nigeria.

References

Adeniji, H.O.,1980. *Development of Limnology in Nigeria*. Proceedings of the First workshop for the promotion of Limnology in Developing Countries:151-157.

Ayam, R.W.,1980. *Some aspects of the fresh water ecology of two man-made lakes near Zaria, Nigeria*. Msc. Thesis, Ahmadu Bello University, Zaria Nigeria.

Ayeni, J.S. and Olatunde, A.A, 1992. Proceedings of the National Conference on two Decades of research in Lake Kainji 588pp NIFFR New Bussa. Dec. 1989.

Azionu, B.C., 2001. Limnological Research in Nigeria: Some Perspective. Proceeding of the 16th Annual Conference of the Fisheries Society of Nigeria (FISON) Maiduguri, 4th-9th November, 2001 pp 260-267.

Balarabe, M.L., 2001. *Effects of limnological characteristics on Zooplankton composition and distribution in Dumbi and Kwangila ponds Zaria, Nigeria*. Unpublished Ph.D Thesis, Ahmadu Bello University, Zaria 177pp.

Bankole, N.O and Omorinkoba, W. S., 1992. *Nigeria Inland Water Fisheries and Management.* A paper presented at the 7th Annual Conference of the Nigerian Association for Aquatic Sciences, Ilorin-Kwara State, Nigeria.

Egborge, A.B.M, 1971. The Chemical Hydrology of River Oshun, Western State of Nigeria. *Freshwater Biology*. 1:257-271.

Egborge, A.B.M(1973. A Preliminary Checklist of Phytoplankton Oshun River, Nigeria. *Freshwater Bio*., 3:569-572.

Egborge, A.B.M, 1977. *The Seasonal Variation and Distribution of Phytoplankton of lake Asejire*. A new impoundment in Nigeria. Proceedings of International Conference on Lake Kainji. University of Ife: The Ecology of Lake Kainji pp136-140.

Holden, M.J. and Green, J., 1960. The hydrology and Plankton of River Sokoto. *J. Ann.Ecol*., 29:65-84.

Imevbore, A.M.A, 1965. A preliminary checklist of the Planktonic organisms in Eleiyele Reservoir; Ibadan, Nigeria. *J.W. Afri Sci. Ass*., 1056-1060.

Imevbore, A.M.A, 1967. Hydrology and Plankton of Eleiyele, reserviour, Ibadan, Nigeria .Hydrobiologia, 30(1):104-176.

Imevbore, A.M.A and Adegoke, O.S, 1975. *The Ecology of Lake Kainji.* The transition from river to Lake. University of Ife Press. 209 pp.

Ita, E.O., Sado, E.K., Balogun, J.K., Pandogari, A and Ibitoye, B, 1985. *Inventory survey of Nigeria Inland waters and their fishery resources. A preliminary checklist of Inland water bodies in Nigeria with special references to ponds, lakes and major rivers*. Kainji lake Research Institute Technical Report Series. No. 14:151pp.

Marshall, B and Maes, M, 1994. *Small water bodies and their Fisheries in South Africa*. (CIFA Technical paper 29.

Mbagwu, I.G and Ita, E.O, 1994. Pesticides use in the Sub-humid zones of Nigeria: Implications for Conservation of Aquatic resources. *Environmental Conservation*. Vol. 21. No. 3:214-219.

NEST, 1991.. Nigeria's threatened Environment: A National Profile. Pp 60-99 Onabamiro, S.D, 1952. On the diurnal migration and seasonal fluctuation in the numbers of *Thermocyclops nigerianus* Keifer in Nigerian Pond. *Ann.Trop.Med. Parasit*. 46:38-37.

Tundisi, J.G, 1980. *Limnology in Brazil: Present Status and Perspective for Future Research Development*. Proceedings of the First Workshop for the promotion of Limnology in developing countries. pp113-117

Wade, J. W and Anadu, D.I , 1987. Observations on Sediment-water pH conditions and Morpho-edaphic Index of disused Tin mining lakes and their potentials to fisheries production. *Nigeria Journal of Applied Fisheries and Hydrobiology* 2:11-17

Recent ecological changes in of Lake Sare, western Kenya

Gichuki, J., J. Maithya and D. M. Masai

Kenya Marine and Fisheries Research Institute, P.O. Box 1881 Kisumu, Kenya.
Email: gichukij@yahoo.com

Abstract

Studies on the ecology of Lake Sare were carried out to provide baseline information on the ecological conditions before a major wetland reclamation projectwas started upstream of the lake. Results indicated that maximum depth had decreased by 0.9 m while Secchi depth readings had decreased by 0.1 m compared to historical values. This implies that the lake was undergoing siltation. pH values had increased from 6.80 to 7.58 reflecting increased primary production. Conductivity had increased from 106 – 137 2 µS/cm. NO_3- N and PO_4-P had significantly increased from 0.8µg N l^{-1} and 0.14 µg P l^{-1} to 23.90.8µg N l^{-1} and 34.80.8µg P l^{-1} respectively while chlorophyll *a* values reached 34.8 µgl^{-1}. The macrophyte environment was dominated by *Cyperus papyrus, Eichhornia crassipes, Phragmites australis, Typha domingensis,* and *Vossia cuspidata.* The invasion of the lake by *Eichhornia crassipes* is reported here for the first time. The study further observed macrophyte succession where the floating plants *Eichhornia crassipes* and *Pistia stratiotes* were gradually giving way to *Vossia cuspidata* in the Sare lagoon. The phytoplankton community was dominated by *Pediastrum* sp., *Pseudoanabaena, Synedra* sp. and *Cyclindospermopsis* sp. *Cyclindospermopsis* sp. has the potential to secrete a toxin Cylindropermopsin which can affects the liver and to some extent the kidney. Zooplankton communities encountered were from the Copepoda, Cladocera and Rotifera genera. The fish community was dominated by *Lates niloticus* and *Haplochromine* sp which coexist in this ecosystem.

Lake Sare is a high priority ecological site for conservation and management of the resources of Lake Victoria basin. It has direct link with Lake Victoria. Fish populations stocked in the Lake Sare are likely to find their way to Lake Victoria. In view of this the lake can be used as a launch site for restocking Lake Victoria with juveniles of endangered fish species.

Key words: Eutrophication, species diversity, endangered species, restocking

Introduction

Lake Victoria is endowed with various habitats in the form of littoral areas, deep water areas, satellite lakes, dams, rivers and swamps. Each of the habitats is characterized by different ecological conditions that distinguish and thus harboring specially adapted organisms. The main satellite lakes within the Lake Victoria basin are; Lake Simbi, Lake Kanyaboli, Lake Namboyo and Lake Sare.

Lake Victoria provides an important livelihood for over 30 million people living in the basin in Kenya , Uganda, Tanzania, Rwanda and Burundi. Fish production data indicates that the fish yields from Lake Victoria have declined. This is decline can be associated with fisheries over-exploitation using small mesh size gears, pollution and predation by introduced predatory species (Bugenyi and Balirwa 1989; Hecky and Bugenyi 1992; Ogutu-ohwayo 1990 a, b, 1992; Lowe-McConnel *et al*., 1992; Lipiatou *et al*. 1996; Ogutu-Ohwayo *et al*. 1996; Hecky and Bugenyi 1992; Hecky 1993; Lipiatou *et al*. 1996; Lehman *et al*. 1998). The decline in fish yields has prompted researchers to focus on alternative solutions to improve fish yields in order to cater for the ever increasing demand due to human population growth and increasing number of fish industries which process fish for export. In this respect, research has shifted to the satellite lakes and dams within the Lake Victoria basin which are believed to have a unique fish fauna which can be compared to the species composition of the Lake Victoria in 1960s and thus have served as refugia for the endangered fish species of L. Victoria.

Lake Sare lies within the greater Yala swamp complex and together with Lake Kanyaboli and the surrounding Yala swamp wetland in western Kenya have been recognized as important biodiversity hotspots (Maithya, 1988). Recent population genetic and phylogenetic studies confirm the evolutionary importance of Lake Kanyaboli and Lake Sare in preserving the cichlid fish fauna of Lake Victoria (Abila *et al.* 2004). Lake Sare has a unique advantage of having a direct link with Lake Victoria. The adjoining Yala swamp harbours the endangered swamp antelope species Sitatunga (*Tragelaphus spekeii*) and several papyrus endemic birds. The lake and adjoining swamp play a critical role in the livelihood of the local communities, who heavily depend on the adjoining wetland resources.

The main objective of this study was to characterize species diversity of both flora and fauna and to provide information on the ecological conditions before a major wetland reclamation project in the upstream of this lacustrine environment was implimented.

Materials and methods

Study area

Lake Sare is located 0º 02' 25''S; 034º 03' 42''E and lies at an altitude of 1140 m.a.s.l within the greater Yala swamp complex (Figure 1). The lake has a surface area of 5 km^2 and forms part of the outlet of Yala river into L. Victoria. It is surrounded by papyrus swamps which merge with the Yala swamp.

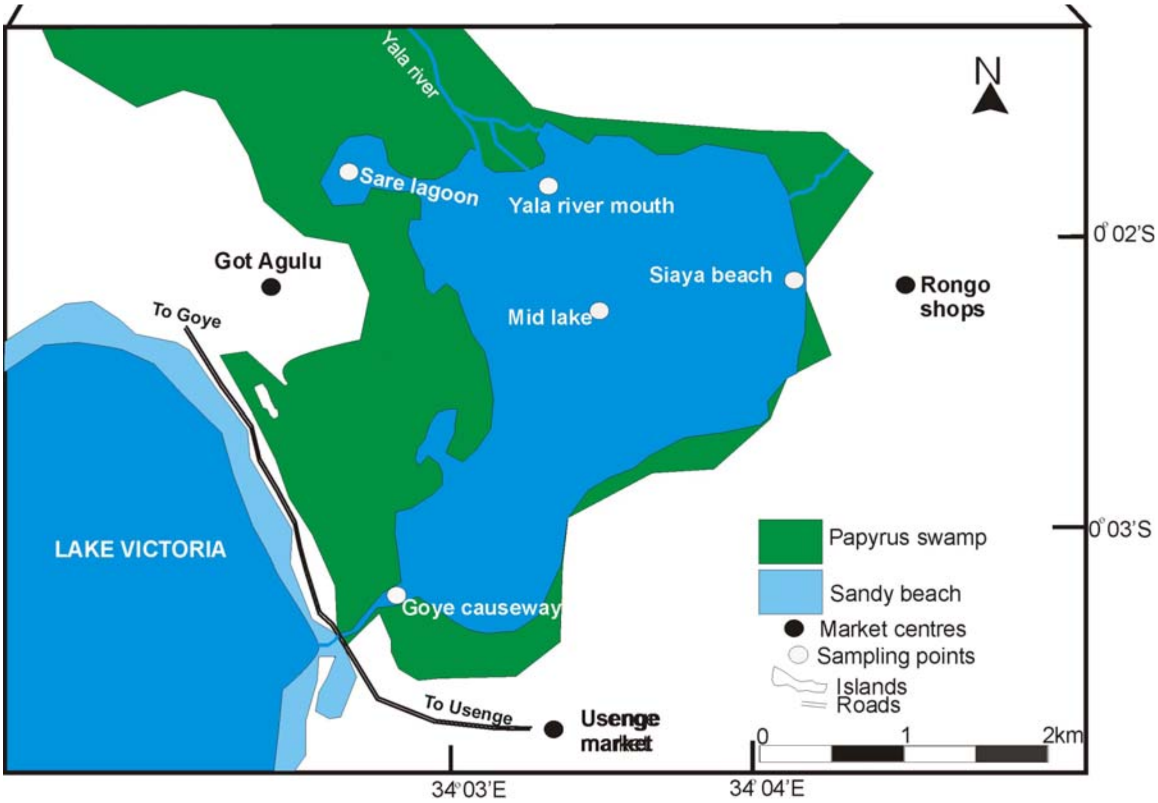


Figure 1. A map of Lake Sare showing its locationin relation to the greater Yala swamp complex in Weatern Kenya.

In total, 4 sampling design were sampled for water physico-chemical characteristics, plant and animal species (Figure 1). The sites were, Yala river mouth (00° 01' 54''S, 034° 03' 03''E), Goye causeway (00° 03' 40''S, 034° 02' 37''E), Midlake (00° 02' 25''S, 034° 03' 42''E) and Sare lagoon (00° 02' 50''S, 034° 02' 57''E). *Insitu* measurements were made for temperature, dissolved oxygen, conductivity, pH automatically using a Hydrolab Surveyor II connected to an SVR 2- DU display and a 401 - CA circulator assembly. Light penetration was estimated with a 20-cm diameter, black and white Secchi disk. Water samples were collected with a 3-litre Van Dorn sampler. A portion of the water samples (50 ml) wwasanalyzed for total alkalinity and total hardness by titration with 0.02 N HCl to final pH of 4.5 using methyl indicator and with 0.02 N EDTA respectively as outlined in GEMS (1992). Spectrophotometric methods were employed to determine the soluble reactive phosphorous (PO_4-P) and nitrate nitrogen (NO_3-N) as outlined by Mackereth *et al* (1978) and soluble reactive silica (SiO_2-Si) according to APHA (1985). For phytoplankton analysis, 250 ml of the water was placed in polyethylene bottle and fixed immediately with Lugol's iodine solution. After 48 hours decantation, the lower layer (20-25 ml) containing the sedimented algae was put in a glass vial and stored in a dark cool box. The known volume of the concentrated sample was used for the identification and counting of the phytoplankton under an inverted microscope. Phytoplankton species were identified using the methods of Huber –Pestalozzi (1968) as well as some publications on East African lakes (see the references in the checklist of Cocquyt *et al.* 1993). Phytoplankton densities (individual's l^{-1}) were estimated by counting all the individuals whether these organisms were single cells, colonies or filaments. Zooplankton samples were collected using a Nansen type plankton net of 60 µm mesh size and 30 cm mouth opening diameter. Three hauls were taken from each sampling site. The net was lowered as close to the bottom as possible without disturbing the sediment. Samples were preserved in 5% formaldehyde solution. In the laboratory, samples were diluted to a known volume and sub samples of known volume taken and placed in a 6x6x1cm counting chamber. Estimates of abundance of zooplankton were made from counts of sub samples under a Leica dissection microscope (x25). Copepods were grouped into immature Copepoda (nauplii and early copepodite stages). *Cyclopoida* and *Calanoida*. Cladocerans were identified to species level using identification keys of Smirnov (1996) and Korovchinsky (1992). With regard to macrophytes, plant parts with diagnostic features such as flowers, fruits, shoots and rhizomes were collected and correctly pressed and labeled to reveal their site of collection, a brief habitat description and associated taxa. Identification was carried out by use of keys of Cook *et al.* (1974), Kokwaro & Johns (1998) and Sainty & Jacobs (1994). Fish samples were collected using a gang of monofilament gill nets with ten different mesh sizes

(30 mm, 38 mm, 48 mm, 60 mm, 77 mm, 97 mm, 124 mm, 157 mm, 199 mm and 255 mm). The fish ata obtained was used to compute size distribution.

Results

Physical and chemical parameters

The results of the physical and chemical parameters are displayed in Table 1.

Table 1. Comparison of the physical chemical variables with other authors for Lake Sare ecosystem

Parameter	Okemwa (1981)	This study
Maximum depth (m)	5.00	4.10
Temperature (ºC)	-	27.02
Surface DO (mg/l)	-	7.48
pH	6.80	7.58
Secchi depth (m)	1.00	0.91
Conductivity (µS/cm)	106.00	137.20
$N0_3$- N (µg N l-1)	0.80	44.80
PO_4-P (µg P l-1)	0.14	23.90
Chlorophyll *a* (µg/l)	-	34.80

Results revealed significant changes in environmental parameters over a period of 24 years (Okemwa, 1981). Maximum depth had decreased by 0.9 m. Water had decreased from 1 to 0.9 m. pH had increased from 6.80 to 7.58 which was associated with increased primary production. Conductivity had increased from 106 – 137 2 µS/cm. NO_3- N and PO_4-P had increased from 0.8µg N l^{-1} and 0.14 µg P l^{-1} to 23.90.8µg N l^{-1} and 34.80.8µg P l^{-1} respectively. Chlorophyll *a* values were in the range of 34.8 µgl^{-1} which suggests that the lake currently exhibits the traditional symptoms of eutrophication.

The fish and fishery of Lake Sare

Fish species identified in Lake Sare are shown in Figure 2. The fish species were dominated by *Lates niloticus* and *haplochromine* spp. These species occupied almost every niche in the lake and constituted the highest propotion in terms of numbers and ichthyomass.

The third most important species in both composition and biomass contribution is *O. leucostictus*. Studies on the diel feeding habits of the major species of the lake are displayed in Figures 3. *L. niloticus* was found to feed throughout the 24-hour period contrary to the belief that this fish feeds at night and not during daytime. The fish fed more at night and in the early hours of the morning. This feeding pattern is also the same for *Haplochromis* species, which form the main food item of *L. niloticus.* Very few specimens of *O.niloticus* were caught and their feeding profile may not convincingly represent that of the entire population in the lake. *O. leucostictus* has peak food ingestion periods at night and daytime.

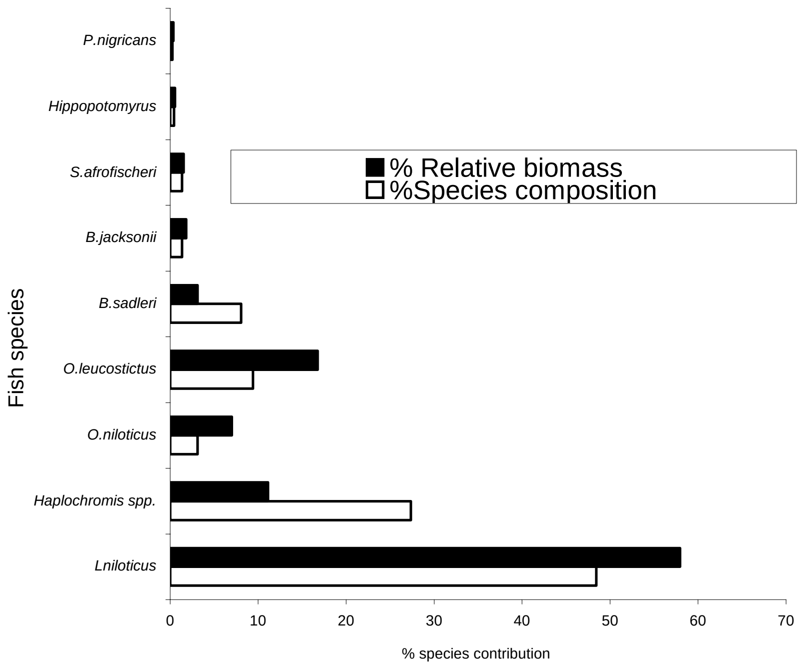


Figure 2. Fish species composition identified in Lake Sare.

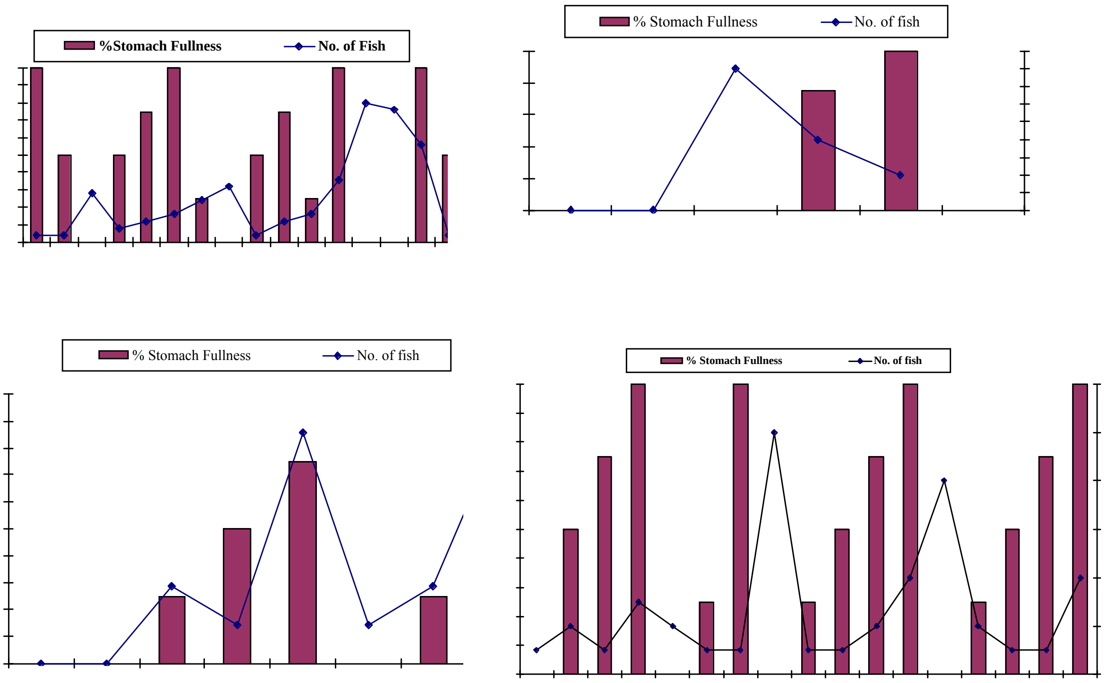


Figure 3. Diel feeding patterns of the major fish species identified in Lake Sare.

Food consumption is rather a factor of its availability in the water. This situation is also applicable for *Brycinus sadleri.* Lake Sare has an elevated primary production as demonstrated by high levels of *chlorophyll a.* Phytoplanktivorous fish have a high concentration of algae to feed on but the algae present (ie the cyanophytes) are unpalatable due to their filamentous and colony forming properties. This could explain the continuous feeding strategy employed by most species, as they may continuously be in search of the more palatable food such as diatoms. The spatial distribution of the fish species the major lake niches is shown in Table 2. Results show that *Lates niloticus* has a lake-wide distribution, occurring in virtually every habitat. Boat surveys during this study revealed smaller size classes of the species that occupy only the entrance to the lagoon and not in the lagoon itself where *O. leucostictus* dominate over other species. Expanded *Lates niloticus* population concentrate along Yala River mouth, the Goye channel and midlake (Table 2). The lagoon area was colonized by *O. leucostictus.* Other species were sporadically distributed in the entire lake ecosystem. The Goye causeway however harbourd the highest number of fish in the lake and has the highest fish biomass representing over 42% of the fish biomass in the lake (Table 2, Figure 4). Sare lagoon is the second most important ecological zone with regard to fish species richness The main fish species identified were mainly *Haplochromines*. *Barbus* species were concentrated at the Yala River mouth while *O.leucostictus* was only present in the lagoon area. Spatial distribution and niche colonization are ecological events that result from availability of food and optimal environmental conditions. But apart from the lagoon, the whole lake area appears to be homogeneous (Table 2, Figure 4). This is probably due to the influence of river Yala.

Table 2. Fish species richness in the main niches of Lake Sare.

Niche	Species richness	% Iterative niche biomass contribution
River Yala mouth	6	10.58
Goye Causeway	12	42.77
Midlake	5	17.41
Lagoon Sare	9	29.2

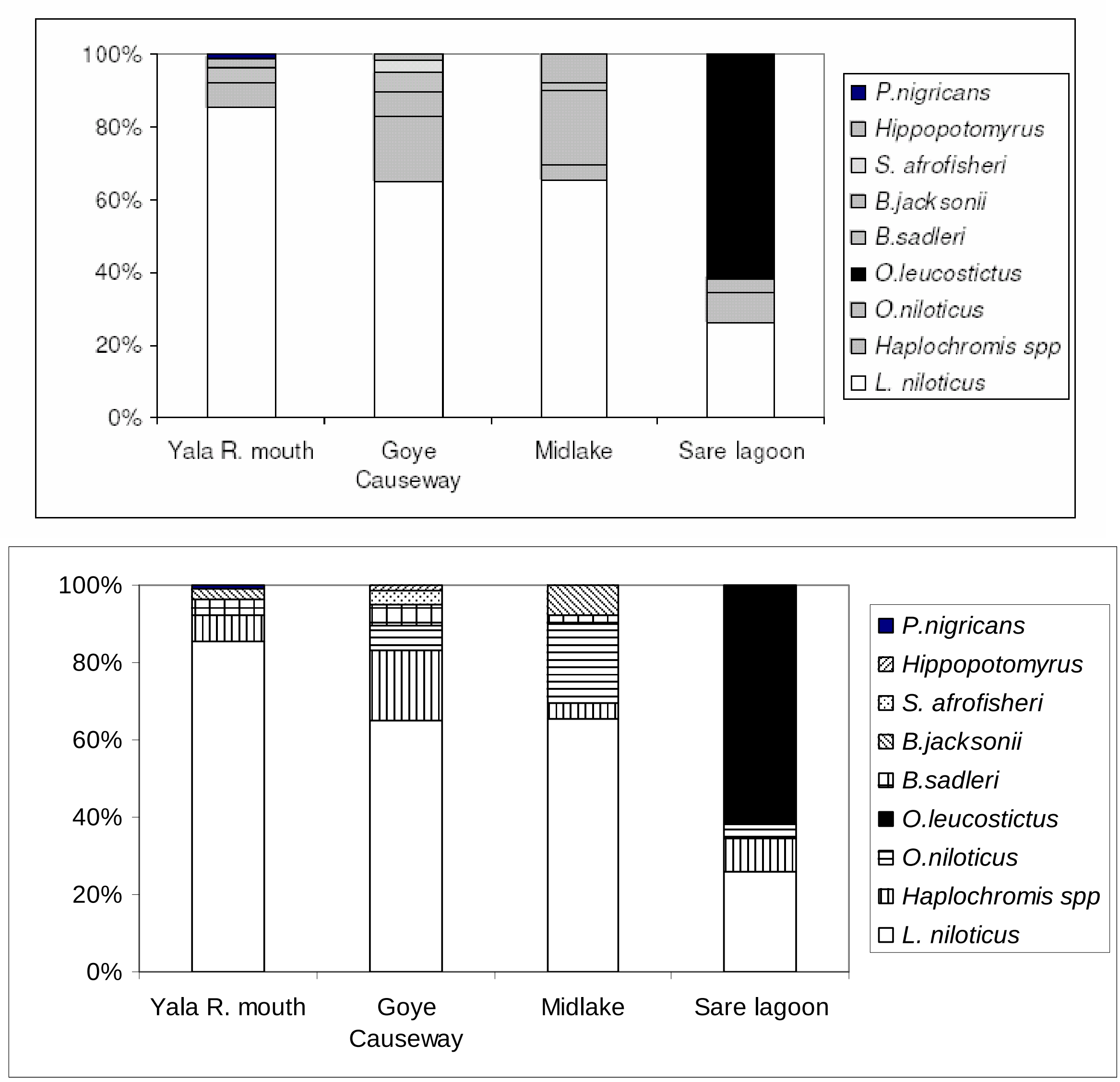


Figure 4. Relative abundance of the fish species in the major niches of Lake Sare.

Size distribution of Nile perch *L. niloticus* of Lake Sare is shown in Figure 5. Despite its wide distribution in the lake *L.niloticus* rarely reaches maturity in the waterbody. The biggest individual sampled was 29.2cm TL and weighed 224g at maturity stage 3. Mature specimens were not caught in all the niches sampled. Due to the preponderance of the young fishes, Lake Sare can be described as a nursery for the major commercial fish species of Lake Victoria. We hypothesize that the young fish leave the main lake through the Goye causeway and enter Lake Sare where they are able to find nutrition and protection from currents and predators. After attainment of adulthood the fish move back into the deep waters of Lake Victoria.

We observed no active commercial fishery in this lake. The fisheries resources are exploited solely by artisanal fishermen in an entirely open-access management strategy. Fishing activities consist mostly of long lining and gillnetting with about half of the fishing effort occuring within the macrophyte fringe areas of the lake. Sporadic fishing activities are carried out in the midlake using indiscriminate multifilament gill nets. Two fishermen foundf used rafts made from stalks of *C.papyrus*, *Typha domingensis* and *Phragmites australis* as fishing boats. Fishermen set their nets in the evening for a night – long fishing, clearing such nets before dawn. But a significant amount of fishing effort occurs during the day, much of which is concentrated along the Lagoon and Goye channel areas and, around Usenge and Siaya beaches. Fishermen intercepted immediately after clearing their nets were able to show our survey team good catches of *O.niloticus*, weighing even more than 500g per individual fish. The current activities of this fishery are probably sustainable, but they lack coordinated management to regulate their long term sustainability.

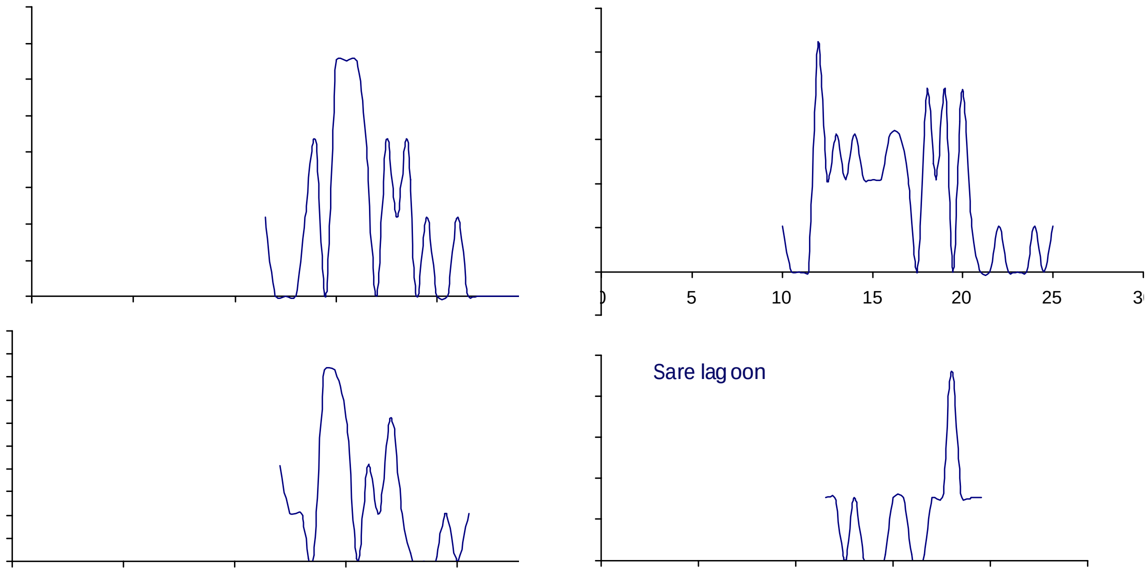


Figure 5. Percentage length frequency distribution of *Lates niloticus* in the major ecological zones of Lake Sare.

Phytoplankton community

Lake Sare has a rich community of phytoplankton (Table 3). The phytoplankton community was dominated by blue-green algae followed by green algae. Only two species of diatoms were identified. Of special interest was the existance of one algae species *Cylindospermopsis africana,* which has been shown to produce toxins known as Cylindrospermopsin.

This toxin has the potential to cause eye and skin irritations, diarrhea, disorders of the nervous system, affects the liver and to some extent the kidney. It also has the potential to stimulate the growth of cancers.

Table 3. The major phytoplankton species identified in Lake Sare.

Cyanophyta (Blue green algae)
Pseudoanabaena sp.
Cyclindospermopsis africana
Anabaenopsis sp.
Planktolyngbya sp.
Anabaena sp.
Microcystis sp.
Chlorophyta (Green algae)
Pediastrum sp.
Solastrum sp.
Ankistrodesmus sp.
Closterium sp.
Closterium aciculare
Coelastrum
Coelomoron sp.
Cosmarium sp.
Bacillariophyta (Diatoms)
Synedra sp.
Nitzschia sp.

Macrophyte community

The macrophyte species were categorized into emergent and submerged species. The dominant emergent species *Cyperus papyrus, Phragmites australis, Typha domingensis, Voacanga thomasii*, *Hibiscus diversifolius,* and *Vossia cuspidata*. *V. thomasii* is a mangrove-like three which was observed to harbour several species of roosting birds. (Table 4).

Table 4. The common macrophyte species identified at Lake Sare.

Emergent plants
Cyperus papyrus
Phragmites australis
Typha domengesis
Voacanga thormarsii
Hibiscus diversifolius
Vossia cuspidate
Ipomea rubens
Cyperus pseudokylingoides
Commelina sp
Melanthera scanders
Ludwiga stolonifea
Sesbania sesban
Sphaeranthus sp.
Hygrophilla auriculeta
Ipomea carica
Polygonum setosulum
Submerged and free floating plants
Eicchornia crassipes
Pistia stratiotes
Nymphaea lotus
Utricularia inflexa

Free floating macrophytes were represented by *Eichhornia crassipes* (Water hyacinth) and *Pistia stratiotes* (Nile water cabbage) occurring mainly in the lagoon. The invasion of the lake by water hyacinth was reported here for the first time and was observed to decrease and its niche being taken over by *Vossia cuspidata* in the lagoon. This ecological succession affected other parts of the shores of neighbouring Lake Victoria.. A flourishing population of floating leafed *Nymphaea lotus* was recorded especially in the northern shores of the main lake. *Ultriculia inflexa* dominated the completely submerged macrophyte population.

The zonation pattern from water to land in this lacustrine environment was represented by *Utricularza inflexa, Nymphaea lotus, Eichhornia crassipes,* and *Vossia cuspidata.* Much of the shoreline had only *H. diversifolious* and *C. papyrus* with climbing species *(Ipomea rubens,* and *I. carica.* Traditional traps made from *Phragmites australis* used to trap fish at the Goye Causeway reduced water velocity and hence increasing the growth of floating plants..

Zooplankton and benthic community

A checklist of the zooplankton and benthic species identified is shown in Table 5. The propotional distribution of abundance (%) is shown in Figure 5. The zooplankton community was dominated by copepods (*Thermocyclops* spp.),a few cladocerans and rotifers (*Brachionus* spp.). The cladoceran densities were possibly low due to predation by fish (Figure 6). Cladoceran, are much larger than the copepods and are therefore preferred by fish more than the smaller and faster copepods and rotifers.

Table 5. Major species of zooplankton and benthic community found in Lake Sare

Copepoda
Calanoida
Cyclopoida
Nauplii of several species
Cladocera
Moina micrura
Bosmina longirostris
D.lumholtzi
Ceriodaphnia cornuta
Diaphanosoma exiscum
Rotifera
Brachionus.calyciflorus
B.angularis
B.falcatus
B.caudatus
Platyias patulus
Trichocerca spp.
Keratella tropica
Epiphanes sp.
Proales sp
Lecane sp.
Asplanchna sp.
Benthic
Chironomid larvae

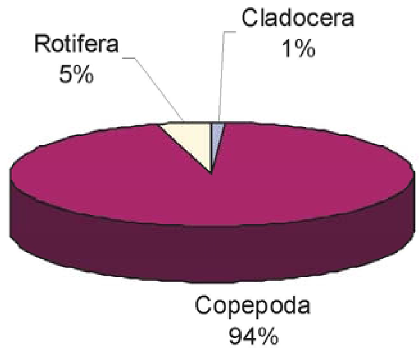


Figure 6. Abundance (%) of the zooplankton community in Lake Sare.

Discussion

Lake Sare is a high priority site for the conservation and management of the fisheries resources of Lake Victoria basin. The lake has several interacting freshwater ecosystems i.e. River Yala system, Lake Victoria system, a lagoon system and adjacent terrestrial ecosystem. All these systems carry a unique aggregation of organisms only differing in species origin. Lake Sare has direct link with Lake Victoria. Fish populations stocked in Lake Sare are likely to find their way to Lake Victoria.

Results obtained in this study show that there is a preponderance of juvenile Nile perch in this ecosystem. The shallowness of this ecosystem (depths < 5m) may not be conducive for large size Nile perch. Therefore, we hypothesize that the juveniles of Nile perch are driven by currents into Lake Sare via the Goye causeway. Lake Sare, the ecosystem is rich in nutrients from the adjacent wetland and river inputs (Mavuti, 1989). When the Nile perch has grown to full size, they then head back to the deeper waters through the Goye causeway and into Lake Victoria. In view of this Lake Sare functions as a nursery ground for the main commercial fish species of Lake Victoria.

Future management and conservation programmes in Lake Victoria should consider the associated ecosystems. For instance, the lake can be used as a launch site for restocking Lake Victoria with juveniles of endangered fish species. If this could be possible, then Lake Sare will have to be made a protected area or national park for sustainability of the restocking program.

Lake Sare and the greater Yala swamp are however, currently undergoing very uncertain changes, ecologically and socio-economically. A large scale reclamation venture that will reclaim upto 40.% of the swamp is already underway (Lake Basin Development Authority and Dominion Group of Companies, *Undated Report*). This project has been approved to improve food security in the area. The short and long term ecological and socio–economic costs of such an undertaking will be enormous. Coupled with this is the accelerated population growth in the region and increased fishing activities.

The changes taking place within and in the vicinity of the lake, threaten the ecological integrity and functioning the lake ecosystem and is a likely cause of conflict between the 'developers', the local communities and the 'conservationists'. It is therefore imperative that proper management and conservation measures be put in place to protect Lake Sare and the associated Yala swamp wetland. This study recommends that ; the existing regulations pertaining to release of agriculture and domestic effluents into Lake Sare need to be enforced to reduce on the rate of eutrophication and pollution and therefore avert potential dangerous impacts to aquatic biota. Secondly, there is need to subject all the development activities in the upstream areas of the lake to rigorous EIA before they are approved for implementation. Thirdly, Lake Sare is a critical habitat for fish survival and thus fisheries activities should be regulated in this ecosystem to allow fish recruitment. Fourthly, an all inclusive management plan for the whole Yala swamp complex is crucial to prevent further degradation of the ecosystem.

Reference

Abila, R., M. Barluenga, J. Engelken, A. Meyer and W. Salzburger, 2004. Population –structure and genetic diversity in a haplochromine cichlids of a satellite lake of Lake Victoria. *Molecular Ecology* (2004). 13, 2589 – 2602.

American Public Health Association (APHA), 1985. *Standard Methods for The Examination of Water and Wastewate.* 16th Edition Port City Press Baltimore MD

Cocquyt, C., Vyverman W. and Compere P, 1993. *A checklist of the algal Flora of the East African Great Lakes (Malawi, Tanganyika and Victoria)*. National Botanic Garden of Belgium, Meise

Cook, C. D. K., E.M. Gut, E.M. Rix, J. Schneller & M. Seitz, 1974. *Water plants of the world: a manual of the identification of the genera of freshwater plants*. Dr W. Junk Publishers, The Hague, Netherlands.

Global Environmental monitoring system Gems, 1992. *WATER operational guide* 3rd Edition

Huber-Pestalozzi G., 1968. *Cryptophyceae, Chloromonadophyseae and Dinophyseae*. Das phytoplankton des Susswars, 3 Teil (ed.G. Huber –Pestalozzi), 2.Aufl.,pp.1-IX + 1-322.Schweizerbartsche-Verlagsbuch-handlung, Stuttgart

Kokwaro J. O., 1972. *Luo English Botanical Dictionary*. East African Publishing House Nairobi

Korovchinsky, N. M., 1992. *Sididae and Holopidae (Crustacean- Daphniformes). Guides to the identification of the macroinvertebrates of the continental waters of the world.* 3 SPB The Hague, 82 pp

Lake Basin Development Authority and The Dominion Group of Companies (Undated report). Environmental Impact Assessment of the Proposed Yala Swamp Irrigation development Scheme.

Maithya, J., 1998. A survey of ichthyofauna of Lake Kanyaboli and other small water bodies in Kenya: Alternative refugia for endangered fish species. Naga, *The ICLARM Quarterly. Jan*. March 1998. PP 54 – 56.

Mavuti, K. M., 1989. Ecology of the Yala swamp. *Resources 1*: 11 – 14.

Okemwa, E., 1981. *A preliminary survey of the fishes and limnology of Lake Kanyaboli and lake Sare in Western Kenya.* Proceedings of the Workshop of KMFRI on Aquatic Resources, Mombasa, Kenya.

Sainty, G. R. & S.W. L. Jacobs, 1994. *Water plants of Australia 3rd edn*. Royal Botanical Gardens, Sydney CSIRO Australia.

Smirnov, N. N., 1996. *Cladocera: the Chydorinae and the Saycinnae (Chydoridae) of the World. Guides to the Identification of the Macroinvertbetrates of the Continental Waters of the World.* SPB Academic Publishing bv the Netherlands, 197 pp.

Lake ecosystems of the Chernobyl exclusion zone under impact of long-term radioactive contamination

Gudkov, Dmitri[1] and Alexandr Nazarov[2]

[1] Institute of Hydrobiology of the National Academy of Sciences of Ukraine, Geroyev Stalingada Ave. 12, UA-04210 Kiev, Ukraine.
[2]State Scientific-Production Enterprise "Chernobyl Radioecological Centre" of the Ministry of Emergency Situations of Ukraine, Shkolnaya Str. 6, UA-07270 Chernobyl, Ukraine
[1]Email:digudkov@svitonline.com

Abstract

The dynamics and distribution of ^{90}Sr, ^{137}Cs, ^{238}Pu, $^{239+240}$Pu and ^{241}Am during 1997–2004 in the main components of the lake ecosystems within the Chernobyl NPP exclusion zone has been studied. The radionuclide content was measured in bottom sediments, water, seston, higher aquatic plants, molluscs and fish in Azbuchin Lake, Dalekoye-1 Lake and Glubokoye Lake. The absorbed dose rate for hydrobionts, living within littoral zone of the researched lakes, due to external irradiation and radionuclides incorporated in tissue was in a range from 0.2 to 3.4 Gy year^{-1}. The highest value was found for hydrobionts from lakes within the embankment territory on the left-bank flood plain of the Pripyat River (Dalekoye-1 Lake and Glubokoye Lake). The molluscs embryos from Dalekoye-1 Lake and Glubokoye Lake were characterised by the highest rate of chromosome aberrations – about 20–25 %, that in 10 times exceeds a spontaneous mutagenesis level for hydrobionts. The maximal rate of chromosome aberrations in roots of higher aquatic plants (about 8 %) has registered in Glubokoye Lake.

Key words: Chernobyl exclusion zone, radioactive contamination, absorbed dose, chromosome aberrations

Introduction

It is 20 years since the Chernobyl NPP accident in 1986, the territory of yet the exclusion zone still remains by an open source of radioactive contamination with complex structure of distribution in various landscapes. The dynamic state of radioactive substances, influences their migration and redistribution in components of ecosystems. Relatively low concentrations of radioactive substances are found in the river ecosystems. Due to high water change rate, the river bottom sediments have undergone decontamination processes (especially during floods and periods of high water). Over the years that passed since the accident occurred, riverine sediments have ceased to play the important role as a secondary source of water contamination. The main sources of radionuclides in rivers are currently the washout from the catchment basin, the inflow from more contaminated water bodies, as well as the groundwater. The closed reservoirs, and in particular the lakes of the inner exclusion zone, have considerably higher levels of radioactive contamination caused by limited water change and by relatively high concentration of radionuclides deposited in the bottom sediments. Therefore, for the majority of lakes the level of radionuclide content is determined mainly by the exchange rates of mobile radionuclide between bottom sediments and water, and the rate of external washout from the catchment.

The objectives of our study were: (1) to identify the distribution of radionuclide in components of lake ecosystems; (2) to study dynamic profiles of radioactive contamination levels in species of different ecological groups; (3) to assess the absorbed dose rate for aquatic organisms living in lakes with different levels and nature of radioactive contamination; (4) to study some cytogenetic effects of radiation exposure on hydrobionts in lakes within the Chernobyl NPP exclusion zone.

Materials and methods

Our research was carried out during 1997–2004 on Azbuchin Lake, Yanovsky Creek, the lakes of the left-bank flood plain of Pripyat River – Glubokoye Lake and Dalekoye-1 Lake within the exclusion zone, defined as a roughly circular area of 30 km radius around the destroyed unit of the Chernobyl NPP. The ^{90}Sr, ^{137}Cs, ^{238}Pu, $^{239+240}$Pu and ^{241}Am content in biological tissues was measured for 28 higher aquatic plant species, 6 species of molluscs, 18 species of fish as well as bottom sediments, water and seston. The results of the radionuclide content measurements in hydrobionts are expressed in Bq kg^{-1} of wet weight at natural humidity. The tendency of the aquatic organisms to accumulate radionuclides, traditionally expressed as the concentration factor (CF), which is determined by calculating the ratio of the specific activity of radionuclides in tissue to the average annual content (for molluscs and fish) or to the average content in the water and in the vegetation (higher aquatic plants). The estimation of the absorbed dose rate for hydrobionts was carried out according to the method of Amiro (1997). The chromosome aberration rate was registered by anaphase method.

Results and discussion

The highest radionuclide activity in water among the studied objects was found in the Azbuchin Lake. During 1997–2004, the content of ^{90}Sr and ^{137}Cs in water of lake reached 120–190 and 18–43 Bq l^{-1} respectively. The radionuclide contamination density values found in the lake bottom sediments for ^{90}Sr, ^{137}Cs, $^{238+239+240}$Pu and ^{241}Am averaged at 6.70, 11.50, 0.24 and 0.22 TBq km^{-2} respectively, with the maximum values of 33.30, 14.40, 1.10 and 0.29 TBq km^{-2}. The ^{90}Sr and ^{137}Cs content in the water of Glubokoye Lake come to 99–120 and 13–14 Bq l^{-1} respectively. The average values of contamination density in the bottom sediments by ^{90}Sr, ^{137}Cs, $^{238+239+240}$Pu and ^{241}Am in 1998 were 2.6, 5.6, 0.07 and 0.06 TBq km^{-2}, with the maximum values being 10.0, 13.7, 0.22 and 0.23 TBq km^{-2} respectively. In Dalekoye-1 Lake the average content of ^{90}Sr and ^{137}Cs in the research period reached 82.5 and 11.8 Bq l^{-1} respectively. The maximum value of radionuclide contamination density in the bottom sediments by ^{90}Sr in 1999 was 18.9, by ^{137}Cs – 15.2, by $^{238+239+240}$Pu – 0.6 and by ^{241}Am – 0.4 TBq km^{-2}. The average values were, accordingly, 4.0, 3.1, 0.08 and 0.08 TBq km^{-2}.

The average specific activity values for ^{90}Sr and ^{137}Cs in water of Yanovsky Creek for the period 1998–2003 were 75.2 and 5.6 Bq l^{-1} respectively. The radionuclide contamination of the bottom sediments of reservoir is extremely heterogeneous, which is obviously caused by the non-uniform character of the nuclear fall-out and by the absence of wind-induced turbulence in deep water. The average content of ^{90}Sr, ^{137}Cs, $^{238+239+240}$Pu and ^{241}Am in bottom sediments was, respectively, 16.3, 14.8, 0.4 and 0.3 TBq km^{-2}. At the same time, within the bounds of silt sediment deposition, some sites with abnormally high density of contamination by ^{90}Sr, ^{137}Cs and $^{238+239+240}$Pu (307.1, 251.6 and 5.3 TBq km^{-2} respectively, which was 20 times higher than the average values in the backwater), were found.

The radionuclide contents in higher aquatic plants in the studied lakes were largely determined by the nature of radionuclide contamination of the water objects and nearby territories, as well as by the hydrochemical regime in the lakes. The latter affects the forms of radionuclides in the reservoirs, thus affecting the level of their bioavailability to the hydrobionts. The specific activity data for the main radionuclides in macrophyte tissues are shown in Table 1.

Table 1. Content of radionuclides in macrophytes (1997–2004), Bq kg^{-1} wet.

Water reservoir	^{90}Sr			^{137}Cs		
	max	**min**	**average**	**max**	**min**	**average**
Glubokoye Lake	14060	67	2212	36470	1215	8730
Dalekoye-1 Lake	5100	200	1808	19470	1167	5170
Azbuchin Lake	24210	730	4895	23860	220	2025
Yanovsky Creek	2200	110	779	1702	38	814

The patterns of ^{90}Sr and ^{137}Cs accumulation have been shown to be species-specific. Among the species with relatively high ^{137}Cs content are the helophytes (air-water plants) of genus *Carex, Phragmites australis*, *Glyceria maxima*, *Typha angustifolia*, as well as strictly water plant species *Myriophyllum spicatum* and *Stratiotes aloides.* The low values of ^{137}Cs activity in all reservoirs were found in plants of family Nymphaeaceae – *Nuphar lutea* and *Nymphaea candida* as well as *Hydrocharis morsus-ranae.* Relatively high content of ^{90}Sr was shown by the species of genus *Potamogeton*. Obviously this is related to this plant's tendency to accumulate large quantities of calcium (which is not washed off during standard sampling) on its surface during photosynthesis. At the same time, calcium carbonate that is removed from the plant could contain 7–20 times more radioactive strontium than the plant tissue (Laynerte, 1977). Thus, *Potamogeton* species makes a good prospective radioecological monitoring object as a specific accumulator of ^{90}Sr.

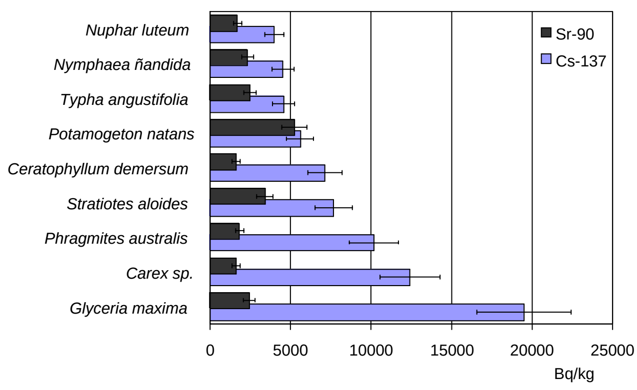


Figure 1. The content of radionuclides in higher aquatic plants of Glubokoye Lake in 2000.

The analysis of the ^{90}Sr and ^{137}Cs content at various vegetative stages have revealed seasonal dynamics of radionuclide accumulation by macrophytes. The majority of higher aquatic plant species showed the increased content and CF of ^{90}Sr and ^{137}Cs at the peak of vegetation (end of July – August) in comparison with the spring and autumn periods.

The content of radionuclides $^{238+239+240}Pu$ and ^{241}Am in higher aquatic plants of the left-bank flood plain of Pripyat River was found, respectively, in the ranges of 1–66 (11) Bq kg^{-1} with CF – 24–4175 and 1–45 (11) Bq kg^{-1} with CF – 83–7458. *Typha angustifolia* showed the highest CF, which was 5–7 times higher than the average CF values for other studied plant species. That allows to consider this species as a specific accumulator of transuranic elements in lakes within the exclusion zone.

Freshwater molluscs are often considered as bioindicators of radionuclide contamination of water objects. These invertebrates accumulate practically all the radionuclides found in water and, due to their high biomass, molluscs play an important part in bioaccumulation processes and radionuclide redistribution in aquatic ecosystems. The ^{137}Cs and ^{90}Sr content in molluscs of lakes of the Chernobyl exclusion zone are shown in Table 2.

Table 2. Content of radionuclides in molluscs (1998–2004), Bq kg^{-1} wet.

Water reservoir	^{90}Sr			^{137}Cs		
	max	min	average	max	min	average
Glubokoye Lake	170300	39770	61830	27150	3156	9067
Dalekoye-1 Lake	62830	17123	32310	2410	847	1523
Azbuchin Lake	87670	34010	51120	4750	2704	3620
Yanovsky Creek	13350	7720	10350	590	430	510

The highest CF for both ^{90}Sr and ^{137}Cs were found in bivalve mollusc *Dreissena polymorpha* and *Unio pictorum*, which are the most active filter molluscs. The highest CF for ^{90}Sr was noted in *Dreissena polymorpha* – in excess of 1100, while for ^{137}Cs the highest CF (about 500) was found in the tissues of *Unio pictorum*. Considerably lower CF was determined for the gastropod species *Lymnaea stagnalis*, *Planorbarius corneus* and *Viviparus viviparus*.

The lowest CF for both ^{90}Sr and ^{137}Cs were found in *Lymnaea stagnalis* (440 and 137 respectively). Whereas the differences in ^{90}Sr CF for gastropods could be explained by their shell morphological structure and its specific weight, the distinctions in value of ^{137}Cs CF are related to the functional ecology and feeding modes of these invertebrates.

Average contents of transuranic elements ^{238}Pu and $^{239+240}Pu$ in mollusc tissues in Glubokoye Lake and Dalekoye-1 Lake were as follows: the lowest value was determined for *Lymnaea stagnalis* – 0.1 and 0.2 Bq kg^{-1} respectively in Dalekoye-1 Lake, 2.7 and 6.4 in Glubokoye Lake. The highest content was determined for *Stagnicolia palustris* from Glubokoye Lake – 14 and 36 Bq kg^{-1} respectively. The highest activity among gastropods was shown in *Planorbarius corneus* – 1 and 2 Bq kg^{-1} respectively from Dalekoye-1 Lake; 25 and 53 Bq kg^{-1} in Glubokoye Lake. *Dreissena polymorpha* from the cooling pond of the Chernobyl NPP showed ^{238}Pu and $^{239+240}Pu$ contents of 3 and 6 Bq kg^{-1} respectively.

The content of ^{241}Am in *Lymnaea stagnalis* tissue was the lowest – in the range of 4–30 (15) Bq kg^{-1} in Dalekoye-1 Lake and 6–51 (27) Bq kg^{-1} in Glubokoye Lake. For *Stagnicolia palustris* from Glubokoye Lake the value was about about 75 Bq kg^{-1}. The highest value was found in *Planorbarius corneus* – 18–29 (24) in Dalekoye-1 Lake and 80–310 (170) Bq kg^{-1} in Glubokoye Lake. The content of ^{241}Am in *Dreissena polymorpha* tissue from the cooling pond of the Chernobyl NPP was at the level of 8 Bq kg^{-1}.

Fish species that are found at the upper levels of the food webs may also constitute a part of human diet and, therefore, are of a particular interest in radioecological research of water ecosystems. The comparative contents of ^{90}Sr and ^{137}Cs in fish of the exclusion zone lakes are represented in Table 3.

Table 3. Content of radionuclides in fish (1998–2004), Bq kg^{-1} wet.

Water reservoir	^{90}Sr			^{137}Cs		
	max	min	average	max	min	average
Glubokoye Lake	3300	660	2582	11000	5200	8520
Dalekoye-1 Lake	13060	410	5103	27020	16110	20030

The concentration of transuranic elements ^{238}Pu, $^{239+240}Pu$ and ^{241}Am was measured in fish of Glubokoye Lake and Dalekoye-1 Lake. The activity of ^{238}Pu in fish tissue was found in the range of 0.4–0.5 (0.4) Bq kg^{-1} with CF – 72–98 (83), $^{239+240}Pu$ – 0.7–0.9 (0.8) Bq kg^{-1} with CF – 68–87 (75) and ^{241}Am – 2.2–10.0 (6.2) Bq kg^{-1} with CF – 367–1667 (1028).

The contents of ^{90}Sr and ^{137}Cs radionuclides in lake fish of the left-bank flood plain of Pripyat River in all cases considerably exceeded maximum permissible level (MPL), according to the standards accepted in Ukraine for fish production: for ^{90}Sr on average 146 times higher (MPL – 35 Bq kg^{-1}), for ^{137}Cs – 134 times (MPL – 150 Bq kg^{-1}). The highest measured values were 373 and 180 times in excess of MPL. The content of ^{90}Sr in fish of the cooling pond practically in all caught specimens also exceeded MPL (on average 8 times higher), with the highest registered values being 43 times higher than MPL. The ^{137}Cs contents in all cases also considerably exceeded MPL – on average 33 times higher, with highest registered values exceeding MPL – 84 times.

The analysis of radionuclide distribution in components of lake ecosystems has shown that about 98–99 % of ^{137}Cs and more than 99 % of transuranic elements of the total radionuclide content has concentrated in the bottom sediments. The content of ^{90}Sr in sediments of lakes, due to higher solubility, amounts to 89–95 %. About 2–10 % of radinuclides concentrated in water and only about 1 % – in biota (Table 4, 5). In this percent a prevailing value for different radionuclides has the macrobenthos species (especially bivalve molluscs) and higher aquatic plants (Figure 2).

Table 4. Total content and ratio of radionuclides in the main components of Dalekoye-1 Lake ecosystem.

Subject of inquiry	^{90}Sr		^{137}Cs		$^{238+239+240}$Pu, ^{241}Am	
	MBq	%	MBq	%	MBq	%
Bottom sediments	37000	95.35	51800	99.11	1100	99.90
Water	1650	4.25	236	0.45	0.27	0.03
Seston	58	0.15	155	0.30	–*	–
Biota	96	0.25	73	0.14	0.81	0.07

* – the measurements were not carried out

Table 5. Total content and ratio of radionuclides in the main components of Glubokoye Lake ecosystem.

Subject of inquiry	^{90}Sr		^{137}Cs		$^{238+239+240}$Pu, ^{241}Am	
	MBq	%	MBq	%	MBq	%
Bottom sediments	444000	89.02	962000	98.64	25900	99.80
Water	50900	10.21	6200	0.64	10	0.04
Seston	800	0.16	2471	0.25	–*	–
Biota	3035	0.61	4598	0.47	42	0.16

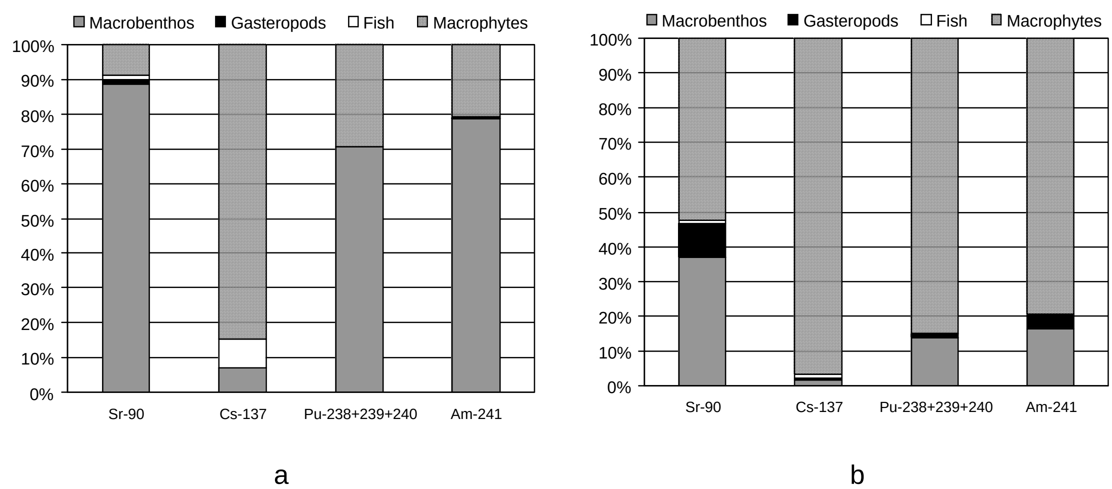


Figure 2. The distribution of radionuclides on the main groups of hydribionts in the biotic component of Dalekoye-1 Lake (a) and Glubokoye Lake (b) ecosystems.

The numerous effects of irradiation on hydrobionts within the Chernobyl exclusion zone are revealed. Some of these effects required for the short period of time for its formation, however it is supposed that an increasing importance will be got by the remote consequences – genetic damages induced by a long-term irradiation. These remote consequences are long-drawn out in time realisation of changes in molecules of heredity, in which the initial molecular damages can be kept for the long period not being shown and being transferred through many generations of cells.

The structural damages of DNA, arising due to ionising radiation impact, are the basic reason of reproductive cells death are registered as chromosome aberrations. The analysis of such cytogenetic effects of irradiation on biosystem are extremely important at study and forecasting of the long-term consequences of the Chernobyl NPP accident.

We studied the rate of chromosome aberrations in cells of freshwater snail (*Lymnaea stagnalis* L.) embryos and in the apical meristem of roots of the higher aquatic plants common reed (*Phragmites australis* (Cav.) Trin. ex. Steud.) and arrowhead (*Sagittaria saggitifolia* L.). The samples has taken in different seasons of 1999–2004 in reservoirs of the Chernobyl NPP exclusion zone (Azbuchin Lake, Dalekoye-1 Lake, Glubokoye Lake, Yanovsky Creek and some rivers of the exclusion zone), characterised by various levels of radioactive contamination and, accordingly, dose rate for hydrobionts. The results of the analyses compared to the data received for hydrobionts from Goloseevo lakes located within Kiev City territory (Figure 3).

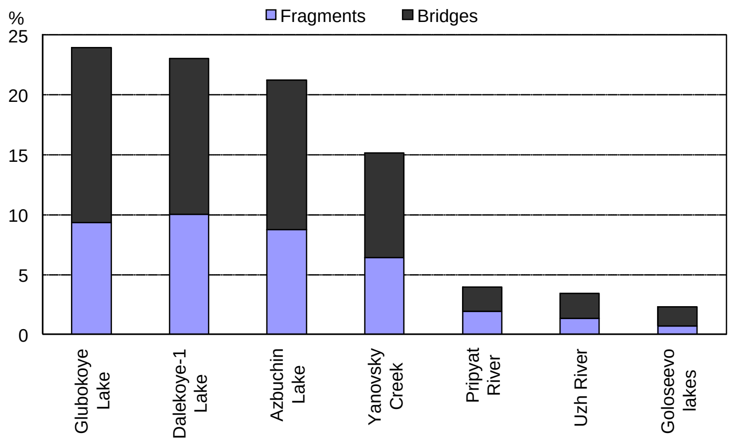


Figure 3. Average rate and spectrum of the main chromosome aberration in cells of the snail embryos from water objects within the Chernobyl NPP exclusion zone in 1999–2004.

The absorbed dose rate for hydrobionts, living within littoral zone of the researched water objects, due to external irradiation and radionuclides incorporated in tissue was in a range from 2.5-04 to 3.4 Gy year^{-1}. The highest value was found for hydrobionts from lakes within the embankment territory on the left-bank flood plain of the Pripyat River (Dalekoye-1 Lake and Glubokoye Lake), the lowest for specimens from the running water objects (Uzh River and Pripyat River). The high level of chromosome aberration in of snail's cells from water objects within the Chernobyl exclusion zone has been registered in comparison with Goloseevo lakes of Kiev City. The molluscs from Dalekoye-1 Lake and Glubokoye Lake were characterised by the maximal rate of chromosome aberration about 20–25 %, that in 10 times exceeds a level spontaneous mutagenesis for hydrobionts. A little bit less rate is registered for snails from Azbuchin Lake and Yanovsky Creek. The chromosome aberration rate of hydrobionts from Goloseevo lakes on average was about 1.5 %, and the maximal rate did not exceed 2.5 %. The maximal aberration rate in roots of higher aquatic plants (7.8 %) has registered in Glubokoye Lake; in plants of Goloseevo lakes this value was about 1.8 %.

The cytogenetic research of aquatic biota within the exclusion zone convinces that the organisation of regular genetic monitoring of the contaminated territories is the important measure, extremely necessary for understanding and forecasting of negative remote consequences of long-term irradiation.

Acknowledgements

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References

Amiro B.D. Radiological Dose Conversion Factors for Generic Non-Human Biota Used for Screening Potential Ecological Impacts. *Journal of Environmental Radioactivity*. 1997. Vol. 35, No 1, 37–51.

Laynerte, P., Seysuma, Z., 1977. The role of littoral-water plants of freshwater reservoirs in accumulation of Ca and Sr (stable and radioactive). Proc. 1st All-Union Conf. "Higher aquatic plants and littoral-water plants, 1977. Borok, Russia, 117–119 (in Russian).

Two ancient lakes in Paniai and Tage, the central mountains range, Papua, Indonesia

Peter E. Hehanussa, Gadis Sri Haryani, and Tjiptasmara

Limnology & Geotechnology, Indonesian Institute of Sciences, Indonesia

Abstract

Lake Paniai and Lake Tage are brother and sister lakes in the mountain range of Papua located at 3°53'S - 136°18'E. Papua Island is the second largest island in the world (after Greenland), two ancient lakes are located in the west end of the Central Mountain Range its highest point at 5,050 m with snow cover. The lake water elevations are at 1742 and 1747 meters above sea level, surronded by 1,800 – 2,625 m Mesozoic to Early Tertiary mountain ranges. The two lakes are oddly located at the island's main water devide that seperate water flow into 120 km rivers flowing north to the Pacific Ocean and 75 km rivers and subsurface (into caves) river flowing to the Arafura Sea in the south.

The two adjoining lakes are narrowly seperated from each other by a dip slope with Jurasic to Cretaceous limestone and marl beds. The large oval shaped Lake Paniai has a maximum depth of 54 m and 160 km^2 in area. The eastern lake rim is shallow sloping but deeper and steeper in the southern border. A well mixed water column was measured in Lake Paniai. The sister lake in the south is the smaller Lake Tage, 52 m deep, elongated in shape, 24 km^2 in area, a thermocline was encountered at 40 m. Secchi disk reading was 397 cm. The south and north lake bottoms are steep while its surrounding above the lake rim formed by eroded limestone.

A large U shape valey at the west end of Lake Tage might be the result of a plunging syncline with its axis along the length of the lake.. Based on the rock distribution in the lake surroundings and the tectonic history, it was considered that both lakes are four million years old.

Key words: Lake Paniai and Tage; ancient lakes; Central Mountain Range, Papua

Introduction

Papua Island is located along the equator and it is bounded in the north by the Pacific Ocean and in the south by Indian Ocean. This is the second largest island in the world, the eastern half belongs to Papua New Guinea and the western half is part of Indonesia. The land area of the Indonesian part of Papua Island is over 400,000 km^2. The island is made out of a complex geology, was part of the acient Gondwana Continent during the very early stage of continental drift era, 450 million years ago. The central mountain range of this island is formed by sediment and other complex rock associations at elevations of over 5000 meters above sea level which is covered by ice/ glaciers. This mountain range is bounded in the southern as well as the northern coastal plain by low lying coastal plain and an areo of coastal swamps with a rich biodiversity.

This central mountain range due to its complex geological history possess a rich mineral variety of copper, gold, silver and other minerals. Thgis has some of the largest copper and gold mining enterprises in the world. Mining started in the early 1970's. At the western end of this mountain range, the geology transformed into a folded mountain range that is also rich in minerals such as coal, oil and natural gas. At the near end of this geological complex that show changes from the Folded Nassau Mountain Range and the Homocline Ridges into the Folded Mountain /Range is the Paniai and Tage sister lakes. Rainfall in this area is high with an average of 3,055 mm annually (1637 – 3953 mm).

These lakes are oddly located at and very near the main-water-divide that seperates the water flows of the island into a norhern flow and the southern flow direction. Both lakes are located in high elevations, at 1742 and 1747 meters above sea level and surrounded by mountains up to 2650 meters. The lake water area are 160 and 24 km^2 respectively and the maximum depth of the lakes was 54 and 52 meters. Northern Paniai shows a weak termocline at 4-10 meters while in Lake Tage it is more clear at 15-30 meters. Lake Tahe drained into Pniai then drained by a river flowing to the south into the Indian Ocean. Lake Paniai and Tage are connected though a limestone bed and have a surface water flow to the south. The catchment area of these lakes are not very large in size but the ammount of water flowing in the river deboucing to the south, shows a rather high discharge. These rivers have created two large caves in the thick limestone beds through which they flow into the Indian Ocean.

In the past few decades population in this area was still sparse, but demographic and cultural schanges have been going on since the last century leaving behind the stone age life style. The city of Enarotali located at the very end of Lake Paniai was recently appointed to become the capital of the new District of Paniai. A population of about 30,000 people from various tribes live in this expanding city. Land transportation with the larger cities at the coastal areas are being build but repeatedly damaged by landslides and rockfall in this mountain area and a highly tectonic area. Air transport was limited but more common and dependable way of communication. Due to this natural background, cost of living in the in the city is very high.

Recently, Lake Paniai and Tage have become important as inter village water transportation and used traditionally for fisheries that support the life of the local people. Tradition and culture of the local people that are closely related to water is considered as the prime of life. Electric power generation in Enarotali is recently provided by diesel generators.

Power generation is expensive due to diesel oil transport by air. Nature's beauty and the mild temperature has great potentials to be developed into tourist industry that could dramatically change the life of people in this area. Scientific explorations of these oddly located lakes, the limnology, biodiversity, and geology as well as the social background are equally important and is a critical input for the better planning so as to sustain future development of this area.

Figure 1. Sister lakes in central Papua Island, Indonesia Note the smaller but higher elevation of Lake Tage on the left and the larger but lower elevation Lake Paniai in the right. The lakes are seperated by a massive limestone bed shown in the picture.

Method

Lake Paniai and Tage are located high in the mountain range that in the past was limited explored and published. Very restricted written and detailed publications are available. Accesibility to the area was limited and if available are expensive. To implement this preliminary survey, support and facilities were provided by the local government authority, by the Bupati of Paniai or head of district and very supportive efforts by his staff members at the Bappeda office in early February 2005. Field observations were made in both lakes, in the debоucing river, and land observations in the surrounding area.

Field surveys in the lake were executed by boat and onland by car and hiking in the small dirt roads. In the lake, several bathymetry transects were measured with a portable echosounder, at selected sites surface water sampled using water samplers, water quality measurements were taken using water quality checker, water tranparancy by secchi disk, and for the location by using handheld GPS. Water samples with elements that easily changed was chemically analysed on the night after field work. BOD and plankton samples were taken in the lake water. A selected set of water samples were taken back for laboratory analyses in Bandung. Chemical laboratory analyses were done in the laboratory following the standard Indonesian water analyses procedures.

Secondary data were gathered from geological maps, climate stations, demographi statistical data, and Landsat satelite images. Geological exploration in the area were extensively done in the late 1980 to 1990's due to the high potential of minerals in the area. Expansion in exploitation of the mega-mining activities of gold and copper in the Timika area could in the future enrich available scientific information on these unique ancient lakes and their endemic species. This information will be needed in setting up a future development plan of the area.

Results and discussion

Geology and tectonics

The geology of the area is characterised by series of rocks ranging in age from Trias to Pliocene. The Tipuma Formation is the oldest (Trias-Jura) and consists of sandstone and shales. Then, above it is the Kemblangan Group (Jura-Createaceous) consisting of quartz sanstone and claystone. The Nuguinea Limestone Group (Cretaceous-Uppe4r Miocene) consists of sandstone and marls, while the Buru Formation (Mio-Pliocene) of claustone, sandstone, and conglomerate is covered by the youngest rock formation of the Buru Formation (Pleistocene).

The tectonic history of this island is one of continuous wrapping, folding, and faulting events that end up in the late Pleistocene. The last tectonic event, which is the breaking up and the proposed spreading of the uplifted mountain range produced these lakes. The formation of these lakes is rather strange since they were formed at the very top of the mountain range, at the main water divide that separate the surface water flows. From these facts, it could be deduced that this is a tectonic lake with no hiatus in depositional sequences it was also deduced that this is an ancient lake.

Bathymetry

The bathymetry of a lake could provide information on the sedimentation processess in the lake, explained its hydrodynamics, and provide informations on its tectonic history. Bathymetric measurements were taken through continuous lake bottom mapping using a echosounder and using GPS. Several measurement profiles are provided, each from Lake Paniai and Lake Tage. The horizontal and vertical scales are not the same for all transects.

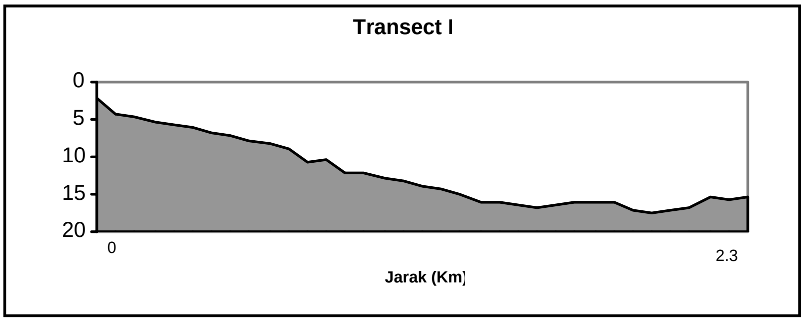

Figure 2. Depth profile on a transect from Enarotali to Tage, ake Paniai.

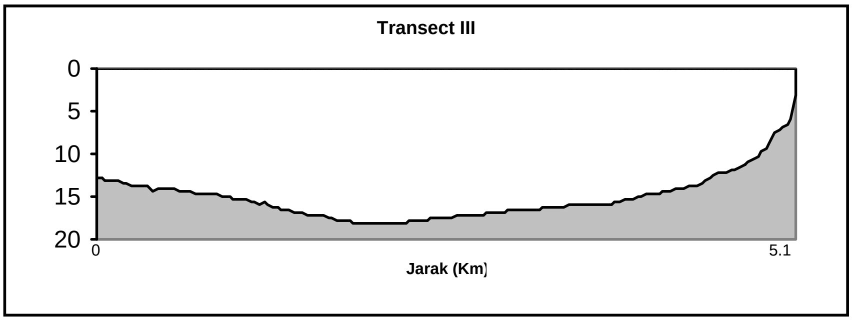

Figure 3. Depth profile along a south-north transect in Lake Paniai.

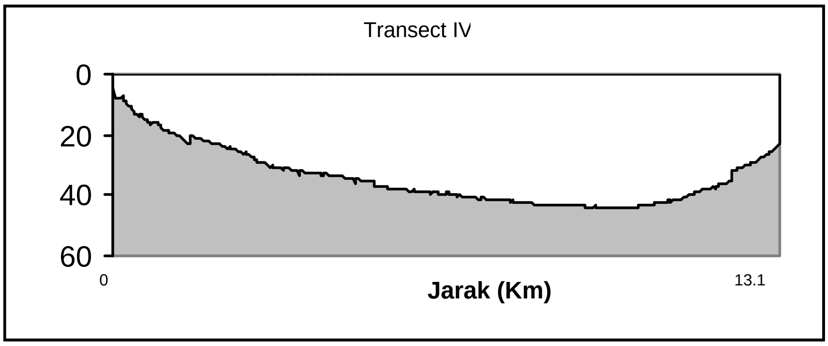

Figure 4. A depth profile along the center of Lake Paniai,from north to south.

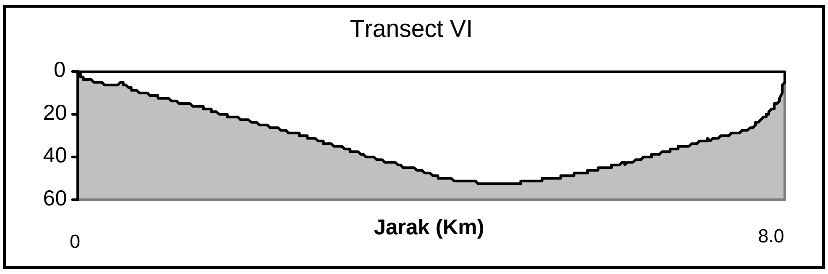

Figure 5. A west to east transect along the elongated center of Lake Tage, from west to east.

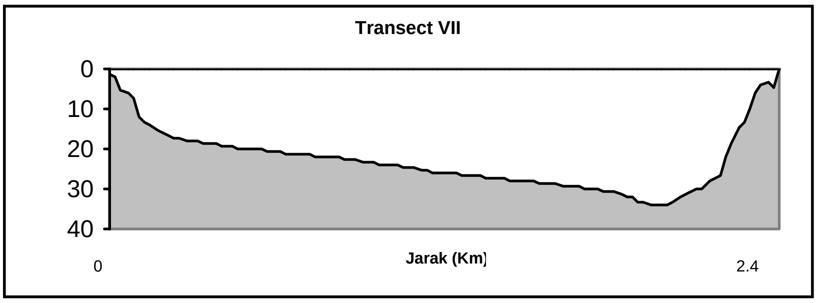

Figure 6. A south to north depth transect along the narrow width of Lake Tage.

Water quality

Table 1 . Water quality of Lake Paniai.

No	Parameter	Unit	LOCATION 1 WPT 186	2 WPT 194	3 WPT 196 Permkan	4 WPT 196 Dasar	5 WPT 203	METHOD
			4 / 2- 05 11.15	5 / 2 –05 08.40	5 / 2 –05 10.55	5 / 2 –05 10.55	5/ 2 –05 14.00	
1.	Conductivity	μmhos/cm	161	188	162	170	163	SNI 06- 2413 - 1991
2.	Kekeruhan	NTU	0.7	2.8	0.5	4.0	1.8	SNI 06- 2413 - 1991
3.	Suhu	^oC	23.5	20.8	23.3	21.8	21.2	SNI 06- 2413 - 1991
4.	Warna	Unit PtCo	1.2	1.0	1.8	1.1	1.0	SNI 06- 2413 - 1991
5.	Zat Terlarut	mg/L	97	113	97	102	98	SNI 06- 2413 - 1991
6.	Zat Tersuspensi	mg/L	12	8.0	6.0	12	10	SNI 06- 2413 - 1991
7.	Alkaliniti	mg/L $CaCO_3$	-	84	74	76	73	SNI 06- 2413 - 1991
8.	Amoniak bebas	mg/L NH_3-N	0.008	0.003	tt	0.010	0.012	Perhitungan
9.	Amoniak total	mg/L NH_3-N	0.536	0.448	0.379	0.914	0.623	SNI 06- 2479 - 1991
10.	Besi	mg/L Fe	tt	0.03	tt	0.78	0.06	SNI 06- 2523 - 1991
11.	Besi total	mg/L Fe	0.627	0.05	0.02	0.99	0.11	SNI 06- 2523 - 1991
12.	Boron	mg/L B	tt	tt	tt	tt	tt	SNI 06- 2481 - 1991
13.	Detergen	mg/L	tt	tt	tt	tt	tt	SNI 06- 2476 - 1991
14.	Fenol	mg/L	tt	tt	tt	tt	tt	SNI 06- 2469 - 1991
15.	Fluorida	mg/L F	tt	tt	tt	tt	tt	SNI 06- 2482- 1991
16.	Fosfat orto	mg/L PO_4 -P	0.024	0.012	0.015	0.015	tt	SNI 06- 2483 - 1991
17.	Fosfat total	mg/L PO_4 -P	0.112	0.042	0.040	0.070	0.020	SNI 06- 2483 - 1991
18.	Kadmium	mg/L Cd	tt	tt	tt	tt	tt	SNI 06- 2466 - 1991
19.	Kadmium total	mg/L Cd	tt	tt	tt	tt	tt	SNI 06- 2466 - 1991
20.	Kalium	mg/L K	0.92	1.3	0.90	1.0	0.90	SNI 06- 2427 - 1991
21.	Kalsium	mg/L Ca	23	26	23	23	23	SNI 06- 2429 - 1991
22.	Kesadahan	mg/L $CaCO_3$	73	81	73	73	73	SNI 06- 4161 - 1991
23.	Klorida	mg/L Cl	4.0	6.6	4.0	5.3	5.3	SNI 06- 2431- 1991
24.	K O B (B O D)	mg/L O_2	2.4	2.4	2.3	3.5	2.6	SNI 06- 2503 - 1991
25.	K O K (C O D)	mg/L O_2	6.0	6.0	6.0	10	6.5	SNI 06- 2504 - 1991
26.	Kromium	mg/L Cr VI	tt	tt	tt	tt	tt	SNI 06- 2511 - 1991
27.	Kromium total	mg/L Cr VI	tt	tt	tt	tt	tt	SNI 06- 2511 - 1991
28.	Magnesium	mg/L Mg	3.8	3.9	3.8	3.8	3.8	SNI 06- 2430 - 1991
29.	Mangan	mg/L Mn	tt	0.01	tt	0.02	0,01	SNI 06- 2497 - 1991
30.	Mangan total	mg/L Mn	0.04	0.03	0.01	0.03	0,03	SNI 06- 2497 - 1991
31.	Minyak & Lemak	mg/L	tt	tt	tt	tt	tt	SNI 06- 2502 - 1991
32.	Natrium	mg/L Na	2.9	5.2	3.1	4.9	3.4	SNI 06- 2428 - 1991
33.	% Na	-	7.8	12	8.3	13	9.1	Perhitungan
34.	Nikel	mg/L Ni	tt	tt	tt	tt	tt	SNI 06- 2520 - 1991
35.	Nikel total	mg/L Ni	tt	tt	tt	tt	tt	SNI 06- 2520 - 1991
36.	Nilai $KmnO_4$	mg/L $KmnO_4$	8.0	6.6	5.6	7.7	7.6	SNI 06- 2506 - 1991
37.	Nitrat	mg/L NO_3-N	0.087	0.111	0.022	0.017	tt	SNI 06- 2480 - 1991
38.	Nitrit	mg/L NO_2-N	tt	tt	tt	tt	tt	SNI 06- 2484 - 1991
39.	Nitrogen organik	mg/L N	0.169	0.011	0.011	0.020	0.013	SNI 06- 2478 - 1991
40.	Oksigen Terlarut	mg/L O_2	5.9	5.9	5.5	1.0	5.2	SNI 06- 2424 - 1991
41	p H	-	7.4	7.1	7.6	7.3	7.5	SNI 06- 2413 - 1991
42	R S C	-	-	0.02	0.01	0.06	0	Perhitungan
43	S A R	-	0.15	0.25	0.16	0.25	0.17	Perhitungan
44.	Seng	mg/L Zn	0.020	0.030	0.010	tt	0.020	SNI 06- 2507 - 1991

45.	Seng total	mg/L Zn	0.050	0.080	0.040	0.030	0.070	SNI 06- 2507 - 1991
46.	Silika reaktif	mg/L SiO_2	14	10	11	12	12	SNI 06- 2477 - 1991
47.	Sulfat	mg/L SO_4	1.5	1.0	1.1	1.1	1.0	SNI 06- 2426 - 1991
48.	Tembaga	mg/L Cu	tt	tt	tt	tt	tt	SNI 06- 2514 - 1991
49.	Tembaga total	mg/L Cu	tt	tt	tt	tt	tt	SNI 06- 2514 - 1991
50.	Timbal	mg/L Pb	tt	tt	tt	tt	tt	SNI 06- 2517 - 1991
51.	Timbal total	mg/L Pb	tt	tt	tt	tt	tt	SNI 06- 2517 - 1991

Table 2. Water quality of Lake Tage, Papua, Indonesia.

No	Parameter	Unit	LOCATION				
			6 WPT 213 Permkan	7 WPT 213 Dasar	8 WPT 220 S.Dimiya	9 WPT 225 D. Tigi	METHOD
			7/ 2- 05 10.30	7 / 2 –05 10.30	7 / 2 –05 12.35	8/ 2 –05 11.00	
1.	Conductivity	μmhos/cm	205	220	206	180	SNI 06- 2413 - 1991
2.	Kekeruhan	NTU	0.8	5.2	2.5	1.7	SNI 06- 2413 - 1991
3.	Suhu	^oC	22.5	21.1	22.8	24.1	SNI 06- 2413 - 1991
4.	Warna	Unit PtCo	1.0	1.2	1.2	2.8	SNI 06- 2413 - 1991
5.	Zat Terlarut	mg/L	123	132	123	108	SNI 06- 2413 - 1991
6.	Zat Tersuspensi	mg/L	10	14	8.0	16	SNI 06- 2413 - 1991
7.	Alkaliniti	mg/L $CaCO_3$	100	103	94	96	SNI 06- 2413 - 1991
8.	Amoniak bebas	mg/L NH_3-N	0.011	0.022	0.001	0.005	Perhitungan
9.	Amoniak total	mg/L NH_3-N	0.386	1.158	0.106	0.223	SNI 06- 2479 - 1991
10.	Besi	mg/L Fe	tt	0.14	tt	tt	SNI 06- 2523 - 1991
11.	Besi total	mg/L Fe	0.12	0.21	0.04	0.27	SNI 06- 2523 - 1991
12.	Boron	mg/L B	tt	tt	tt	tt	SNI 06- 2481 - 1991
13.	Detergen	mg/L	tt	tt	tt	tt	SNI 06- 2476 - 1991
14.	Fenol	mg/L	tt	tt	tt	tt	SNI 06- 2469 - 1991
15.	Fluorida	mg/L F	tt	tt	tt	tt	SNI 06- 2482- 1991
16.	Fosfat orto	mg/L PO_4 -P	tt	tt	tt	tt	SNI 06- 2483 - 1991
17.	Fosfat total	mg/L PO_4 -P	0.026	0.031	0.032	0.028	SNI 06- 2483 - 1991
18.	Kadmium	mg/L Cd	tt	tt	tt	tt	SNI 06- 2466 - 1991
19.	Kadmium total	mg/L Cd	tt	tt	tt	tt	SNI 06- 2466 - 1991
20.	Kalium	mg/L K	1.0	1.0	1.3	1.0	SNI 06- 2427 - 1991
21.	Kalsium	mg/L Ca	29	29	29	26	SNI 06- 2429 - 1991
22.	Kesadahan	mg/L $CaCO_3$	89	89	91	77	SNI 06- 4161 - 1991
23.	Klorida	mg/L Cl	3.3	4.0	5.3	4.0	SNI 06- 2431- 1991
24.	K O B (B O D)	mg/L O_2	2.2	2.4	2.5	2.2	SNI 06- 2503 - 1991
25.	K O K (C O D)	mg/L O_2	4.0	6.0	6.0	4.2	SNI 06- 2504 - 1991
26.	Kromium	mg/L Cr VI	tt	tt	tt	tt	SNI 06- 2511 - 1991
27.	Kromium total	mg/L Cr VI	tt	tt	tt	tt	SNI 06- 2511 - 1991
28.	Magnesium	mg/L Mg	4.0	4.0	2.1	3.0	SNI 06- 2430 - 1991
29.	Mangan	mg/L Mn	tt	0.15	tt	tt	SNI 06- 2497 - 1991
30.	Mangan total	mg/L Mn	0.04	0.21	0.03	0.06	SNI 06- 2497 - 1991
31.	Minyak & Lemak	mg/L	tt	tt	tt	tt	SNI 06- 2502 - 1991
32.	Natrium	mg/L Na	4.3	4.5	9.4	5.3	SNI 06- 2428 - 1991
33.	% Na	-	8.5	8.9	20	13	Perhitungan
34.	Nikel	mg/L Ni	tt	tt	tt	tt	SNI 06- 2520 - 1991
35.	Nikel total	mg/L Ni	tt	tt	tt	tt	SNI 06- 2520 - 1991
36.	Nilai $KmnO_4$	mg/L $KmnO_4$	5.5	5.9	6.3	9.1	SNI 06- 2506 - 1991
37.	Nitrat	mg/L NO_3-N	0.040	0.041	0.066	0.089	SNI 06- 2480 - 1991
38.	Nitrit	mg/L NO_2-N	tt	tt	tt	tt	SNI 06- 2484 - 1991
39.	Nitrogen organik	mg/L N	0.012	0.015	0.030	0.015	SNI 06- 2478 - 1991
40.	Oksigen Terlarut	mg/L O_2	5.7	1.0	5.3	5.7	SNI 06- 2424 - 1991
41	p H	-	7.7	7.6	7.1	7.6	SNI 06- 2413 - 1991
42	R S C	-	0.27	0.28	0.06	0.23	Perhitungan
43	S A R	-	0.20	0.21	0.45	0.26	Perhitungan
44.	Seng	mg/L Zn	0.02	0.03	0.21	0.06	SNI 06- 2507 - 1991
45.	Seng total	mg/L Zn	0.07	0.08	0.38	0.07	SNI 06- 2507 - 1991
46.	Silika reaktif	mg/L SiO_2	10	10	14	18	SNI 06- 2477 - 1991
47.	Sulfat	mg/L SO_4	2.0	1.4	1.4	6.0	SNI 06- 2426 - 1991
48.	Tembaga	mg/L Cu	tt	tt	tt	tt	SNI 06- 2514 - 1991
49.	Tembaga total	mg/L Cu	tt	tt	tt	tt	SNI 06- 2514 - 1991
50.	Timbal	mg/L Pb	tt	tt	tt	tt	SNI 06- 2517 - 1991
51.	Timbal total	mg/L Pb	tt	tt	tt	Tt	SNI 06- 2517 - 1991

Table 3. Plankton and Chlorophyl-a analyses in Lakes Paniai and Tage, Papua.

No.	**Species**	**Number of Individual/ Liter of water LOKASI**			
		WPT 186 D. Paniai	WPT 196 D. Paniai	WPT 213 D. Tage	WPT 225 D. Tigi
Phytoplankton					
1.	*Closterium sp.*	180	220	140	-
2.	*Microcystis sp.*	700	420	280	340
3.	*Muogeotias sp.*	300	100	-	-
4.	*Schroderia sp.*	-	-	-	160
5.					
Zooplankton					
1.	*Brachionus sp.*	-	60	40	-
2.	*Cyclops sp.*	-	-	-	20
3.	*Daphnia sp.*	-	-	20	-
4.	*Nauplius sp.*	-	-	-	20
5.					
Jumlah Individu (N)		1180	800	480	540
Jumlah Species (S) / Liter		3	4	4	4
Indek Keanekaragaman (H)		0,94	1,15	1,10	0,90
Chlorophyl-@ (mg / m^3)		0,78	-	0,70	0,62

Table 4. Eutrophication level, OCDE (1982)

No	**Trophic level**	**N Total (mg/L)**	**P Total (mg/L)**	**Chlorophyl-@ (mg/m^3)**	**Tranparancy (M)**
1.	Oligotrofik (O)	0,371 – 1,181	0,0048 – 0,0133	0,8 – 3,4	5,9 – 16,5
2.	Mesotrofik (M)	0,485 – 1,170	0,0145 – 0,049	3,0 – 7,4	2,4 – 7,4
3.	Eutrofik (E)	0,861 – 4,081	0,048 – 0,189	6,7 – 31	1,5 – 4,0
4.	Hypertrofik (H)	0,1 – 0,15	0,75 – 1,2	-	-
	Trophyc level				
1.	WPT 186 (D.Paniai)	0,642	0,112	0,78	3,86
2.	WPT 196 (D.Paniai)	0,412	0,040	-	3,62
3.	WPT 213 (D.Tage)	0,438	0,026	0,70	3,97
4.	WPT 225 (D.Tigi)	0,327	0,028	0,62	-

Concluding remarks

Lake Paniai and Tage in the Central Mountain Range of Papua Island, Indonesia are two ancient lakes, located oddly near the main water divide that seperate surface water flow of the island. The lake water depth are 54 and 52 meters respectively but with a different thermocline depth. Lake Tage has deeper and more clear thermodine than lake Pamai. The high elevation of these lakes and their isolated location may contain valuable scientific information on their geological history, their limnology, and their possible content of endemic species. Spatial planning of the lake and the surrounding area should be initiated together with the collection of scientific information that could be beneficial for making a sustainable development plan.

References

Annonym, 2003. *Pengkajian Sumber Daya Air untuk Pembangunan DPS Memberamo dan Danau Paniai*, Water Resources Potentials of the Memberamo Catchments area and the Lake Paniai, Dept.of Public Works, Bandung.

------- , 2003. *Baseline Study for the Developments of the Paniai Lake Area.* Regency of Paniai, Province of Papua.

Bemmelen, R.W. van 1970. *The Geology of Indonesia.* 2nd edition, 732 pp. Martinus Niihoff, the Netherlands.

Pangabean H., 1981. *The Origin of Lakes in the Nassau Mountain Range, Papua*. PIT X, Ikatan Ahli Geologi Indonesia, Bandung.

Kumoro, Yugo and M. Ruslan, 1993. *Geological Evaluation of the Road Transect Between Waghete-Enarotali-Kumopa.* Papua, Indonesian Institute of Sciences, Bandung.

Phosphorus budget and effect of phosphorus recovery in Lake Biwa watershed

Hideaki Nagare and Hiroshi Tsuno

Department of Urban and Environmental Engineering, Kyoto University, oshida-honmachi, Sakyo-ku, Kyoto 606-8501, Japan
Email: nagare@water.env.kyoto-u.ac.jp

Abstract

This study investigated phosphorus budget in Lake Biwa watershed, and discussed its relationship with water quality in the lake, followed by the discussion of the application of phosphorus recovery technology in watershed. 3,130 tons of phosphorus was disposed in watershed in a year. This amount was around 4 times of the input to the lake through rivers, and 12 times of the phosphorus mass in lake water. In agriculture and livestock industry, 1,920 tonP/yr of waste was generated to make 750 tonP/yr of products in the watershed. This inefficiency must be eliminated to improve water quality in Lake Biwa. Phosphorus recovery technology could recover almost the same amount of phosphorus as that in farmland products harvested from the watershed.

Key words: Phosphorus, budget, watershed.

Introduction

Phosphorus is one of the nutrients that stimulate eutrophication in lakes. Many lakes in the world are suffering from this issue resulting from the increase in phosphorus concentration in the lake.

Phosphorus is brought into a watershed in various ways; contained in fertilizer distributed to farm land, in food for human and livestock or in materials for industrial manufacturing. Discussion of the way for preservation of water quality in a lake, especially for prevention of eutrophication, should stand on the mass balance analysis taking into account such input to the watershed.

This study investigates phosphorus budget in Lake Biwa watershed focusing on agriculture, livestock industry and food consumption in dairy life, and discusses its relationship with water quality in the lake, followed by the discussion of the application of phosphorus recovery technology in watershed.

Materials and methods

Lake Biwa and its watershed

Lake Biwa is the largest lake in Japan with a surface area of 674 km^2 and a volume of 27.5 km^3 (ILEC, 1993). It is also the largest water resource supplying municipal and industrial water for 14 million residents in the Osaka/Kyoto/Kobe megalopolis including 1.4 million people in its watershed (Shiga Prefecture, 2005a).

Figure 1 gives the location of the lake and its watershed indicated as solid line. The border of Shiga Prefecture (local government) is shown as dashed line in the figure as well. Since the area of Shiga Prefecture is almost as same as the area of Lake Biwa watershed as shown in Fig. 1, it makes sense to take the area of Shiga Prefecture as the watershed of the lake.

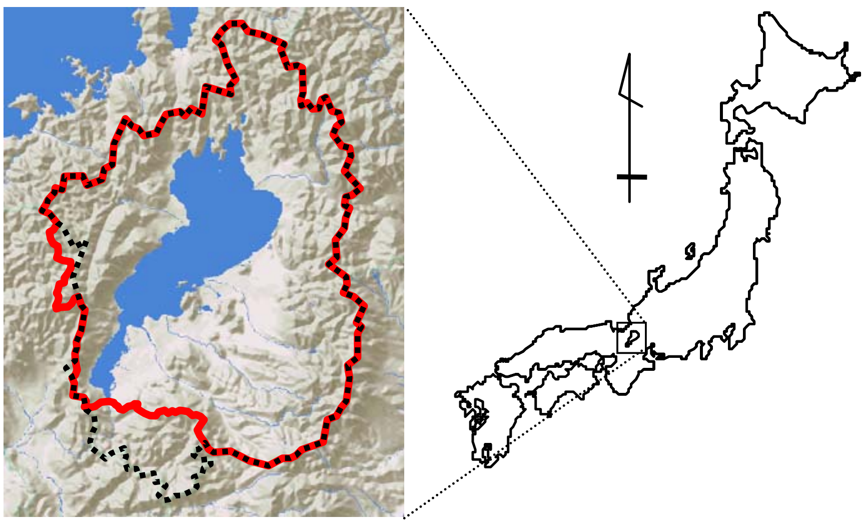

Figure 1. Location of Lake Biwa, borders of watershed and Shiga Prefecture. Dashed line indicates the border of the prefecture, while solid line expresses the watershed.

Land use pattern in year 2002 in Shiga Prefecture is shown in Figure 2. 70% of the area is occupied by forest (51.2%) and water (19.8%). Most of people live in south-eastern area by the lake. The area is equivalent to 9.6% of the prefecture. 13.8% of the area is used for agriculture, most of which is for wet-rice cultivation.

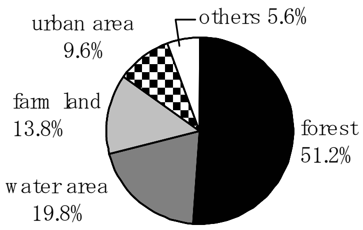

Figure 2. Land use pattern in Shiga Prefecture in year 2002.

The total area of the prefecture is 4,017 km^2 to which most of the Lake Biwa watershed is included as shown in Figure 1.

The lake consists of two parts: the north and the south basins. The characteristics of the lake and two basins are summarized in Table 1. The north basin is the major part of the lake with a maximum depth of 104 m and a water volume of 27.3 km^3. The south basin lies at the outlet of the lake. Water quality is vertically uniform in this basin due to its shallow depth.

Table1. Morphological parameters of Lake Biwa (ILEC, 1993).

Parameter	Units	North Basin	South Basin	Total
Surface Area	(km^2)	616	58	674
Max. Depth	(m)	104	8	104
Mean Depth	(m)	44	3.5	41
Volume	(km^3)	27.3	0.2	27.5

Phosphorus concentration change with years from 1970s is shown in Fig. 3. Water quality of this lake has shifted from an oligotrophic to a mesotrophic state since 1960s due to the increase in pollutants from surrounding areas following high economic development (Okuda and Kumagai, 1995). As one of the counter measures for this issue, sewage system has been constructed in the watershed since 1970. The coverage of the system reached to 72.6% in 2002. Due to the effort including such construction of sewage, the lake has been relieved from catastrophe. However, water bloom can still be seen in the lake every year, and phosphorus concentration in the north basin, the main body of the lake, has not been greatly improved since 1970s as shown in Fig. 2.

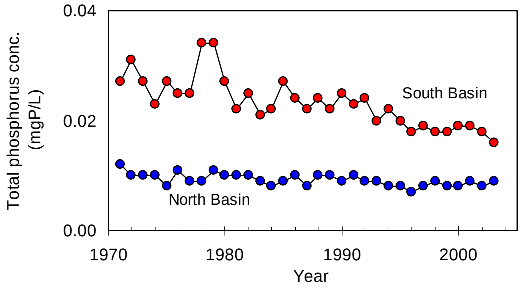

Figure 3. Phosphorus concentration change in Lake Biwa (Shiga Prefecture, 2005b).

Estimation of phosphorus budget and flow in the watershed

In the estimation of the phosphorus budget, fertilizer use and crop harvesting in agriculture, feeding for animals and shipping of products in livestock industry, and food and waste in daily human life were taken into consideration in this study. The analysis was done for the year of 2002.

Agriculture

In the field of agriculture, phosphorus is supplied to farm land as fertilizer, and is harvested as crop products.

The amount of phosphorus in fertilizer was calculated as the product of planted area (Kinki Regional Agricultural Administration Office, 2003) and application rate of fertilizer (The Ministry of Agriculture, Forestry and Fisheries of Japan, 1996).

The amount of phosphorus harvested in farm land was calculated as the product of amount of crops harvested in the prefecture (Shiga Prefecture, 2003a) and phosphorus content of the crops (JST, 2000).

Livestock industry

Phosphorus is fed to animals, such as cattle, swine and chickens, to be livestock products including meat, milk and eggs. The input was calculated as the sum of phosphorus in wastes and products. Animals excrete waste during breeding. The number of livestock (Shiga Prefecture, 2003b) was multiplied with phosphorus emission factor (Harada, 1998) to calculate the amount of phosphorus excreted from animals.

The amount in livestock products was estimated as the product of shipment amount (Kinki Regional Agricultural Administration Office, 2003) and phosphorus content in the products (JST, 2000).

Dairy human life

People eat and waste in daily life, meaning consumption and excretion of phosphorus. The intake of food per person per day was derived from Table on Demand and Supply of Food (MAFF, 2003). The amount food consumed in Shiga Prefecture was then calculated as the product of intake of food per person and population of the prefecture (MIC, 2003).

Some of the food was cultivated and produced in the prefecture. The amount was estimated as shown in (1) above. The deficit was regarded as the transport from outside the prefecture.

The amount of phosphorus in food was calculated multiplying phosphorus content (JST, 2000) with the amount of food.

Results and discussion

Phosphorus material flow in watershed

Phosphorus budget in agriculture and livestock industry

Phosphorus budget in agriculture and livestock industry was evaluated as shown in Fig. 4. The amount of input to agriculture as fertilizer was estimated to be 2,400 tonP/yr, while 360 tonP/yr was supplied to animals as food stuff in livestock industry. The total of 2,760 tonP/yr was supplied in both sections. On the other hand, the recovery as products was estimated only 750 tonP/yr, and reuse of animal waste as fertilizer was 90 tonP/yr. This indicates the difference of input and output, 1,920 tonP/yr was left in watershed unrecovered. This amount is equivalent to 70% of the input.

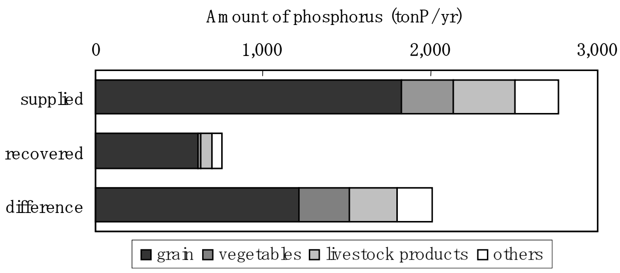

Figure 4. The amounts of phosphorus supplied and recovered as products in the field of agriculture and livestock industry. The difference indicates the amount remained in watershed unused.

Phosphorus flow in the watershed

Considering the fishing from the lake, the phosphorus flow was the discussed in Fig. 5. The rates of solid waste generated in food processing were derived from Table on Demand and Supply of Food (MAFF, 2003).

The input to agriculture and livestock industry was estimated to be 2,760 tonP/yr, 90 tonP/yr of which was from animal waste as discussed above. Therefore, the difference of 2,670 tonP/yr was newly supplied in these sectors. 750 tonP/yr was recovered as products, 650 tonP/yr of which was used as coarse food for residents as well as 460 tonP/yr of input from outside the prefecture. 900 tonP/yr was consumed as real food in daily life, producing same amount of waste. In addition to this 900 tonP/yr of waste, 320 tonP/yr of solid waste was generated in food processing. Including 1,920 tonP/yr of unintended stock in agriculture and livestock industry, 3,130 tonP/yr was disposed in the watershed in all.

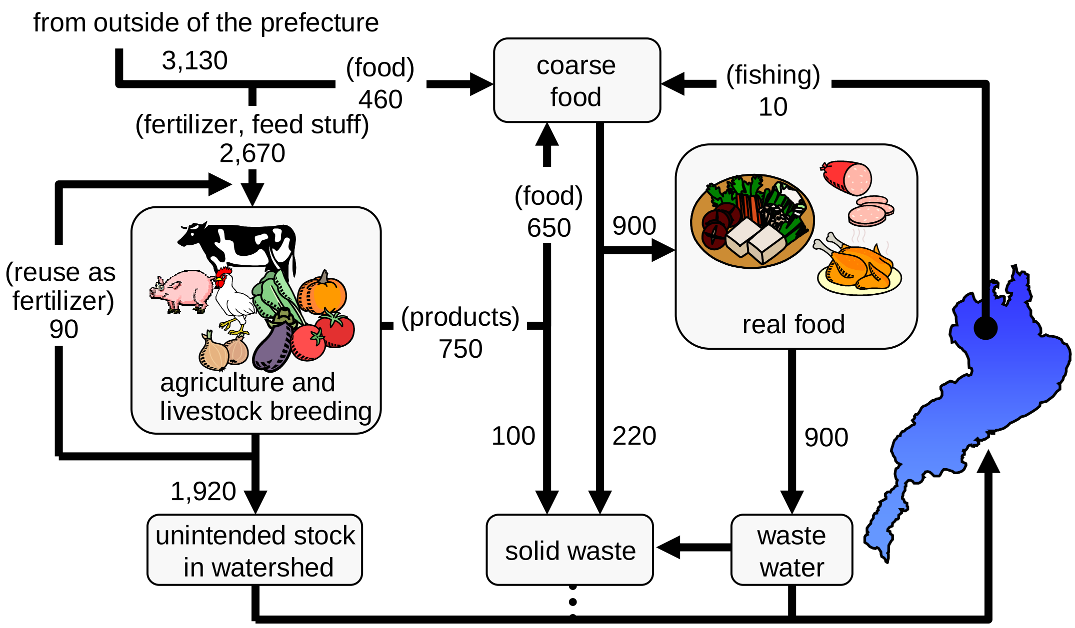

Figure 5. Phosphorus material flow in Shiga Prefecture.

Phosphorus load from watershed to Lake Biwa through rivers has been estimated in a wide range. Then, the average of them was calculated to be 720 tonP/yr (Nagare, 2000). Phosphorus mass in water body of Lake Biwa was estimated as well: that was around 260 tonP (Nagare et al., 2001). The amount of 3,130 tonP/yr disposed in watershed is around 4 times of the input through rivers, and 12 times of the mass in lake water. From this result it is apparent that phosphorus waste in watershed has to be reduced in order to improve water quality in the lake.

3. Effects of introducing phosphorus recovery technology

There has been developed a new water treatment technology of phosphorus recovery from wastewater (Saktaywin et al, 2001). With this technology, phosphorus is removed from wastewater as hydroxyapatite to be reused in farm land as fertilizer. This kind of technology can increase the recirculation rate of phosphorus in watershed, resulting in reduction of phosphorus input to the watershed and waste generated. Then the effect of application of this technology was evaluated.

The technology can recover more than 80% of phosphorus in wastewater. Thus, 720 tonP (=0.8 x 900) could be recovered in this watershed in a year. This is almost same as the 750 tonP/yr in products from agriculture and livestock industry. This means almost enough phosphorus could be recirculated with this technology.

However, there is another challenge to be taken: that is the reduction of waste in agriculture and livestock industry. These sectors generated 1,920 tonP of waste to gain 750 of products. The recovery ratio of products must be increased to improve water quality in the lake and to construct sustainable society.

Conclusion

This study investigated phosphorus budget in Lake Biwa watershed, and discussed its relationship with water quality in the lake, followed by the discussion of the application of phosphorus recovery technology in watershed. Main conclusions derived in this study are as follows:

1. 3,130 tons of phosphorus was disposed in watershed in a year. This amount was around 4 times of the input to the lake through rivers, and 12 times of the phosphorus mass in lake water.
2. In agriculture and livestock industry, 1,920 tonP/yr of waste was generated to make 750 tonP/yr of products. This inefficiency must be eliminated to improve water quality in Lake Biwa.
3. Phosphorus recovery technology could recover almost the same amount of phosphorus as that in farmland products harvested from watershed.

Acknowledgements

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References

Harada Y. (1998) Property of wastes from livestock animals and basic idea for treatment or reuse. Available from URL: http://leio.lin.go.jp/syoko/data05/data05b.html.

International Lake Environmental Committee (ILEC) (1993) *Data book of world lake environments 1. Asia and Oceania*.

Japan Science and Technology Agency (JST) (2000) *Food Composition Database*. Available from URL: http://food.tokyo.jst.go.jp/.

Kinki Regional Agricultural Administration Office (2003) *Statistics on agriculture, forestry, and fishery*.

Ministry of Internal Affairs and Communications (MIC) (2003) *Population estimates*. Available from URL: http://www.stat.go.jp/english/data/jinsui/index.htm

Nagare H. (2000) Formation Mechanisms of Water Quality in Lake Biwa (Doctoral Thesis of Kyoto University).

Nagare H., Somiya I. and Fujii S. (2001) Phosphorus mass-quantity change in Lake Biwa. *Water Science and Technology: Water Supply*, 1(2), 49-56.

Okuda, S. and Kumagai, M. (1995) *PHYSICAL PROCESSES IN A LARGE LAKE: LAKE BIWA, JAPAN*, American Geophysical Union.

Saktaywin W., Tsuno H., Nagare H., Soyama T., Weerapakkaroon J. (2005) Advanced sewage treatment process with excess sludge reduction and phosphorus recovery. *Water Research*, 39, 902–910.

Shiga Prefecture (2003a) *Statistics on crop production*.

Shiga Prefecture (2003b) *Basic survey on livestock industry*.

Shiga Prefecture (2005a) *Population and number of households*. Available from URL: http://www.pref.shiga.jp/data/population/renew/. Accessed 26 August 2005.

Shiga Prefecture (2005b) *White paper on the Environment*.

The Ministry of Agriculture, Forestry and Fisheries of Japan (1996) *Statistics of agricultural production*. Available from URL: http://www.maff.go.jp/toukei/sokuhou/data/12-15tiiki/.

The Ministry of Agriculture, Forestry and Fisheries of Japan (MAFF) (2003) *Table on Demand and Supply of Food*. Available from URL: http://www.kanbou.maff.go.jp/www/fbs/fbs-top.htm

Features of the structurally - functional organization and ecosystems evolution of unique brackish Karstic Lakes in Russia

Nafisa Mingazova[1], Nikolay Aladin[2], Marina Ivanova[2], Sergey Golubkov[2], Olga Derevanskaya[1], Olga Palagushkina[1], Elena Unkovskaya[1], Igor Plotnikov[2], David Piriulin[2], Lubov Dgakova[2], Aleksey Smurov[2], Lubov Pavlova[1], Fania Barieva[1], Marat Monasipov[1], Larisa Nazarova[1]

[1]Kazan State University, 18, Kremlevskaya street, Kazan, RUSSIA
[2]Zoological Institute Russian Academy Sciences, 1, Naberegnaya street, S-Peterburg, RUSSIA
Email: Nafisa.Mingasova@ksu.ru

Abstract

The karstic lakes concern to azonal lakes of the world, the hydrochemical regime depends on a underground water feed. Sulphatic, breakish (salinity about 2,5 %o and more), coldwater karstic lakes with a pressure head feed concern to the extremely rare for Russia to type of unique "azure" lakes. At early stages of development they differ characteristic azure color and a high transparency of waters, have balneological medical value. Researches are carried out to 1995-2002 by the example of 10 karstic "azure" lakes of the Middle Volga region. Features of the structurally - functional organization and development of these lakes at different stages, dependence of development communities from succession of lakes are revealed. A biodiversity of this lakes type is submitted by 136 species and forms of phytoplankton, 39 – macrophytes, 28 - infusorians, 81 - zooplankton (*Rotatoria* - 31, *Crustacea* - 50), 10 - *Ostracoda*, 207 - zoobenthos and nektobenthos (4 - *Turbellaria*, 24 - *Oligochaeta*, 3 - *Hirudinea*, 3 - *Crustacea*, 3 - *Arachnida*, 137 - *Insecta*, 29 - *Gastropoda*, 5 - *Bivalvia*), 10 species of fishes. For these lakes are characteristic prevalence *Crustacea* in zooplankton, specific complexes of species in zoobenthos and macrophytes, underwater vegetation. In receipt and transformation of organic substance at early stages the large role plays macrophytobenthos, planktonic communities are badly advanced, all stream of energy practically goes through benthic systems. At early stages development of communities basically is influenced with such factors as temperature, light exposure, speed of water exchange and a mineralization, on a measure of ageing grows a role of a gas regime, salinity and trophy. At last stages of development "azure" lakes become hydrosulphuric and hypertophic (heterotrophic succession).

Key words: biodiversity, karstic lakes, lake evolution.

Introduction

The territory of Russian Federation was always rich in lakes and lake-like water bodies of various types. And Middle Volga region is not an exception in this case. According to geomorphological features of constitution and formation, the intensive karst processes are characteristic of this region. For this reason, karstic lakes of the various forms, volumes, depths, type of a water balance, temperature, hydrochemical and hydrophysical regimes, and other parameters are frequent here. Besides the usual karstic lakes, the Middle Volga region has more infrequent (in hydrological, hydrochemical and balneological aspects) type of brackish water lakes formed due to pressure of water ascending waters in places of karstic potholes.

During researches of 1995-2001 Laboratory of Water Ecosystems in the Ecological Faculty of Kazan State University reviewed and collected data on 9 brackish water karstic lakes, such as Green Stream, Small Blue Lake-1, Small Blue Lake-2, Big Blue Lake, Yugidem, Blue Oxbow, Shungaldan, Karakaer and Salt Lake, lying in Tatarstan and Mari El republics. Investigated brackish water lakes were characterized by the following distinctive features of uniqueness: karstic genesis of bolson; presence of deep potholes (“abysses”) with a powerful output of underground water pressure; specific water balance; amplified water exchange; sulphate type of waters with high mineralization (more than 2 g/l); high transparency and azure (ultramarine) colour of water.

The investigated lakes settle down in basins of the small rivers of Middle Volga region, such as Kazanka (Tatarstan Republic), Ilet and Big Kokshaga (Mari El Republic), which are the left inflows of Volga river.

Materials and methods

The study was led according to the standard methods. The fistudy sites were 2 m deep and coastal zone width from water’s edge to 1,5-2 m isobath. The hydrochemical samples from water bodies were collected at the surface and bottom horizons (with use of the bathometer). The hydrochemical analysis followed the standard methods of water quality ssesment.

Results

All investigated brackish water lakes were set in a series (Mingazova, 2001), describing different stages of development and seral changes. In general, view of the seral stages are represented in Figure 1.

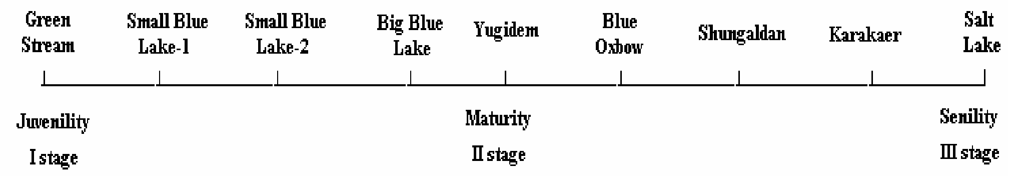

Figure 1. Seral stages of development of investigated brackish water lakes in Middle Volga Region (Mingazova, 2001).

The group of lakes of the first part of the series (from a Green Stream up to Blue Oxbow) characterizes development of lakes of this type from a young stage up to a stage of maturity and senility (autogenic, autotrophic succession). The water bodies have sulphate-calcic type of water, with the mineralization of about 2,2 - 2,5 g/l, and the lakes are not stratified in salinity. These are oligohaline lakes with characteristic “azure” colour of water with different shades. These water bodies characterize type of “azure” lakes, which show stages of youth and maturity. The group of lakes of the second part of the series characterizes development of lakes of the “azure” type at stages of fading and decrepitude (Shungaldan, Karakaer and Salt Lake). Here there is a retardation of a drain and sliming of bottoms, and accumulation of salts. Abyssal lakes, such as Shungaldan and Salt Lake are sharply stratified lakes. The shallow lake Karakaer desalinates along a shunt of drainage waters from marshes. At these stages we have an increase in salinity, up to polyhalinity at bottom layer (for Salt Lake - till 7-27 g/l), with change of ionic structure of water to sulphate-soda-potassium type of water. These are low-drain, hypertrophic lakes with a low water transparency (Mingazova, 2001).

Any habitat has characteristic features (elements of environment). The living organisms tend to adapt to ther conditions, both to usual and the extreme in process of interactions with the environment. A hydrological regime, temperature, the optical features, salinity and mineral composition of water are considered as major abiotic factors of the environment, which influence qualitative structure, quantitative characteristics and distribution of biotic components of water ecosystem.

In Figures 2 to 4 the distribution of values of some abiotic characteristics of lake environment of seral stages of brackish water lakes are shown. The difference in chemical structure of lakes, in our opinion, can be attributed to different stages of limnological development, and the opportunity of maintening of constancy of a hydrochemical regime. This is connected to the amplified underground water feeding as well as advanced inflow feeding and outflow of waters from lake potholes.

Considering water exchange coefficient in the series, it is possible to notice, that its greatest values are observed in first half of the series (beginning of group of “azure" lakes), with the apparent tendency to decrease by the end of the series (figure 2).

The change of the contents of oxygen (figure 3) does not demonstrate the brightly expressed logic in the plan of limnogenesis of lakes, assaying significant fluctuations in limens of the series both on coastal and abyssal zonea of lakes. In general the contents of oxygen are characterized by rather low values (from 10 up to 150 % on saturation) on a surface. Lakes Blue Oxbow, Shungaldan, Karakaer and Salt Lake are characterized practically by anoxic conditions in bottom layers.

The concentration of hydrogen sulphide (figure 2) was characterized by high enough values, even in lakes of first half of the series, despite the observed high water exchange coefficient in them. Probably the cause of this situation is the constant unload of sulphate waters with subsequent desulphatization of the lakes. Salt Lake, is still recovering from the consequences of heavy organic pollution in 1970. It is characterized by biggest concentration of hydrogen sulphide at the bottom layer (134,5 mg/l). In general, the lakes at the end of the series (Karakaer, Salt Lake), hydrogen sulphide is formed by biogenic ways owing to bacterial decomposition processes of large amounts of anthropogenic material.

The gas regime of brackish water karstic lakes is characterized by an insignificant saturation by oxygen (not exceeding 100 %) in surface layers of water of lakes. There is deficiency of oxygen at the bottom (especially at lakes of the end of series) and high amounts of hydrogen sulphide in bottom layers of the water.

It is possible to note about parameters of conductivity and general mineralization for a surface layer of brackish water lakes that they are increase up to the maximal values for lakes Small Blue Lake-2 (on mineralization) and Big Blue Lake (on conductivity) with the subsequent recession and presence of two lower peaks on lakes Blue Oxbow (bilateral peak) and Salt Lake (secund). The irregular character of diagram may be explained by an essential difference in volumes of a water bodies. Despite such irregularity of distribution of values on thelake series, nevertheless is looked through particulate isolation of group of “azure" lakes in relation to other lakes.

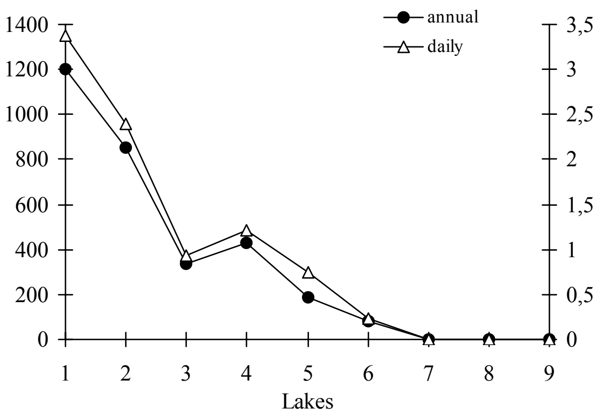

Figure 2. Water exchange coefficient in brackishwater karstic lakes: Green Stream (1), Small Blue Lake-1 (2), Small Blue Lake-2 (3), Big Blue Lake (4), Yugidem (5), Blue Oxbow (6), Shungaldan (7), Karakaer (8) and Salt Lake (9).

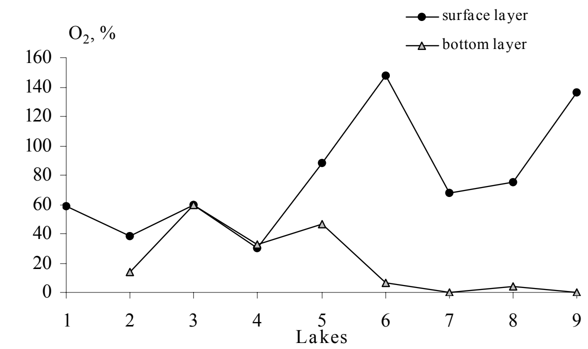

Figure 3. The contents of oxygen in brackish water lakes: Green Stream (1), Small Blue Lake-1 (2), Small Blue Lake-2 (3), Big Blue Lake (4), Yugidem (5), Blue Oxbow (6), Shungaldan (7), Karakaer (8) and Salt Lake (9).

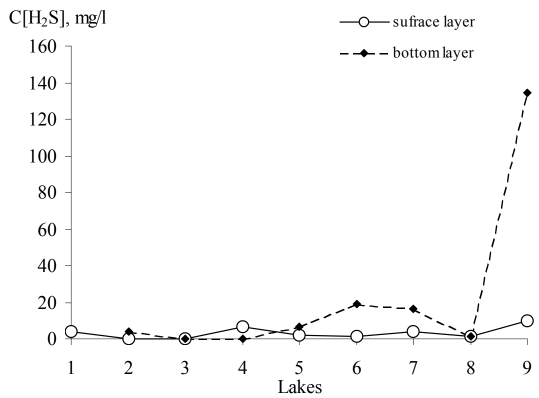

Figure 4. The contents of hydrogen sulphide in brackish water lakes: Green Stream (1), Small Blue Lake-1 (2), Small Blue Lake-2 (3), Big Blue Lake (4), Yugidem (5), Blue Oxbow (6), Shungaldan (7), Karakaer (8) and Salt Lake (9).

The group of “azure" lakes (from Small Blue Lake-1 up to Blue Oxbow) is much more sharply distinguished as a whole because they are characterized by very close values and mineralization at the bottom. The above pattern of karstic lakes reflects the distribution of values of the sulphate ion contents.

Discussion

During consideration of such parameter as the biological consumption of oxygen, showing the contents of organic matter, it is possible to showthat general both for litoral, and for abyssal zone is the close contents in lakes of “azure” group and sharp increase, sometimes up to the highest values, on such lakes as Shungaldan, Salt Lake is comparable.

The change in the contents of organic matter in limens of seral stages of lakes result in change of their trophic status (table 1). There is increase of the trophic level by the end of series owing to corresponding decrease in other sectors.

Table 1. Trophic status of investigated brackish water lakes on the parameters of productivity.

Reservoir	**Trophic level**
Green Stream	oligotrophic
Small Blue Lake-1	oligotrophic
Small Blue Lake-2	oligotrophic
Big Blue Lake	oligotrophic
Yugidem	oligo-mezotrophic
Blue Oxbow	oligo-mezotrophic
Shungaldan	eutrophic
Karakaer	mezo-eutrophic
Salt Lake	eutrophic-hypertrophic

Abiotic factors show but to a lesser degree similar f limnogenetic development in seral lakes. It is possible to assume, that none of the considered abiotic factors in itself does not define limnogenesis of all considered lakes.

Biotic factors show an interesting pattern tof changes with limnetic development of karstic.

The minimum quantity of species is characteristic for lakes which are settling down on edges of seral stages. That is formation taking place in their initial stage as a Green Stream. To middle of series, where trophic level grows up to mezotrophy (Big Blue Lake and Yugidem) the maximal number of species of the zoobenthos was observed. The number, biomass and indexes of biodiversity are greatest. Further, by the end of the series, deterioration of conditions and increase of trophic level (down to hypertrophic status) biodiversity is reduced remain euribiontic species, which maintain significant pollution (figure 5).

Characteristic also is the close association between shallow and profundal zones of biodiversity attained by the end of the series, such differentiation on indexes of diversity between these limnetic zones as a whole especially at lake Shungaldan (figure 6).

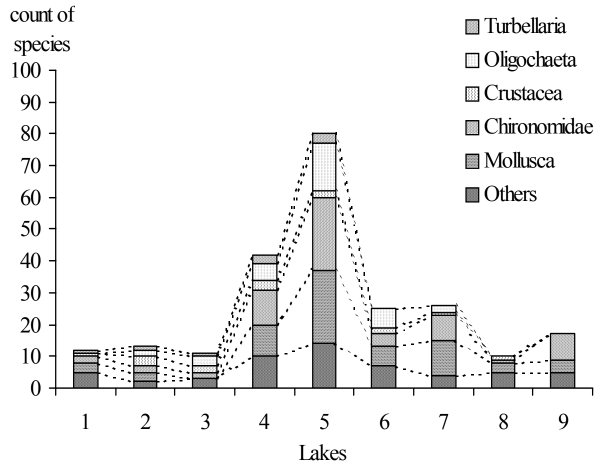

Figure 5. Count of species of major groups of benthic invertebrates found in brackish water lakes: Green Stream (1), Small Blue Lake-1 (2), Small Blue Lake-2 (3), Big Blue Lake (4), Yugidem (5), Blue Oxbow (6), Shungaldan (7), Karakaer (8) and Salt Lake (9).

The structural and functional characteristics of macrozoobenthos are closely linked to abiotic parameters of environment. This has been shown in bottom-dwelling invertebrates investigated in the series of brackish water lakes. (Golubkov *et al.,* 2001). Also, the analysis by comparison of series has shown that changes in macrobenthos reflect corresponding changes of such factors of environment as temperature, mineralization, contents of sulphates, and organic matter. These attributes unite “azure” lakes as a special group of lake with brackish water.

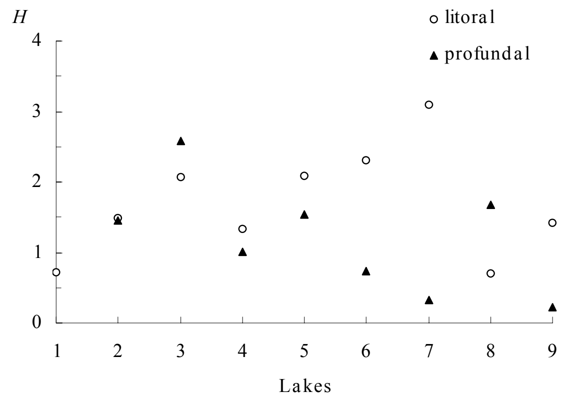

Figure 6. Shannon index (H) describing the diversity of zoobenthos of brackish water lakes: Green Stream (1), Small Blue Lake-1 (2), Small Blue Lake-2 (3), Big Blue Lake (4), Yugidem (5), Blue Oxbow (6), Shungaldan (7), Karakaer (8) and Salt Lake (9).

References

Golubkov S.M., Monasypov M.A., Yakovlev V.A., Mingazova N.M., Nazarova L.B., 2001. *Macrozoobenthos of brackish water karstic lakes of the Middle Volga region // Unique ecosystems of brackish water karstic lakes of the Middle Volga region*. Kaz. State Univ. press, - pp. 175-202.

Mingazova N.M., 2001. Stages of development of brackish water karstic lakes // Unique ecosystems of brackish water karstic lakes of the Middle Volga region. Kaz. State Univ. press, - pp. 87-94.

Variations of stable carbon isotopic compositions in surface waters of two Taiwan lakes

Tien-Nan Yang[1], Kuo-Yen Wei[2], Chung-Ho Wang[1], Teh-Quei Lee[1], Cheng-Wei Fan[3] and Yue-Gau Chen[2]

[1]: Institute of Earth Sciences, Academia Sinica, Taipei, 115, Taiwan, ROC
Email: tnyang@earth.sinica.edu.tw
[2]: Department of Geosciences, National Taiwan University, Taipei, 107, Taiwan, ROC
[3]: Research Center for Environmental Changes, Academia Sinica, Taipei, 115, Taiwan, ROC

Abstract

Carbon isotope ratios of total dissolved inorganic carbon (DIC) and particulate organic carbon (POC) in surface waters of two lakes of Taiwan sampled monthly during 2003-2004 were investigated. The two lakes are Mei Hwa Lake (MHL; altitude of 50 m) and Shuang Lian Pond (SLP; altitude of 470 m) in northeastern Taiwan. The C/N ratios of the POC in both lakes indicate that the POC was mainly the product of aquatic algal photosynthesis and show characteristic δ^{13}C values between -21.1.‰ and -28.9‰, with relatively negative values during the cold seasons. The time-series of POC δ^{13}C values for two lakes are parallel with each other and show a general decreasing trend through the two-year period. δ^{13}C values of the DIC in MHL and SLP also exhibit similar trends except during March – October of 2003, in which period the MHL δ^{13}C values tend to be enriched by 1- 3‰. Such heavier δ^{13}C values in DIC of MHL are likely to be caused by a local drought during the early part of the 2003.

Key words: Carbon and Oxygen Isotope, Lake, Taiwan.

Introduction

Several characteristic features of lakes, such as short residence time of element recycling, quick response of biota and high sedimentation rate, make them advantageous for monitoring climatic changes on local ecosystems. Particularly, δ^{13}C values of dissolved inorganic carbon (DIC), particulate organic carbon (POC), and sedimentary organics have been demonstrated very useful for tracing the movement and fate of carbon in lacustrine ecosystems (e.g., Quay et al., 1986; Meyers, 1997; Cole et al., 2002; Street-Perrott et al., 2004). Using such isotopic tracers, Scartazza et al. (2004) indicated that carbon isotope ratios of photosynthetic products and respiratory CO_2 in an ecosystem would also respond to seasonal climate change.

The δ^{13}C-DIC of lake water is mediated by many factors, e.g., geochemical (e.g., DIC, pH, alkalinity), morphometric (lake size and shape), biological (gross primary production, respiration), and others (see Bade et al., 2004 and reference therein for a review,). Using a model processed with data from 104 lakes, Bade et al. (2004) hypothesized that metabolism induced substantial variation in δ^{13}C-DIC around the potential isotope signature that was set by geochemical factors of the watershed. However, works of McKenzie (1985), Talbot (1990) and Leng & Anderson (2003) show the importance of exchange with atmospheric CO_2, which results in isotopically heavy aqueous bicarbonate ion equilibrated with atmospheric CO_2. This process occurs in close-basin lakes, with large surface areas, long residence times and enhanced evaporation (Leng & Anderson, 2003). How to differentiate the long-term trend in δ^{13}C-DIC from the seasonal variation and other specific factors of individual lake becomes crucial in gauging the principal effect of those mechanisms. Furthermore, the carbon isotopic fractionation between phytoplankton and water inorganic carbon would be influenced by temperature, lipid content of phytoplankton cells, kinetic fractionation and consistancy carbon pathway (see Descolas-Gros & Fontugne, 1990 and references therein for review).

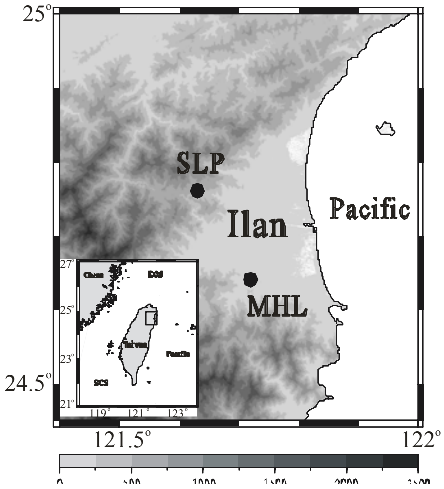

Figure 1. Topography map showing locations of the Mei Hwa Lake (MHL) and Shung Lian Pond (SLP) in the Ilan County, northeastern Taiwan. Inset: Taiwan situated between the East China Sea (ECS), Pacific and South China Sea (SCS). The small rectangle indicates the area of Ilan County.

Two adjacent lakes at different altitudes in Ilan County (Figure 1), northeastern Taiwan were chosen to investigate the δ^{13}C variations in organic and inorganic carbons for the period from 2003 to 2004. The Mei Hwa Lake (MHL) is located at 24.64°N, 121.72°E at elevation of 50 m, and has an area of 18.2 ha with a water depth ~ 1 m, whereas the Shuang Lian Pond (SLP), 4.6 ha in size and <2 m in water depth, is located nearby in a hilly basin at 27.76°N, 121.63°E at 470 m above sea level. In the northeast Taiwan, the reversal of summer southwest monsoon and winter northeast monsoon causes seasonal precipitation fluctuation (Hsu and Chen, 2002). We expect that the δ^{13}C signals from these two lakes would reflect seasonal fluctuation but with different values reflecting differences in size, altitude and water properties.

Method and materials

Surface waters were sampled and filtrated monthly from Feb. 2003 to Dec. 2004. Water samples of rainfall were also collected at both lakes on 21 Feb. 2005 for comparison. We followed sample collection and preparation procedures for DIC, POC and oxygen is described in Peng (1995), Kao and Liu (2000) and Peng et al. (2002), respectively. Briefly, an aliquot of the water was filtered immediately after retrieval using a Whatman Cellulose membrane (5 μm in pore size, 47 mm in diameter). Then, barium hydroxide was put into water to precipitate enough amount of dissolved inorganic carbon (Σ DIC). The purified $BaCO_3$ was then treated with orthophosphoric acid to release CO_2 under 50^{o}C in laboratory. Another aliquot of water was filtered as quickly as possible using a precombusted (450 ^{o}C) Whatman glass fiber filter (GF/F, 0.7 μm in pore size, 47 mm in diameter). Membrane with suspending particulate organic matters (POM) were then dried and delivered for isotopic measurements. For oxygen measurement, water was preserved by addition of mercury chloride, and was equilibrated with CO_2 in a vacuum glass in thermostat at 30^{o}C for two hours in laboratory. The CO_2 was subsequently extracted for oxygen isotopic analysis.

Carbon isotopic ratio of DIC and oxygen isotope composition of lake waters were measured by a VG SIRA-10 triple collector mass spectrometer at the Institute of Earth Sciences, Academia Sinica, whereas carbon isotopic ratio of particulate organic carbon (POC) by a Finnegan Delta Plus mass spectrometer at the Department of Geosciences, National Taiwan University. The results are expressed in δ-notation with per mil relative to VPDB (Coplen, 1996). The precision of the measurements was ±0.06‰ for DIC, ±0.1‰ for POC, and ±0.1‰ for oxygen.

Results and discussion

Time-series of δ^{13}C of DIC in both lakes show seasonal variation with values ranging from -11.80‰ to -15.46‰ in the MHL and from -13.94‰ to -16.30‰ in SLP (Figure 2). The δ^{13}C of DIC in rain-water in winter season of 2004/2005 at the MHL and SLP was -16.27‰ and -16.55‰ (Figure 2), respectively, while that in the air source was approximately -8‰ (Langenfelds et al., 2002). Both the mean values of δ^{13}C of DIC (-14.14‰ in MHL and -15.16‰ in SLP) are close to the rain-water sources indicating effect of rainfall was stronger than that of exchange with air through time. The fractionation toward heavier lake DIC value is more evident for the period of March - October, 2003 when a local drought occurred at the MHL. During this period, the mean monthly precipitation amount dropped to relatively low levels while the δ^{18}O of lake waters became heavier (Figure 3a). The δ^{13}C of DIC became also heavier in corresponding to the drought (Figure 3b). However, it appears that δ^{13}C-DIC decreases with altitudes as indicated by the more negative mean value of -15.16‰ in SLP (alt. 470 m) compared to -14.14‰ in MHL (alt. 50 m) (Figure 2). It is more likely that the larger lake size, stronger evaporation and exchange with air in the MHL all have interplayed with one another to make the DIC values more positive. Moreover, the δ^{13}C of DIC in the SLP tends to show lighter values than that in the MHL during the warm seasons in 2003 and 2004. We consider this as a result of a much enhanced primary production in the SLP during the intervals (Figure 2).

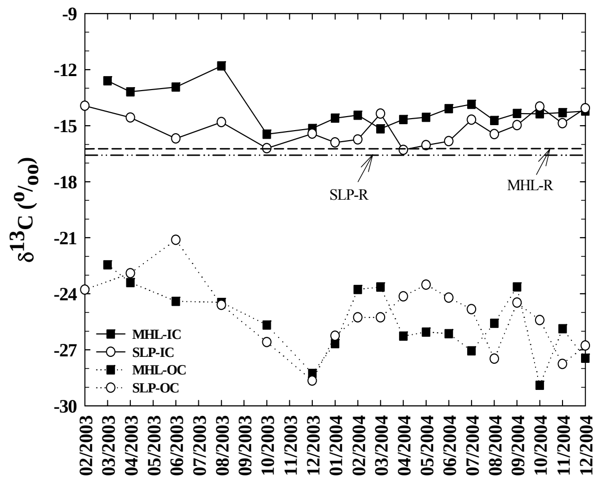

Figure 2. Monthly variation of δ^{13}C values of dissolved inorganic carbon (IC, solid lines) and particulate organic carbon (OC, dotted lines) in lakes MHL and SLP between Feb. 2003 and Dec. 2004. The data of MHL are marked by filled squares, whereas those of SLP by opened circles. The δ^{13}C value of DIC in rain of Dec. 2004 is indicated with dash and dash-dot-dot line for the MHL and SLP, respectively.

The isotopic ratios of the particulate organic carbon (POC) in the two lakes were similar but fluctuated seasonally (Figure 2). The δ^{13}C-POC values during warm seasons tend to be heavier than those in the cold seasons. The trend is apparently corresponding to thermal effect of CO_2 dissolubility and its isotopic composition in both lakes during the monitoring period. Dissolution of CO_2 in surface water increases when air temperature drops. δ^{13}C of CO_2 is less negative, while bicarbonate usually has an isotopic fractionation value around 0‰ (Desco;as-Gros & Fontugne, 1990). So, δ^{13}C-POC values reflect the increase of dissolved CO_2 during cold time.

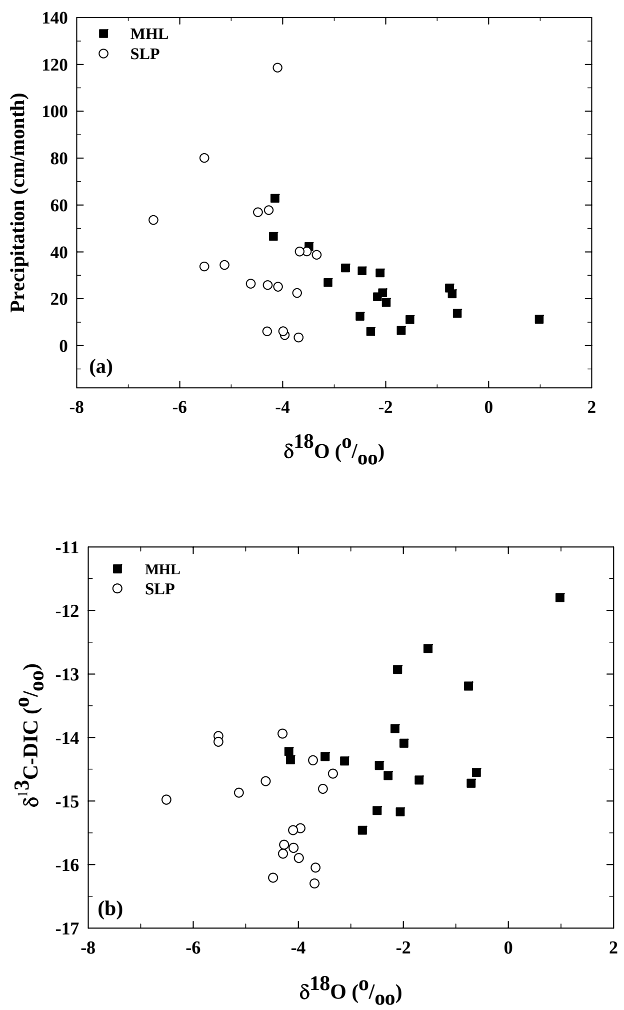

Figure 3. Plots of δ^{18}O against (a) monthly mean precipitation amount and (b) δ^{13}C-DIC in MHL and SLP during years 2003 and 2004. Solid squares represent the MHL whereas open circles the SLP values. The monthly mean precipitation at MHL and SLP are from Suao Station, Central Weather Bureau, and from the Fu Shan Experimental Forest TERN (Taiwan Ecological Research Network), respectively.

This temperature-dependent phenomenon has also been verified by Sackett et al. (1965), Fontugne & Duplessy (1981) and Rau et al. (1982), in showing that the phytoplankton δ^{13}C values in sea surface waters become more negative when sea surface temperatures were lower. However, it is suggested that the less discrimination in δ^{13}C of POC in the SLP than in MHL is a result of enhanced primary production during the warm intervals in years 2003 and 2004. In a review work, Meyers (1997) noted that smaller lakes are generally more productive than larger ones. The enhanced primary production simultaneously impacted a much depleted in ^{13}C of the DIC pool in the SLP than that in MHL during the warm seasons. The ratios of organic carbon over nitrogen (OC/N) in particulate organic matter are always smaller than 10 (Figure 4), indicating an algae origin (Meyers, 1994). The lack of correlation between δ^{13}C-POC and OC/N (Figure 4) suggests that the photosynthetic products are composed of lipids, proteins and sugars (Degens et al., 1968), and the composition of primary producers have varied through the monitoring period.

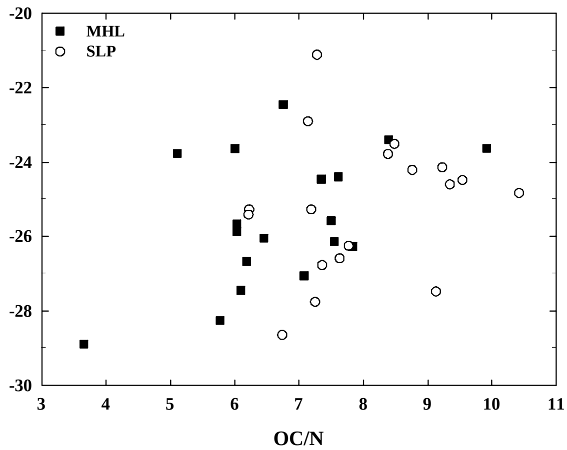

Figure 4. Plot of OC/N ratios against δ^{13}C-POC of lake surface waters in MHL and SLP during years 2003 and 2004. The MHL data are marked by solid squares and SLP by open circles.

Conclusions

This study shows that the δ^{13}C values of DIC were generally heavier in the plain lake MHL compared to the other lake SLP up on the hill. We interpret that larger lake size, higher evaporation and less photosynthesis activity in MHL are the principal controlling factors for the difference. A more enhanced primary production revealed by δ^{13}C-POC in the SLP than the MHL during both warm seasons in year 2003 and 2004 is suggested. The δ^{13}C-POC lighter trends in both studied lakes from warm to cold time reflect the strong thermal effect on amount of CO_2 disolved during winter season.

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References

Bade D.L., Carpenter S.R., Cole J.J., Hanson P.C. & Hesslein R.H., 2004. Controls of δ^{13}C-DIC in lakes: Geochemistry, lake metabolism, and morphology. *Limnol. Oceanogr.* 49, 1160-1172.

Cole J.J., Carpenter S.R., Kitchell J.F. & Pace M.L., 2002. Pathways of organic carbon utilization in small lakes: Results from a whole-lake ^{13}C addition and couple model. *Limnol. Oceanogr.* 47, 1664-1675.

Coplen T.B., 1996. New guidelines for reporting stable hydrogen, carbon, and oxygen isotope-ratio data. *Geochim. Cosmochim. Acta* 60, 3359-3360.

Degens E.T, Behrendt M., Gotthardt B. & Reppmann E., 1968. Metabolic fractionation of carbon isotopes in marine plankton, part II. *Deep-Sea Res.* 15, 11-20.

Descolas-Gros C. & Fontugne M., 1990. Stable carbon isotope fractionation by marine phytoplankton during photosynthesis. *Plant, Cell Environ*. 13, 207-218.

Fontugne M.R.& Duplessy J.C., 1981. Organic carbon isotopic fractionation by marine plankton n the temperature range — 1 to 31^{o}C. *Oceanol. Acta* 4, 85-90.

Hsu H.H. & Chen C.T., 2002. Observed and projected climate change in Taiwan. Meteor. Atmos. *Phys*. 79, 87-104.

Kao S.J. & Liu K.K., 2000. Stable carbon and nitrogen isotope systematics in a human-disturbed watershed (Lanyang-His) in Taiwan and the estimation of biogenic particulate organic carbon and nitrogen fluxes. *Global Biogeochem. Cycles* 14, 189-198.

Langenfelds R.L., Francey R.J., Pak B.C. et al., 2002. Interannual growth rate variations of atmospheric CO_2 and its δ^{13}C, H_2, CH_4, and CO between 1992 and 1999 linked to biomass burning. *Glob. Biogeochem. Cycles* 16, 1048, doi: 10.1029/2001GB001466.

Leng M.J. & Anderson N.J., 2003. Isotopic variation in modern lake waters from western Greenland. *The Holocene* 13, 605-611.

McKenzie J.A., 1985. Carbon isotopes and productivity in the lacustrine and marine environment. *In.* W. Stumm, ed.*chemical Processes in lakes* pp. 99-118. Wiley, New York.

Meyers P.A., 1994. Preservation of elemental and isotopic source identification of sedimentary organic matter. *Chem. Geol.* 144, 289-302.

Meyers P.A., 1997. Organic geochemical proxies of paleoceanographic, paleolimnologic, and paleoclimatic processes. *Org. Geochem*. 27, 213-250.

Peng T.-R., 1995. *Environmental Isotopic Study (δ^{13}C, δ^{18}O, ^{14}C, T. on Meteoric Water and Groundwaters in I-Lan Area*. Ph.D. thesis, National Taiwan University. (In Chinese, with English Abstract)

Peng T.-R., Wang C.-H. & Liu T.-S., 2002. Hydrogen and oxygen isotope variation in Ilan precipitation. *Taiwanese J. Agricul. Chem. And Food Sci.* 40, 336-346. (In Chinese, with English Abstract)

Quay P.D., Emerson S.R., Quay B.M. & Devol A.H., 1986. The carbon cycle for Lake Washington — A stable isotope study. *Limnol. Oceanogr*. 31, 596-611.

Rau G.H., Sweeney R.E. & Kaplan I.R., 1982. Plankton C^{13}/C^{12} ratio changes with latitude: differences between northern and southern ocean. *Deep Sea Res*. 29, 1035-1039.

Sackett V.M., Ekelmann W.R., Bender M.L. & Bé, A.W.H., 1965. Temperature dependence of carbon isotope composition in marine plankton and sediments. *Science* 148, 235-237.

Scartazza A., Mata C., Matteucci, G., Yakir D., Moscatello S. & Brugnoli E., 2004. Comparisons of δ^{13}C of phytosynthetic products and ecosystem respiratory CO_2 and their responses to seasonal climate variability. *Ecosys. Ecol.* 140, 340-351.

Street-Perrott F.A., Ficken K.J., Huang Y. & Eglinton G., 2004. Late Quaternary changes in carbon cycling on Mt. Kenya, East Africa: an overview of the δ^{13}C record in lacustrine organic matter. *Quat. Sci. Rev*. 23, 861-879.

Talbot M.R., 1990. A review of the palaeohydrological interpretation of carbon and oxygen isotopic ratios in primary lacustrine carbonates. *Chem. Geol.* 80, 261-279.

Bioidentification of xenobiotics in water as a part of pollution control

Valerii Tonkopii

Institute of Limnology, Russian Academy of Sciences, Sevastyanova str., 9, 196105 St.Petersburg, Russia
E-mail: tonkopi@hotmail.com

Abstract

We have been developing non-traditional methods of the identification of pollutants, using various hydrobionts as biological objects and the study of the mechanism of toxic action of xenobiotics. The experiments were carried out with using of *Daphnia magna.* This is a Crustacean in the order of Cladocera. These aquatic animals are extensively used in aquatic toxicology due to their small size, short life cycle and amenability to lab culture. *Daphnia magna* is the most sensitive test-object in relation of different pollutants among all known biological objects including experimental animals. Experiments were performed with a 2-days old culture of *Daphnia magna*. The toxicity of xenobiotics was determined by the value of LC_{50}, a concentration of the compounds causing death to 50% of hydrobionts during incubation with toxicants for 24 hours. In the first stage of the work, toxicity of organophosphates (Dipterex, DFP, DDVP, Paraoxon, Malathion, Malaoxon), carbamates (Aminostigmine, Physostigmine, Sevine), heavy metals (Hg, Pb, Cu, Co, Cd, Cr, As, Al), organochlorines (Aldrin, Dieldrin, Endrin, Aroclor, DDT, Lindane, PCBs etc.) and pyrethroids (Cypermethrin, Fenvalerate, Deltamethrin, Permethrin, Allethrin, Resmethrin, Phenothrin, Kadethrin, Cyphenothrin) was determined. The effects of a number of antagonists on the toxicity of xenobiotics were studied. At the first time we discovered that in experiments with *Daphnia magna,* some muscarinic cholinoreceptor blockers (atropine, glipine, pediphen etc.) reduced the toxic effect of organophosphates and carbamates. In the case of heavy metals the chelating agents (EDTA, Dithioethylcarbamate, Unithiolum, Sodium thiosulphuricum, L-Aspartic acid) were effective, for certain organochlorine poisonings - anticonvulsive drugs (diazepam, phenobarbital). In the case of pyrethroid's poisonings the antagonist of glutamate receptor (ketamine), DOPA receptors (haloperidole) and blocker of calcium channel (nimodipine) reduced the toxicity of xenobiotics. As far as these antidotes have a specific treatment action only against definite classes of pollutants, we have elaborated the sensitive express-methods of bioidentification of pollutants.

Key words: Xenobiotics, pollution, bioidentification, Dapnhia magna, pesticides, control.

Introduction

With a constant growth of the anthropogenic pressure on water bodies, the development and usage of bioindication methods supplementing physical and chemical methods of xenobiotic identification acquires especial significance. In view of the fact that chemical analysis require special equipment, they are expensive to perform and do not allow to evaluate the environmental toxicity, during the recent decade, large scale investigations have been performed to study various test-objects that are suitable for bioassay. At present biotesting plays an important role in the system of water quality control. On the currently used methods of bioassay provide only the integral evaluation of the pollutants effect but not the determination of the xenobiotics origin (Flerov, 1989).

We have been developing non traditional method for determination of different classes of pollutants using various hydrobionts as biological test-objects and our knowledge of the mechanism of toxic action of xenobiotics. Knowing the mechanisms of the specific toxic action of poisons, it is possible to use various pharmacological compounds to decrease or increase the effects of toxicants. This approach allows us to use biological objects to identify certain xenobiotic, poisoning which can be prevented by means of antagonists. All above mentioned methods are widely used when employing experimental animals (mice, rats) as test-objects, but it has not been developed at all for alternative biological objects, particularly for hydrobionts. The elaboration of a new method of bioidentification was founded on the study of Cholin-, GABA-, Dopamin- and Glutamate- ergic system of *Daphnia magna* and usage of pharmacological antagonists of xenobiotics. Such new pharmacological approach with usage of *Daphnia magna* as bioobject have made possible to perform the general identification of different classes of the most toxic substances for aquatic ecosystem health. These organophosphates, carbamates, heavy metals, organochlorines, pyrethroids without usage of chemical analysis.

Materials and methods

As background test materials, we chose a wide range of anticholinesterase (antiChe) compounds, different heavy metals, organochlorine pesticides and pyrethroids. Many compounds of these classes are used as pesticides, drugs and chemical warfare agents. Currently, dozens of pesticides capable of polluting the aqueous media through the runoff from agricultural lands or as a result of chemical industry accidents are produced. The experiments were carried out sing *Daphnia magna. Daphnia magna* is a Crustacean in the order of Cladocera. This aquatic animal extensively used as a test organism in aquatic toxicology due to their small size, short life cycle and amenability to lab culture. *Daphnia magna* is the most sensitive test-object in relation of different pollutants (organophosphates, heavy metals, organochlorines, pyrethroids etc.) among all known biological objects including experimental animals (Peters and De Bernardi, 1987).

Experiments were performed with a 2-days old culture of *Daphnia magna*. During the experiments, hydrobionts were placed in beakers with 25 ml of dechlorinated settled tap water at 18-20°C. The toxicity of xenobiotics was determined by the value of LC_{50}, a concentration of the compounds causing death to 50% of hydrobionts during incubation with toxicants for 24 hours. In the first stage of the work, toxicity of organophosphates and carbamates (Dipterex, DFP, DDVP, Paraoxon, Malathion, Malaoxon, Aminostigmine, Physostigmine, Sevine), heavy metals (Hg, Pb, Cu, Co, Cd, Cr, As, Al), organochlorines (Aldrin, Dieldrin, Endrin, Aroclor, DDT, Lindane, PCBs etc.) and pyrethroids (Cypermethrin, Fenvalerate, Deltamethrin, Permethrin, Allethrin, Resmethrin, Phenothrin, Kadethrin, Cyphenothrin) was determined. The effects of a number of poison reducers on the toxicity of xenobiotics were studied. Xenobiotics and their antagonists were added to the incubation mixture simultaneously.The results of the protection experiments are expressed as the protective coefficient (PC) – the ratio of LC_{50} value in treated and in untreated *daphnids*.

Results

On the base of study of mechanism of xenobiotics action to *Daphnia magna* and the usage of pharmacological antagonists of poisonings the new methods of bioidentification of different pollutants were elaborated. At the first time we discovered that in experiments to *Daphnia magna* some muscarinic cholinoreceptor blockers (atropine, glipine, pediphen etc.) reduced the toxic effect of organophosphates and carbamates (Table 1).

Table 1:The influence of cholinolytics on toxicity of DDVP and aminostimine in experiments to *Daphnia magna.*

Drugs mg/l	LC_{50} DDVP, mg/l	PC	LC_{50} aminostigmine, mg/l	PC
Control	0.00021±0.00005	-	0.012±0.002	-
Atropine				
1.0	0.00052±0.00004	2.5	0.018±0.002	1.5
2.0	0.00063±0.00007	3.0	0.042±0.009	3.5
6.0	0.00073±0.00006	3.5	0.042±0.009	3.5
Glipine				
1.0	0.00042±0.00008	2.0	0.021±0.005	1.75
2.0	0.00074±0.0001	3.5	0.069±0.019	5.75
Pediphen				
1.0	0.00032±0.00008	1.5	0.024±0.002	2.0
2.0	0.00063±0.00008	3.0	0.036±0.004	3.0

In the case of heavy metals the chelating agents (EDTA, Dithioethylcarbamate, Unithiolum, Sodium thiosulphuricum, L-Aspartic acid) were effective (Table 2), for certain organochlorine poisons - anticonvulsive drugs phenazepam, phenobarbital (Table 3).

Table 2 .The influence of GABA-mymetics on toxicity of DDT and lindane in experiments to *Daphnia magna*.

Compounds mg/l	LC_{50} mg/l DDT	PC	LC_{50}mg/l Lindane	PC
Control	0.08+0.02	-	0.12+0.02	-
Ethyl alcohol (g/l)				
0.1	0.15+0.03	1.9	0.23+0.07	1.9
0.025	0.24+0.03	3.0	0.26+0.07	2.2
Phenobarbital				
2.0	0.21+0.03	2.6	0.24+0.06	2.0
1.0	0.23+0.07	2.9	0.32+0.09	2.7
Phenazepam				
0.1	0.32+0.09	4.0	0.28+0.09	2.3
0.05	0.25+0.07	3.1	0.23+0.08	1.9

Table 3. The influence of EDTA and Unithiolum on toxicity of $Pb(NO_3)_2$ and $HgCl_2$ to *Daphnia magna* in experimental conditions*.*

Chelates (mg/l)	$HgCl_2$ LC_{50}(mg/l)	PC	$Pb(NO_3)_2$ LC_{50}(mg/l)	PC
Control	0.16±0.06	-	1.70±0.52	-
EDTA				
2.5	1.59±0.47	9.97	3.38±0.56	1.98
5.0	2.1±0.05	13.1	5.1±0.56	3.0
10.0	3.2±0.06	20.0	6.6±1.48	3.88
25.0	-	-	13.3±2.9	7.8
Unithiolum				
25.0	0.46±0.11	2.87	3.38±0.56	1.98
50.0	1.38±0.38	8.60	7.67±1.79	4.5
100.0	1.69±0.28	10.6	13.45±2.24	7.9

In the case of pyrethroid's poisonings the antagonists of glutamate (ketamine), DOPA (haloperidole) receptors and blockers of calcium channel (nimodipine) reduced the toxicity of xenobiotics (Table 4 and 5).

Table 4. The influence of ketamine and nimodipine on the toxicity of pyrethroids in experiments to *Daphnia magna* in experimental conditions.

Pyrethroids (mg/l)	Cypermethrin LC_{50} (mg/l)	PC	Phenothrin LC_{50} (mg/l)	PC
Control	0.057±0.003	-	0.05±0.12	-
Ketamine				
7.5	0.62±0.17	10.9	0.52±0.12	10.4
4.0	0.51±0.13	8.9	0.46±0.10	9.2
Nimodipine				
25.0	0.29±0.03	5.1	0.42±0.03	8.4
12.5	0.16±0/02	2.8	0.21±0.09	4.2

Table 5. The influence of haloperidole on the toxicity of pyrethroids to *Daphnia magna* in experimental conditions.

Pyrethroids LC_{50} (mg/l)	**Cypermethrin LC50** (mg/l)	PC	Phenothrin LC_{50} (mg/l)	PC	Permethrin LC_{50} (mg/l)	PC
Control	0.057	-	0.05	-	0.05	-
Haloperidole						
3.0	0.46	8.1	0.6	12.0	0.36	7.2
1.5	0.48	8.4	0.55	11.6	0.50	10.0

As far as these antidotes have a specific treatment action only against definite classes of pollutants, we have elaborated the sensitive express-methods of bioidentification of pollutants. Such new pharmacological approach with use of hydrobionts as test-objects have made possible to perform the general identification of different classes of xenobiotics in fresh water.

Conclusion

We have been developing non-traditional express-method of the identification of pollutants (organophosphates,carbamates,organochlorines, heavy metals and pyrethroids) using Daphnia magna as experimented in animals. The study shows the mechanism of toxic action onxenobiotics. The new method is proposed for water pollution control is efficient and has potential for testing wide use in water quality assesment..

References

Flerov, B., 1989. *"Ecological and physiological aspects of toxicology of aquatic animals"*. Nauka, Leningrad, 205 p.

Peters, R. and De Bernardi, R., 1987. *"Daphnia"*, Verbania, Pallanza, 399 p.

The abundance, biomass and composition of pelagic ciliates in East African lakes of different salinity and trophy

A. W. Yasindi[1,2] and W. D. Taylor[1]

[1]Department of Biology, University of Waterloo, Ontario N2L 3G1, Canada.
[2] Corresponding author, present address: Department of Zoology, Egerton University, P. O. Box 536, Njoro, Kenya
E-mail: awyasindi@yahoo.com, wdtaylor@sciborg.uwaterloo

Abstract

Planktonic ciliates were studied in 17 tropical East African lakes of different salinity and trophic status. Oligotrichs (e.g., *Strombidium*, *Strobilidium* and *Halteria*) and scuticociliates (e.g., *Cyclidium*, *Pleuronema*, *Cristigera*), dominated the ciliate communities. Conductivity and trophic status were the most important environmental variables influencing the distribution of ciliate species in East African lakes. Herbivorous oligotrichs were important in oligotrophic and mesotrophic lakes, as they are in temperate and subtropical lakes, but their importance decreased with increasing chlorophyll *a* concentration and conductivity. On the other hand, the importance of scuticociliates (primarily bacterivores) increased with increasing chlorophyll *a* and conductivity.

Mean ciliate abundance ranged from 2 to 1,220 ciliates$\cdot$mL^{-1} while the biomass range was from 1.9 to 1, 900 µg C$\cdot$L^{-1} respectively from oligotrophic to eutrophic lakes. Abundance and biomass had positive relationships with phytoplankton biomass. The ciliate abundance and biomass were higher than those reported in temperate (Quebec) and subtropical (Florida) lakes of similar trophic status. However, regression models predicting abundance and biomass of ciliates from chlorophyll suggest that temperate (Quebec), subtropical (Florida) and tropical (East Africa) lakes have similar ciliate abundance and biomass per unit chlorophyll except some saline tropical lakes which have very high abundance and biomass of ciliates relative to chlorophyll.

Key words: Ciliated protozoa, high abundance and biomass, tropical lakes.

Introduction

Ciliates comprise a significant component of the abundance and biomass of freshwater plankton communities. Their densities may be between 1-30 ciliates/ml in oligotrophic and mesotrophic lakes but may be exceed 1000/ml in eutrophic lakes (Yasindi et al. 2002). Ciliates feed on bacteria, pico- and nanophytoplankton but they are prey for zooplankton and fish larvae (Porter et al. 1979). As such, ciliates form a link in the process of energy flow from the smallest plankton groups to the higher trophic levels. Some studies have found a significant relationship between the abundance and biomass of planktonic ciliates and lake trophic state as measured by chlorophyll *a* concentration (Beaver and Crisman, 1982). Ciliate species diversity and richness are also influenced by lake productivity whereby the most productive lakes have the highest diversity and richness. The type of food items also influences the distribution of ciliate species. Large oligotrichs (>30 µm), which graze on nanophytoplankton, dominate oligotrophic lakes while small ciliates (<30 µm) that graze on bacteria dominate eutrophic lakes in temperate and subtropical lakes (Beaver and Crisman, 1982). However, most studies of planktonic ciliates have mainly been confined to the temperate lakes with relatively few studies in subtropical and tropical waters. Previous studies of ciliates in tropical freshwater lakes have been confined to single lakes (Finlay *et al*. 1987; Taylor & Zinabu 1989; Yasindi 1995). Therefore, our knowledge of planktonic ciliates of tropical freshwaters in general and East African lakes in particular is still very poor. The goal of this study was to investigate the abundance, biomass, composition and the ecological role of ciliates in the food webs of tropical East African lakes of different salinity and trophic status and compare with previous studies intemperate lakes.

Materials and methods

Ciliate samples were taken from 17 lakes including Lakes Awassa, Abijata, Chamo, Langano and Zwai in Ethiopia and Lakes Baringo, Bogoria, Solai, Nakuru, Elmenteita, Naivasha, Oloidien, Sonachi, Simbi, Victoria, Malawi and Crescent lake in Kenya. 250-mL ciliate samples were collected in triplicate using a 4-L Van Dorn sampler at different depths, and immediately fixed in concentrated Bouin's fluid (5% v/v final concentration). In the laboratory, the water samples were concentrated to 50 ml by settling for 48 h. Subsamples of 0.5 to 5 mL (from eutrophic to oligotrophic lakes) were filtered onto gridded filters and stained by Quantitative Protargol Staining (QPS) technique of Montagnes & Lynn (1993). Ciliates were identified to genus or species level and enumerated. To determine ciliate biomass, ciliate dimensions were measured on a Summasketch III digitizing tablet and using microbiota software, which equates ciliates to standard geometric shapes such as cones, sphere and prolate spheroids (Roff & Hopcroft, 1986). The biovolume was then converted to biomass using a volume to carbon ratio of 0.14 pg C$\cdot$ µm^{-3} (Putt & Stoecker, 1989). Ciliates were counted either by scanning the whole stained filter when ciliate density was low or by counting ciliates in three randomly selected squares until 100 to 200 ciliates were counted. Species diversity was calculated based on Shannon diversity index using the MVSP (Kovac 1999). Ciliate size was expressed as equivalent spherical diameter (ESD) using the formula: ESD = 0.4775 Vol $^{1/3}$. Chlorophyll *a* concentration was measured and used as an indicator of trophic status. Bacteria were counted using the acridine-orange direct-count method of Hobbie *et al*. (1977). Physico-chemical factors such astemperature, pH, DO, alkalinity, conductivity and transparency were also measured.

We used the classification of lakes based on their ranges of conductivity by Talling & Talling (1965) to assign the lakes with conductivity < 600 $\mu S \cdot cm^{-1}$ to freshwater lakes, lakes with conductivity between 600 - 6000 $\mu S \cdot cm^{-1}$ as moderately saline lakes and saline lakes as lakes with conductivity > 6,000 $\mu S \cdot cm^{-1}$.

Bivariate regression and multiple regression (Statsoft, Inc. 1995) were used to assess relationships between ciliate abundance and biomass with environmental factors.

According to chlorophyll range group 1 (or oligotrophic) comprised Malawi (< 5 $mg.m^{-3}$), group 2 (mesotrophic) had Naivasha, Oloidien, Victoria, Solai, Nakuru, Elmenteita, Sonachi, Simbi, and Bogoria (5-50 $mg \cdot m^{-3}$), and Baringo, Abijata, Chamo, Awassa, Zwai, and Langano (> 50 $mg \cdot m^{-3}$) formed Group 3 (eutrophic). We used the classification of lakes based on conductivityby Talling and Talling (1965) to assign the lakes with conductivity range of 207 - 350 $\mu S \cdot cm^{-1}$ to freshwater lakes (< 600 $\mu S \cdot cm^{-1}$), lakes with 850 - 5,457 $\mu S \cdot cm^{-1}$ as moderately saline lakes (600 – 6,000 $\mu S \cdot cm^{-1}$) and lakes between 11,824.5 –16,837 $\mu S \cdot cm^{-1}$ as saline (> 6,000 $\mu S \cdot cm^{-1}$) lakes.

Table 1. Abundance, biomass and environmental variables measured. Abun = Abundance, Biom = biomass, Cond = conductivity, Temp = temperature, DO = dissolved oxygen, Alk = alkalinity, Chl *a* = Chlorophyll *a* and Bact = bacteria.

Lake	Abun	Biom	Cond	pH	Temp	DO	Alk	Chl *a*	Bact	Secchi	n
Baringo	5.6 ± 0.7	22 ± 19.6	912.0	8.2	25.2	9.3	233.3	43 ± 12	9.1	0.61	18
Bogoria	43.5 ± 8.2	220 ± 150	69792.0	9.4	26.9	12.9	14316.7	266 ± 8.9	66.7	0.13	6
Solai	21.7 ± 19	25 ± 22	5457.0	8.6	26.0	6.4	885.7	209 ± 11	42.9	0.07	6
Nakuru	32.4± 40.7	31± 68	16837.0	9.4	24.7	17.8	70.4	123 ±0.5	255.9	0.12	3
Elmenteita	640.3 ± 578	971 ± 763	15807.9	9.7	20.9	8.5	1973.6	123 ± 9.2	245.0	0.15	33
Sonachi	377.9 ± 70	1900 ± 1344	11824.5	9.5	21.3	8.8	1855.4	89 ± 6.9	40.5	0.16	63
Crescent	3.6 ±2.6	17 ± 11	294.4.0	7.7	20.0	3.9	24.6	32 ± 15	9.5	1.22	15
Naivasha	3.4 ± 0.4	17 ± 7	319.4	7.8	19.9	6.4	49.9	26 ± 1.9	8.2	1.07	75
Oloidien	6.9 ± 1.2	30 ± 12.6	2466.7	9.3	21.4	-	312.0	23.0	97.2	0.25	15
Simbi	2.9 ± 1.2	12 ± 8.5	21241.0	9.3	25.5	4.9	1860.1	82 ± 3	35.7	0.93	15
Victoria	12 ± 4.3	43 ± 13.2	207.0	6.4	26.2	6.7	-	31 ± 2.3	8.3	0.92	45
Malawi	2.2 ± 0.7	2.3 ± 0.7	256.1	-	24.9	7.3	-	1.5	2.9	20.00	66
Zwai	8.8 ± 0..2	42 ± 9	350 ± 5.8	8.5 ± 0.2	18 ± 0.1	-	4 ± 0.4	27.3	11.3	0.30	6
Langano	3.1 ± 0.9	22.7 ± 11	1580 ± 200	9.5 ± 0.1	24 ± 0.6	-	14 ± 0.4	13.4	7.0	0.23	12
Chamo	5.3 ± 44.7	11 ± 2.2	1320	8.9	-	-	12	44.2	16.7	0.65	3
Abijata	15.2 ± 13.8	60 ± 10.5	21500 ± 866	10 ± 0.1	24 ± 0.5	-	250 ± 32	11.6	43.8	0.65	3
Awassa	8.1 ± 1.9	8.9 ± 2.2	850 ± 288	8.3 ± 0.2	21 ± 0.4	-	9 ± 1.4	11.8	9.5	1.38	15

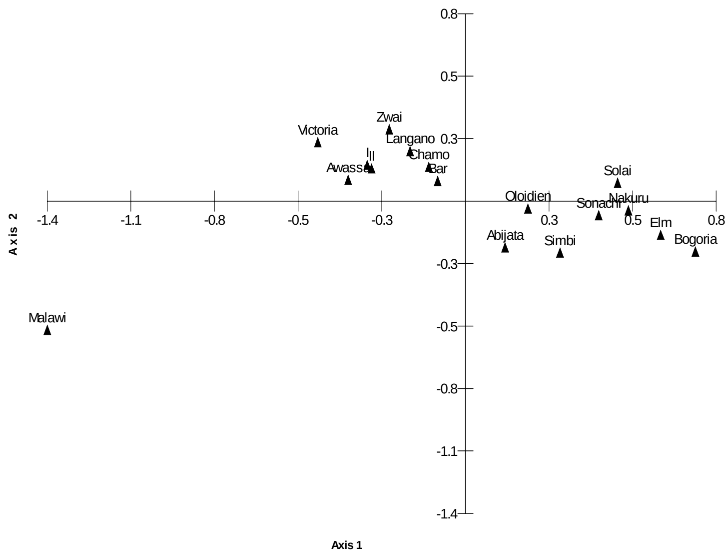

Figure 1. Principal components analysis diagram based on 6 environmental variables showing the distribution of the East African lakes on the first two principal components. The first axis is horizontal and the second is vertical. Bar = Baringo, Elm = Elmenteita, I = Crescent, II = Naivasha.

Table 2. The correlation matrix for ciliate biomass and environmental variables. Log_{10}-transformed means of all data for two years combined were used. Abbreviations as in Table 1. * = r is significant at 0.05 level of significance.

1	**2**	**Biomass**	Temp	pH	Cond	Alk	DO	Bact	Chl *a*	Secchi
Biomass	1.00		0.18	0.53*	0.59*	0.39	0.62*	0.74*	0.65*	-0.68*
Temp	0.18		1.00	-0.13	0.24	0.41	0.43	0.11	0.07	-0.07
PH	0.53*		-0.13	1.00	0.61*	0.56*	-0.06	0.62*	0.63*	-0.79*
Cond	0.59*		0.24	0.61*	1.00	0.79*	0.40	0.81*	0.64*	-0.59*
Alk	0.39		0.41	0.56*	0.79*	1.00	0.33	0.69*	0.63*	-0.59*
Do	0.62*		0.43	-0.06	0.40	0.33	1.00	0.43	0.56*	-0.26
Bact	0.74*		0.11	0.62*	0.81*	0.69*	0.43	1.00	0.75*	-0.71*
Chl *a*	0.65*		0.07	0.63*	0.64*	0.63*	0.56*	0.75*	1.00	-0.82*
Secchi	-0.68*		-0.07	-0.79*	-0.59*	-0.59*	-0.26	-0.71*	-0.82*	1.00

Results

Principal Component Analysis (PCA) of environmental variables divided the lakes into oligotrophic, mesotrophic and eutrophic lakes (Figure 1). 49 ciliate genera were identified from the 17 lakes. Small ciliates (< 30 µm ESD), mainly *Strobilidium*, *Strombidium*, and *Halteria* (Oligotrichida), and *Cyclidiun*, *Cristigera*, and *Pleuronema* (Scuticociliatida) dominated abundance. Other abundant ciliates belonged to Cyrtophorida, Peniculida, Heterotrichida, Prostomatida, Hypotrichida, and Haptorida. Shannon-Weaver species diversity had a negative and significant relationship with chlorophyll *a* concentration ($r = -0.53$, $P < 0.05$). The ciliate abundance and biomass, as well as the environmental variables measured are summarized in table 1 while table 2 shows the correlations of environmental variables with ciliate biomass and with each other. Mean ciliate abundance ranged from 1.8 ciliates $\cdot mL^{-1}$ in Lake Malawi to 1,220 ciliates $\cdot mL^{-1}$ in Lake Elmenteita during the two years of study. Mean ciliate biomass ranged from 1.9 µg $C \cdot L^{-1}$ in Lake Malawi to 1,900 µg $C \cdot L^{-1}$ in Lake Sonachi. Ciliate abundance and biomass were positively related to chlorophyll *a* concentration (Figure 2) and to conductivity (Figure 3). Ciliates in Oligotrichida and Scuticociliatida were the most abundant in East African lakes. However, the relative importance of Oligotrichida was greatest in Lake Malawi (< 5 $mg.m^{-3}$ chl.), where they contributed 93.1% of total ciliate abundance and decreased with increasing chlorophyll, contributing 5.2% in lakes with > 50 $mg.m^{-3}$ chlorophyll. The importance of scuticociliates increased with increasing chlorophyll accounting for 0.7% in Lake Malawi and 80.2% in lakes with > 50 $mg.m^{-3}$ chlorophyll (Table 3). The dominance of Oligotrichida and Scuticociliatida along the conductivity gradient was similar to the trend observed along the chlorophyll gradient. The mean size of ciliates ranged from 10.3 ±0.2 µm to 193 ±35 µm (ESD) among lakes. Ciliate biovolume ranged from 10^2 to 10^6 µm^3.

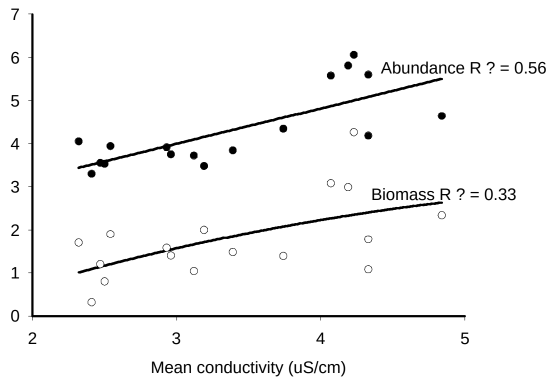

Figure 2. The relationship between mean $\log_{10}$ ciliate abundance and biomass with mean $\log_{10}$ chlorophyll *a* concentration in East African lakes. Both abundance and conductivity were $\log_{10}$-transformed.

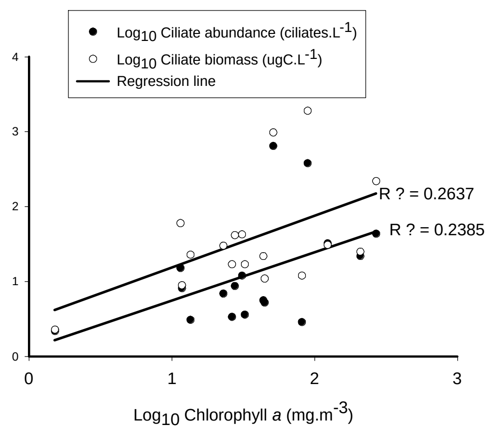

Figure 3. The relationship between mean ciliate abundance and biomas with mean conductivity in East African lakes. Both abundance and conductivity were $\log_{10}$-transformed.

Discussion

The dominant abundance of Scuticociliates and Oligotrichs in tropical (East Africa) lakes contrasts to sub-tropical (Florida) lakes where Oligotrichida, Scuticociliatida and Haptorida co-dominated (Beaver & Crisman 1982). Scuticociliates, especially *Cyclidium,* were the most abundant ciliates in most lakes and increased with increasing trophy as in subtropical and temperate lakes (Beaver & Crisman 1982). The weak negative relationship between ciliate diversity and trophy in East African lakes contradicts the positive relationship in subtropical lakes (Beaver & Crisman 1989b), and may be due to high salinity and probably the large blue-green algae that dominate saline, eutrophic tropical lakes and low diversity of edible algae. Beaver & Crisman (1989) attributed low species diversity of ciliates in oligotrophic lakes to reduced phytoplankton abundance and diversity.

Table 3. The contribution to total ciliate abundance of ciliate orders in East African lakes of different chlorophyll *a* concentration.

Ciliate orders	Chlorophyll *a* concentration (mg·m^{-3})		
	< 5	5 - 50	> 50
Armophorida	0.00	0.03	0.07
Cyrtophorida	0.00	0.88	3.36
Haptorida	0.50	3.30	0.99
Hetereotrichida	0.67	3.24	1.30
Hypotrichida	0.00	0.73	1.42
Oligotrichida	93.14	42.61	5.24
Peniculida	0.00	1.96	2.93
Peritrichida	0.00	14.52	1.15
Pleurostomatida	0.00	0.13	1.46
Prostomatida	0.50	3.18	1.77
Scuticociliatida	0.67	14.58	80.15
Stichotricha	0.00	1.14	0.01
Suctoria	0.00	0.19	0.12
Tintinnida	0.00	7.07	0.00
Others	4.52	6.46	0.04

High salinity reduces the number of species that can live in lakes (Williams *et al*. 1990). The ciliate biovolume ranged from 10^2 to 10^6 µm^3 but majority of ciliates were from 10^3 to 10^4 µm^3, similar to most ciliates in temperate lakes (Müller 1989). The ciliate abundance range of 2 to 1,220 ciliates·mL^{-1} in East African lakes is larger than reported for temperate of 3.3 cells·mL^{-1} to 21.6 cells·mL^{-1} (Pace 1986) and subtropical lakes of 10.8 to 155.5 ciliates·mL^{-1} (Beaver & Crisman 1982). Similarly, the biomass range of 1.9 to 1,900 µg C·L^{-1} is also higher than observed in both temperate and subtropical lakes of 12.3 to 56.3 µg C·L^{-1} (Pace 1986) and 9.3 µg C·L^{-1} to 126 µg C·L^{-1} (Beaver & Crisman 1982), respectively. Both ciliate abundance and biomass increase with increasing trophy in lakes of the three zones and from temperate to tropical lakes. Although not all oligotrophic lakes in East Africa have higher abundance and biomass than lakes in the temperate and subtropical zones, tropical lakes generally have higher ciliate abundance and biomass, a phenomenon attributed to higher chlorophyll concentration in tropical lakes. When regression models predicting abundance and biomass of ciliates from chlorophyll were compared for our East African lakes and temperate lakes, the slopes were not significantly different (ANOVA interaction, P > 0.05). These results suggest that temperate, subtropical and tropical lakes have similar ciliate abundance and biomass per unit chlorophyll. However, some tropical lakes have very high abundance and biomass of ciliates relative to chlorophyll probably due to high salinity. Eutrophic, saline lakes such as Elmenteita, Sonachi and Bogoria supported a higher abundance of ciliates probably due to high phytoplankton biomass and bacterial abundance (> 10^7 bacteria·ml^{-1}) (e.g., Finlay *et al*. 1987). Since chlorophyll is significantly higher in East Africa lakes than in subtropical and temperate lakes (ANOVA, $P < 0.05$), it appears that ciliate cell sizes become smaller with increase in chlorophyll, probably as a response to increasing availability of bacterial food which also increases with chlorophyll. Thus, the majority of ciliates in East African lakes are mainly bacterivorous and herbivorous. These two trophic groups account for 75% of ciliate population production (Yasindi, 2001). According to Yasindi (2001) most of this ciliate production in East African lakes is consumed by zooplankton, which are, in turn, consumed by fish (Beadle, 1981; Mwebaza-Ndawula, 1994).

It is clear from this study that both ciliate abundance and biomass increase with increasing trophy in lakes of the three zones and from temperate to tropical lakes. Also East African lakes have higher ciliate abundance and biomass than subtropical and temperate lakes mainly due to higher chlorophyll but the high abundance of bacteria is also important. Like in temperate and subtropical lakes, herbivorous ciliates dominate oligotrophic lakes while small bacterivorous ciliates dominate eutrophic lakes.

Therefore, ciliates form an important linkage through which production from the microbial loop may be transferred to the classic food chains in East African lakes as in temperate waters**.**

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References

Beadle, L. C. (1981) *The inland waters of tropical Africa*. An introduction to tropical limnology. 2nd edition, Longman London. pp. 475.

Beaver J. R. & Crisman T. L. (1982) The trophic response of ciliated protozoans in freshwater lakes. *Limnol. Oceanogr*. 22, 246-253.

Beaver J. R. & Crisman T. L. (1989) Analysis of the community structure of planktonic ciliated protozoa relative to trophic state in Florida lakes. *Hydrobiologia* 174, 177-184.

Finlay B. J., Curds C. R., Bamforth S. S. & Bafort F. M. (1987) Ciliated protozoa and other microorganisms from two African lakes, Lake Nakuru and Lake Simbi, Kenya. *Arch. Protistenk*. 133, 81-91.

Hobbie J. E., Daley R. J. & Jasper S. (1977) Use of nuclepore filters for counting bacteria by fluorescence microscopy. *Applied and Environmental Microbiology*, 33, 1225-1228.

Laybourn-Parry J. (1992) *Protozoan plankton ecology* pp 231. Chapman and Hall, London.

Montagnes D. J. S. & Lynn D. H. (1993) A quantitative protargol stain (QPS) for ciliates and other protists. In: Kemp P. F., Sherr B. F., Sherr E. B. & Cole J. J. (eds.), *Handbook of methods in Aquatic Microbial Ecology* pp 229 – 240. Lewis Publishers, Boca Raton.

Müller H. (1989) The relative importance of different ciliate taxa in the pelagic food web of Lake Constance. *Microb. Ecol*. 18, 261-273.

Pace M. L. (1986) An empirical analysis of zooplankton community size structure across lake trophic gradients. *Limnol. Oceanogr*. 31, 45-55.

Pace M. L. & Orcutt J. D. (1981) The relative importance of protozoans, rotifers, and crustaceans in a freshwater zooplankton community. *Limnol. Oceanogr*. 25, 822-830.

Porter K. G., Pace M. L. & Battey J. F. (1979) Ciliate protozoans as links in freshwater planktonic food chains. *Nature,* 277, 563-565.

Putt M. & Stoecker D. K. (1989) An experimentally determined carbon:volume ratio for marine 'oligotrichs' ciliates from estuarine and coastal waters. *Limnol. Oceanogr*. 34, 1097-1103.

Statsoft Inc. (1995) *Statistica for windows* [Computer Program Manual]. Tulsa, OK. Statsoft, Inc., 2325 East 13th Street, Tulsa, OK, 74104.

Williams W. D., Boulton A. J. & Taaffe R. G. (1990) Salinity as a determinant of salt lake fauna: a question of scale. *Hydrobiologia* 197, 257-266.

Yasindi, A. W. (2001) The ecology of planktonic ciliates in tropical lakes of East Africa, Africa. Ph.D Thesis, University of Waterloo, Waterloo, Ontario, Canada, pp. 221.

Yasindi A. W., Lynn D. H. & Taylor W. D. (2002) Ciliated protozoa in Lake Nakuru, a shallow alkaline-saline lake in Kenya: seasonal variation, estimated production, and role in the food web. *Arch. Hydrobiol*. 154, 311–325.

Environmental impact of land use change on water quality of inflowing tributaries of Lake Kivu

Bagalwa M.

Department of Biology, Center of Research in Natural Sciences, Democratic Republic of Congo, C/O: Petit Seminaire de Mugeri, B.P. 02, Bukavu
Email: mashibagalwa@yahoo.fr

Abstract

Impact of land use on water quality of lake Kivu inflowing tributaries was assessed from April to July 2005. This work evaluates the impact of land use change on water quality of inflowing tributaries of Lake Kivu, with the objective of determining if the water draining from forested areas is significantly different than that draining from unforested areas.

Lake Kivu plays an important role in the economies of the riparian countries (Rwanda and Democratic Republic of Congo). Fish provide a major source of protein in the local diet. Fishing related activities are source of income and employment for more than 1 million people. Transport is another major industry on the Lake, which serves as a super high way connecting people and goods within and between the two countries and provinces.

In this study we consider Phosphorus, Nitrogen, Dissolved Oxygen, Alkalinity, Temperature and Suspended solids as biologically significant variables affecting either by natural processes or by anthropogenic impacts. Maximum and minimum observed values are as follow: Total Nitrogen (676 – 9 µmole/L), Total Phosphorus (647 – 0.2 µmole/L), Dissolved Oxygen (12 – 4 mg/L), Temperature (20 – 15 oC) and Suspended solids (0.35 -0.03 mg/L). The discharge presentes a characteristic seasonal variation showing a peak in May when the rain is high. Differences are observed between forested and unforested areas. Rivers Kauwa and Chula show the high concentration of Total Nitrogen, Total Phosphore and Suspended solids. Increasing of theses parameters with increasing discharge could be associated to the high population growth rates, accelerating urbanization and industrialisation accompanied by intensification of agriculture, soil erosion and use of agro-chemicals.

Key words: land use, water quality, Lake Kivu

Introduction

Lake Kivu is the smallest of the East African Great Lakes. It is a meromictic lake with an oxic mixilimnion of 70 m and a deep monimolimnion rich in dissolved gases, particularly methane. Lake Kivu has unique limnological characteristics with temperature and salinity increasing in the deep water layers due to the input of geothermal sources of the bottom of the Lake. The fauna and flora of Lake Kivu is rather poor. At present the fish community of Lake Kivu comprises no more than 26 species probably as a result of intense volcanic and tectonic activity in this region during the late Pleistocene and Holocene (Haberyan and Hecky, 1987).

Lake Kivu plays an important role in the economies of the riparian countries (Rwanda and Democratic Republic of Congo). Fish provide a major source of protein in the local diet. Fishing related activities are source of income and employment for more than 1 million people. Transport is another major industry on the Lake, which serves as a super high way connecting people and goods within and between the two countries and provinces (Mughanda, 1993).

Water quality constituent in a river derive from both natural and anthropogenic sources. Natural sources include non point sources as weathered rock and soils, atmospheric deposition and decay of organic material. Water quality constituents from these sources can enter a river in surface, runoff, direct precipitation or in ground water that discharges to a river or stream. Anthropogenic sources include point sources such as wastewater discharge from industrial and domestic treatment plants and non point sources including land affected by agriculture or urban development.

Humankind has been altering the chemical composition of rivers for millennia, with early human settlements discharging foecal pollutants, organic substances and even trace of metals into rivers (Meybeck and Helmer, 1989). Numerous lowland rivers in the world are contaminated and lose their characteristic biodiversity (Moss, 1988). Chemical changes of their waters come from different causes such as the inflow of wastes, nutrient and pesticides from urban and agricultural areas or nutrients and dissolved forms of organic carbon from drained organic soils. Theses are intensified by large-scale industrial development and more intensive agricultural practices.

The nature of threats to rivers differs from region to region. The destruction of rivers habitats has been so extensive in some developed countries that there is now a perceived necessity to protect what is left or restore degraded systems. Information on anthropogenic impacts on the rivers in this region is patchy. There are few studies that document long-term trends in biota in relation to increased nutrient loading in rivers. There are important gaps in knowledge concerning the impacts of many pollutants and a paucity of data for developing regions. Although all increases are concomitant either with increased land-use change or human development elucidating precisely the factors which are responsible for the trends, there are multiple underling causes. areas that may be particularly vulnerable in the near future. Differences in concentrations and loads among rivers can be considered as a result of many interrelated characteristics including land use, point sources, basin size, runoff characteristics, streambed, stream flow velocities and hydrogeologic setting.

The understanding of mechanisms regulating surface water chemistry is a fundamental issue for aquatic ecologist and water resource managers (Meyer *et al.,* 1988). In developed countries, several rivers showed increased nutrient concentration over the past century. Dissolved P and N play an important role in varied biogeochemical cycles (Dowing, 1997; Dodson, 2005). The influence of the hydrological parameters of rivers on water chemistry and its changes along the river course has not yet been comprehensively studied, especially in Democratic Republic of Congo (Bagalwa, in press). Lake Kivu is threatened by several potentially disastrous environmental problems.

This work evaluated the impact of land use change on water quality of inflowing tributaries of Lake Kivu, with the objective of determining if the water draining from forested areas is significantly different from that draining from unforested areas.

Materials and methods

The Kauwa, Chula rivers in the southern and the Lwiro, Cirhanyobowa the largest tributaries rivers in north-eastern of lake Kivu in Democratic Republic of Congo have been selected for investigation as relatively unforested and forested rivers ecosystems. This region includes both heavily impacted rivers due to population increases, cultivation and burning as well as the undistrurbed rivers with vegetated watersheds. Multiannual discharge of these rivers varies from 5 m^3/s in dry season to 150 m^3/s in rain season (Shamaa *et al,* 1981) . The mean river discharge of the lake Kivu tributaries in Democratic Republic of Congo do not exceed 10 m^3/s, but the specific runoff from the Kauwa river catchment is almost two times higher than others rivers. The Kauwa and Chula catchment areas are dominated by extensive agriculture, high populated area, small degree of forestation and high denivelation to 170 m.

In April, May, June and July (4 times a year in 2005) water samples have been collected from two stations located along theses rivers. The samples have been taken from the upper water layer in place with medium velocity of the water flow. In each station samples were collected using pre-washed glass bottles for measurement of Dissolved Oxygen (DO) and others chemical parameters. At each sampling station water temperature, DO and pH have been measured. DO was determined following the iodometric Winkler's method (Golterman et al, 1978). Chemical parameters, 5-day Biological Oxygen Demand (BOD_5), alkalinity and suspended solids are measured following the procedures described in Golterman *et al*., (1978). BOD_5 is measured as the decrease in DO after incubation in the dark at 20 ºC for 5 days. Unfiltered water is used for total alkalinity (meq/L) using 0.1 N HCl titration. Suspended solids (mg/L) were estimated by filtration of 1 L of water with pre-weighed analytical filter paper (Whatman 589, 185 µm pore size) which is dried at 105 ºC and pre-weighed. Phosphorus and Nitrogen are analysed by standard methods for examination water (Wetzel and Likens, 2000). Analyses for Total Phosphorus (TP, persulfate digestion and ascorbic acid method) and Total Nitrogen (TN, persulfate digestion, colorimetric indophenol blue method), are performed at the laboratory.

T tests are performed to compare the y-distribution of forested and deforested rivers for chemistry parameters.

Results and discussion

Eight physico-chemical parameters have been determined for four rivers over a four months period in rainy season (April- May) and in dry season (June - July). Table 1 and 2 summarises the mean values of the various parameters monitored at the four selected rivers.

Table 1. Temporal variation of physico - chemical parameters in unforested rivers tributaries of Lake Kivu.

	Kauwa River				**Chula River**			
Parameters	April	May	June	July	April	May	June	July
Temperature (oC)	20	19	18.5	18.5	20	19	20	19
PH	6.4	5.5	6.8	7	6.4	5.3	6.6	6.9
DO (mg/L)	3.47	3.68	4.12	5.69	5.17	5.86	5.74	5.66
DBO5 (mg/L)	1.41	1.21	1.89	3.36	4.32	5.46	3.52	4.1
Alkalinity (mg/L)	4	4	11	14	4	6	8	14
Suspended Solids (mg/L)	0.35	0.21	0.07	0.06	0.06	0.09	0.09	0.01
Total nitrogen (µg/L)	283.2	266.5	84.9	14.3	183.3	266.5	9	15.8
Total phosphore (µg/L)	647.38	648.27	219.86	25.143	82.32	58.22	2.853	3.31

Table 2. Temporal variation of physico - chemical parameters in forested rivers tributaries of lake Kivu.

	Lwiro River				Cirhanyobowa River			
Parameters	April	May	June	July	April	May	June	July
Temperature (oC)	15.5	14.7	15.2	17.5	14.5	15	15	15.2
pH	5.6	5.6	6.8	6.4	5.3	5.6	6.4	6.4
DO (mg/L)	8.17	8.58	9.1	8.9	7.21	8.33	9.1	9.9
DBO5 (mg/L)	4.95	4.64	5.83	5.46	5.46	5.66	5.72	5.69
Alkalinity (mg/L)	7	6	12	10	4	4	6	6
Suspended Solids (mg/L)	0.1	0.11	0.04	0.04	0.001	0.003	0.001	0.002
Total nitrogen (µg/L)	333.2	299.8	616.4	116.6	316.5	99.9	216.6	299.8
Total phosphore (µg/L)	17.85	27.49	115.57	64.31	26.42	15.88	29.631	88.8

A perusal of the data shows the increases of some pysico-chemicals parameters from unforested rivers (Kauwa and Chula) but the forested rivers (Lwiro and Cihranyobowa) are still pristine (Meybeck and Helmer, 1989).Water temperature presents a defined pattern of seasonal variation with a maximum of 19,5 ºC in rainy season and minimum in the dry season (15 ºC) (Table 3). The pH trend was slightly acid with an average value around 5.6. This trend was probably due to a decrease of suspended particulate materials, loading to a large transparency in dry season.

Table 3. Seasonal variation of physico-chemical parameters in forested and unforested rivers tributaries of lake Kivu.

	Kauwa River		Chula River		Lwiro River		Cirhanyobowa River	
Parameters	Rainy	Dry	Rainy	Dry	Rainy	Dry	Rainy	Dry
Temperature (oC)	19.5	18.5	19.5	19.5	15.1	16.35	14.75	15.1
pH	5.95	6.9	5.85	6.75	5.6	6.6	5.45	6.4
DO (mg/L)	3.58	1.91	5.52	5.7	8.375	9	7.77	9.5
DBO5 (mg/L)	1.31	2.625	4.89	3.81	4.795	5.645	5.56	5.705
Alkalinity (mg/L)	4	12.5	5	11	6.5	11	4	6
Suspended Solids (mg/L)	0.28	0.065	0.075	0.05	0.105	0.04	0.002	0.0015
Total nitrogen (µg/L)	274.85	49.6	224.9	12.4	316.5	366.5	208.2	258.2
Total phosphore (µg/L)	647.825	122.5	70.27	3.08	22.67	89.94	21.15	59.2155

The mean levels of oxygen demanding substances in the unforested rivers varied between 3.47 mg/L and 5.86 mg/L. For forested rivers, it was about 6.17 mg/L and 9.9 mg/L. Dissolved oxygen is an important element for water quality control. The effect of waste discharge on unforested rivers is largely determined by the oxygen balance of the system and its presence is essential to maintain biological life within a system (Fatoki et al, 2003). The use of the rivers by population for many activities and the change of climate are good reasons for predicting this decline in the unforested rivers. It is admitted that self depuration potential of a stream is exceeded when oxygen demand saturation sinks under 50 %. Pristine surface waters are normally saturated with oxygen demand but such DO can be rapidly removed by the DO of organic wastes. The measurement of DO concentration in unpolluted water are normally about 8 to 10 mg/L (at 25 ºC). The concentration below 5 mg/L negatively affects aquatic life. The concentrations of DO in the unforested rivers are less than 5 mg/L and would not be suitable for use of the aquatic organism.

BOD values indicate the extent of organic pollution in the aquatic system which negatively affect the water quality. In all the sample, BOD values were low and less than 6 mg/L. The Kauwa river had very low BOD values raging between 0.41 mg/L (rainy season) and 3,36 mg/L (dry season). Relatively high values are observed in the Cirhanyobowa river. Low BOD values reflect low burden of organic pollution in Kauwa river. The water from the runoff that carries the seepage from Bukavu city clearly increases pollution burden of Kauwa river. This is reflected by the lower pH, phosphore and nitrogen obseved to others rivers in Africa (Vandelannoote *et al.,* 1996).

The suspended solids concentration varied from a river to another. The unforested rivers seem to have higher concentration than the forested ones. The recorded values in the rainy season were greater than those recorded in dry season. In the dry season, agriculture is not intensively done and erosion is minimal. Agriculture development and deforestation may alter the suspended solid load by increasing erosion and altering local sediment budgets (Hecky *et al.,* 2003). The low suspended solids concentrations observed in forested rivers reflect the filtering effect of forest in upstream of these rivers.

Alkalinity is indicative of the type of soils and underlying rock of the area. In our rivers it was high comparatively to others African rives. Kilham (1990) reports an alkalinity range of about 2.5 ± 1 meq/L for Africa rivers which is high in our sampling rivers. Several reasons are responsible for the difference of alkalinity in these rivers such as the natural variability of soils and geologies, the discharge of river, agricultural activities at the shoreline and weathering (Kilham, 1990; Meybeck and Helmer, 1989). Most of the water entering rivers must first pass through terrestrial catchment which differ markedly in their geologies, soils and vegetation and hence contribute to a large natural range of river chemistry. In the basin of lake Kivu, a volcanic region, soil contains limestone minerals such as calcite (CaCO3) and dolomite ($CaMg(CO_3)_2$) and therefore alkalinity in the rivers tributaries of lake Kivu is high compared to others African rivers (Meybeck and Helmer, 1989). The effect of rock weathering on ion chemistry can be expected to be the dominant mechanism influencing water chemistry in the rivers tributaries of Lake Kivu. Our observation is consistent to the results observed in elsewhere in other african rivers. (Baca and Threlkeld, 2000; Gibbs, 1970).

Phosphorous and Nitrogen are important parameters to assess the water quality. Mean levels of total phosphore were about 211 µg/L in unforested rivers and 41 µg/L in forested rivers. The high concentration was recorded in Kauwa river (648.27 µg/L) an unforested river. This river flow through Bukavu town drawing waste waters from domestic uses to the lake. The highest phosphorus values found in the rainy season can be associated with the bottom organic matter mineralization by bacteria and microzooplancton under high temperature, resulting from fat mixture in the water column (Nixon, 1981). Total nitrogen levels average 140.4 µg/L in unforested rivers and 287.4 µg/L in forested rivers. It was find that where the riparian zone has been cut, the streamside buffering capacity is reduced and higher levels of dissolved nitrogen will enter the stream via groundwater baseflow (Dodson, 2005). Baseflow itself increases because of reduced plant uptake of infiltrating rainwater. The increase of the concentration of this nutrient in rainy season, as well as its decrease during the dry season suggest that the input of untreated domestic sewers might be strongly influencing the dynamic of dissolved organic nitrogen in river system. According to Golterman (1975) among the sources of nitrogen to the aquatic environment, stand out those of human origin derived from agriculture, humans excretes and detergents.

Nutrient concentration in the present study are higher than the levels described for unpolluted rivers (Meybeck and Helmer, 1989). High levels of these species increase the growth of vegetation in water systems and the oxygen demand. The enrichment in nutrients and the enhancement of productivity and respiration leads to such imbalance. The forested rivers have lower values than unforested rivers. The table 3 shown that the forested rivers contained high concentration of total nitrogen and total phosphorus in dry season then unforested rivers. In the dry season the used of swam for agriculture which used many pesticides for leguminous can be the reason of the highest concentration. In this period the unforested rivers have not received water provide from erosion which can bring materials to increase nutrient concetration. But in the rainy season the unforested rivers get much water from erosion and than the concentration of nutrient increased.

The large oscillation of the water level and the consequent variation of flooded area lead to a dynamic interaction process between aquatic and terrestrial environments. Thus, the terrestrial environment can act as a source of nitrogen and phosphorus to the river water. Another source can be associated to the agricultural practices (burning, harvesting, fertilisation and irrigation) which can transfer nitrogen and phosphorus compounds to the rivers channel especially during the rainy period. The burning of forest before the harvest transfer large amounts of nitrogen from the soil-vegetation system to the atmosphere. Also, the input of domestic sewage of several cities is a continuous source of nutrients that become particularly important in rivers.

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References

Baca R.M.& Trelkeld S.T., 2000. Inland dissolved salt chemistry: Statistical evaluation of bivarate and ternary diagram models for surface waters. *Journal of Limnology*, 59, 156 - 166.

Bagalwa M., (submitted). *Impact of land use on water quality of the Lwiro River, Democratic Republic of Congo, Central Africa*.

Dodson S., 2005. *Introduction to limnology*. McGraw Hill, New York, 400 p.

Dowing J.A., McClain M., Twilley R., Melack J.M., Elser J., Rabalais N.N., Lewis W.M., Turner R.E., Corredor J., Soto D., Yanez-Arancibia A., Kopaska J.A., Howarth R.W., 1999. The impact of accelerating land-use change on the N-cycle of tropical aquatic ecosystems current conditions and projected changes. *Biogeochemistry*, 46, 109 - 148.

Fatoki D.S., Gogwana P., Ogunfowokan A.O., 2003. Pollution assessment in the Keistamma River and it the impoundment downstream. *Water S.A.,* 29,2, 183 - 187.

Gibbs R.J., 1970. Mechanisms controlling world water chemistry. *Science* 170, 1088 - 1090.

Golterman H.L., 1975. *Physiological limnology*. Elsivier, New York, Amsterdam, 489pp.

Golterman H.L., Clymo R.S. & Ohnstad M.A.N., 1978. *Methods for physical and chemical analysis of freshwaters*. Blackwell scientific publication, London, 213p.

Haberyan K.& Hecky R.E., 1987. The late Pleistocene and Holocene stratigraphy and paleolimnology of Lakes Kivu and Tanganyika. P*alaeogeography, Palaeoclimatology, Palaeoecology,* 61, 169 - 197.

Hecky R.E., Bootsma H.A. & Kingdon M.L., 2003. The importance of river basin characteristics and impact of land use change on sediment and nutrient yields to lake Malawi/Nyassa (Africa). *Journal of Great lakes Research*. 29 (suppl.2): 139 - 156.

Kilham P., 1990. Mechanisms controlling the chemical composition of lakes and rivers: Data from Africa. *Limnol. Oceanogr.,* 35, 1, 80 - 83.

Meybeck M. & Helmer R., 1989. The quality of rivers: from pristine stage to global pollution. Palaeogeography. *Palaeoclimatology, Palaeoecology*, 75, 283 - 309.

Meyer J.L., McDowell W.H., Bott T.L., Elwood J., Ishizaki C., Melack J.M., Peckarsky B., Peterson B., Rublee P., 1998. *Elemental dynamics in streams*. J.M., Am, Benthol. Soc. 7: 410 - 432.

Moss B., 1988. *Ecology of fresh waters- man and medium*. Blackwell Scientific Publications, Oxford.

Nixon S.W., 1981. Remineralization and nutrient cycling in coastal marine ecosystems. *In Estuaries and Nutrients*, 111 - 138.

Shamaa M., Bidou J., Boureau P.Y., 1981. *Atlas de la ville de Bukavu*. CERUKI, ISP/Bukavu.

The influence of groundwater on lake-water management: the Naivasha Case

Robert Becht[1] and John Rao Nyaoro[2]

[1]TC-International Institute for Geo-Information Science and Earth Observation, PO Box 6, 7500 AA, Enschede, the Netherlands.
[2]Water Services Regulatory Board, Nairobi, Kenya
Email: Becht@itc.nl

Abstract

Lake Naivasha, a lake in the Kenyan Rift Valley, is surrounded by shallow aquifers intimately linked to the lake. The analysis of the groundwater levels north of the lake show the development of a large depression cone caused by to large-scale abstractions for irrigation and drinking water supply. The natural flow in this area has reversed and water is now flowing from the lake towards the well field. Three distinct groundwater zones can be distinguished with different water use efficiencies and pollution hazard. A simple water balance model with a lumped aquifer reservoir clearly shows the influence of groundwater describing the system: if the groundwater node is de-activated the simulation of the lake levels overshoots 1 meter after periods of recovery or recession. A special version of USGS ground water model code Modflow, with a lake module, is used to simulate the long-term effects of groundwater abstraction. The simulation shows that it takes 10 of years to establish a new equilibrium state and an equal long period to recover from the exploitation. The slow response of the lake-groundwater system requires a long planning horizon from water managers.

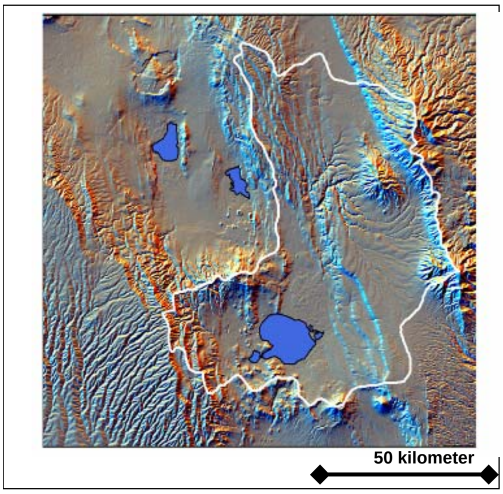
Key words: groundwater, modeling, water management, lake.

Introduction

Ground water is critical for understanding most lake systems because it influences a lake's water budget and nutrient budget. Several studies report on the use of a simple mass balance approach to simulate lake levels from hydrological and meteorological data (Becht and Harper, (2002), R. N. Jones *et al.,* (2001), Almendinger (1990)). Groundwater is not considered at all or simplified by lumping the aquifer system in as one reservoir. A lake water balance model linked to a groundwater model is required to study the behavior of the system in more detail. Efforts to simulate ground water-lake interaction include the LAK1 (Cheng and Anderson, 1993) and LAK2 (Council, 1998) add-on "packages" to the widely used U.S. Geological Survey modular finite-difference flow model (MODFLOW). Most studies relate to research-basins and very few address the implications of a lake-groundwater system on the water management of the basin.

Physical description of the study area

Lake Naivasha is a shallow, endorheic, freshwater lake situated in warm and semi-arid conditions in the Rift Valley of Kenya. The lake is situated approximately 80 km North-West of Nairobi. The lake receives an average rainfall of 600 mm year^{-1}. lake transpiration is approximately 1700 mm year^{-1}. Lake Naivasha is fed by two perennial rivers, the Malewa and the GilGil, receiving their water from the highlands bordering the rift valley. During heavy rains the lake also receives water from direct runoff into the lake and some ephemeral streams. The natural variation of lake level was 12 meters over the last century. Around the lake a vibrant horticultural economy is fully dependent on the available water resources.



Hydrogeology

It has long been known that a substantial part of the surface inflow recharges the surrounding shallow and deep aquifer systems. This recharge that flushes the lake is the reason that the lake remains fresh. The lake and its surrounding aquifers form one tightly linked system. The shallow aquifers are composed of a complex mixture of water and air laid pyroclastic material, and lacustrine deposits. The intercalated layers of pumice lapili play an important role because of the very high transmissivity. Values of more than 10,000 m^2/day have been derived from pumping tests. The specific yield of the aquifers has an average value of 0.15. Due to the high transmissivity of the shallow aquifer the gradients of the water table low, and the lake surface extends almost horizontally into the aquifer. This water of the shallow aquifer recharges the deeper regional aquifer system. This system is partly geothermal and discharges to the lakes in the North and in the South (Clarke,1990), *Becht et al.,* 2005). The groundwater levels in the vicinity of the lake closely follow the water level variations of the lake. Therefore a substantial portion of the available water resources and the changes in storage driven the variations of precipitation occurs in the groundwater reservoir. Based on accurate measurements of the water table, isotope analysis and flow system analysis the general direction of groundwater flow has been determined. The direct groundwater recharge to the aquifer is low and does not play an important role in the groundwater balance.

Exploitation of water resources for irrigation.

Lake Naivasha being a fresh lake has been exploited for irrigation since the 1940s. In these days the water was pumped directly from the lake and in the lower Malewa a weir raised the water to allow gravity irrigation. In 1972 a large cattle farm on the Northern side of the lake sunk 12 wells used for the irrigation of fodder, mainly grass end alfalfa. Several of these wells are still in use. In the mid 1970's 3 boreholes were sunk North of the lake for the drinking water supply of Naivasha town. Since the early 1980's the flower industry around the lake has changes the rules of the game. The first farm started in 1980 at the South-Western side of the lake. The success of the flower production has caused a booming of this industry and at present a large part of the Southern shores is occupied with flower farms all using lake water for irrigation. Until 1999 the groundwater North of the lake was exploited for the water supply of Naivasha town and for fodder production. Than groundwater based horticultural industry started also North of Lake Naivasha and rapidly a large area was put under cultivation.

The inflow into the lake through the Rivers Gilgil and Malewa is also reduced. In 1992 a pipeline became operational pumping 20,000 $m^3 day^{-1}$ from the Malewa basin to GilGil and Nakuru town. Recently horticultural farm have started operations in the upper catchment taking water from the river.

At present the total irrigated area is approximately 40 km^2. During the large 20 years the irrigation technology has seen a shift from overhead sprinkler to more sophisticated systems as central pivots and drip irrigation. Also many farmers have moved from outdoor cultivation to greenhouses. The latest trend is hydroponics culture in greenhouses. The changes in irrigation practices definitely help to conserve water.

Shallow groundwater around Lake Naivasha

Figure 2 shows a satellite image of the lake with the known wells and the direction of groundwater flow.

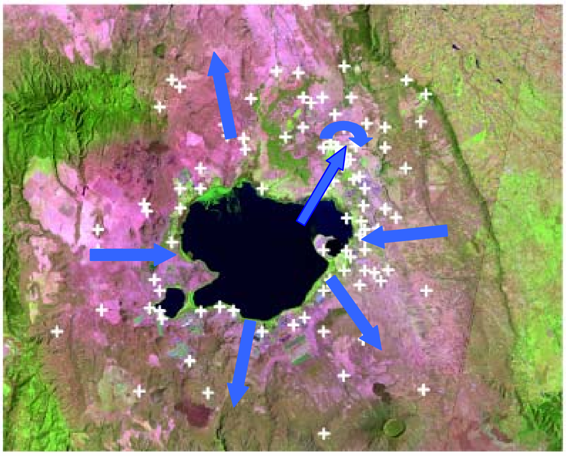


Figure 2 : Lake Naivasha with wells (+) and general direction groundwater flow

The major flows are the outflow to the North and South draining the lake. Some ground water inflow occurs from the flanks of the Rift Valley towards the lake. North of the lake, in the area with important groundwater exploitation, the natural gradient is inversed and a groundwater depression has been formed. The water pumped from this area comes indirectly from the lake. The rather complicated flow pattern has implications for the management of the lake. The southern zone is an outflow zone with much lake water based irrigation. Over-irrigation is this zone will cause a net water loss since the excess water drains towards the shallow aquifer. Eventually, this water will re-appear in Lake Magadi after 10,000s of years. Agrochemical pollutants carried by the water will have the same fate. In outflow zones careful irrigation management preventing over irrigation is important, the risk that pollutants are carried to the lake by groundwater is small. Exactly the opposite is true for the inflow zones. In case of over irrigation the access water will find its way to the lake and can be reused. Of course the same holds true for pollutants, constituting a danger for the ecological quality of the lake. In the groundwater depression North of the lake water is re-circulated. All irrigation in this zone is based on groundwater. Irrigation return flow from over-irrigation will recharge the aquifer and the concentration of salts and pollutants is likely to increase.

The temporal evolution of the cone of depression caused by the pumping North of the lake is shown in figure 3. The transect runs from the lake shore over the irrigated area to the Malewa river.

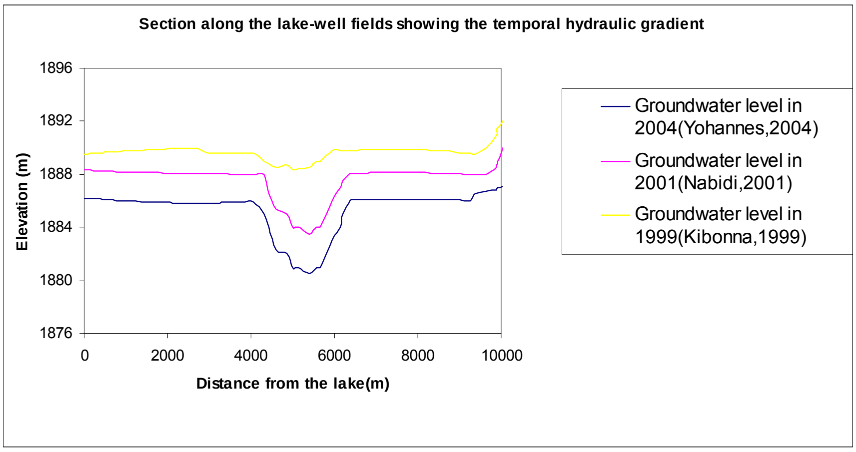


Figure 3: Transect of groundwater levels north of Lake Naivasha.

The cone of depression has reached a depth of more than six meters as compared to the natural water table. The natural flow in this area was directed towards the lake but has inversed. Water is now flowing from the lake towards the pumped zone.

The groundwater has an important effect on the water balance of the lake. If the lake levels rise the lake will recharge the surrounding aquifers. If the lake recedes the aquifers will discharge into the lake. This interaction causes inertia to the lake-groundwater system causing delayed reactions to external (meteorological) stresses. The groundwater acts as an extra reservoir absorbing water wet periods and releasing water during droughts.

This phenomenon can be shown using the Lake Water Balance Model (Becht, *et al.,* 2002). The model schematizes the aquifer as one reservoir hydraulically linked to the lake.

In figure 5 the effect of the groundwater on the lake levels is indicated by the arrows.

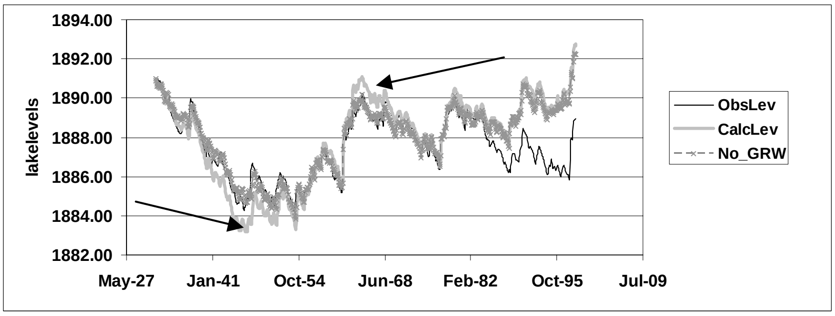


Figure 4. Observed and simulated water levels.

At the end of the long drought periods between 1930 and 1950 the lake levels are simulated one meter lower than the observed levels due to the lack of groundwater inflow. The opposite is true during the period of rising lake levels in the 1960's where a one-meter overshoot of the simulated levels takes place. This can be explained by the absence of a flow recharging the aquifer. The model with a groundwater component accurately follows the observed levels, whereas the same model with no groundwater component overshoots after periods of rise or recession. . The deviation starting 1982 between observed and modeled levels is due to the abstraction for irrigation from the basin (Becht *et al.,* 2002)

The groundwater regime around Lake Naivasha is complicated. Groundwater models with increasing complexity have been constructed by Owor (1999), Kibona (1999) and Nabide (2002) and Yohanis (2005). All models are based on Modflow, the groundwater modeling code of the United States Geological Survey (USGS) with the lake pachkage.

These models fully link the lake and the groundwater and enable a more reliable and sophisticated analysis of the role of groundwater than the simple water balance model presented earlier.

To illustrate the importance of groundwater in the management of Lake Naivasha the following scenario has been simulated using the models. The lake groundwater resources North of the lake are exploited at a rate equal to the actual abstraction. (55 m^3 $year^{-1}$) starting in 1932. The full abstraction is taken from groundwater, none from surface water.

The results are shown in figure 6.

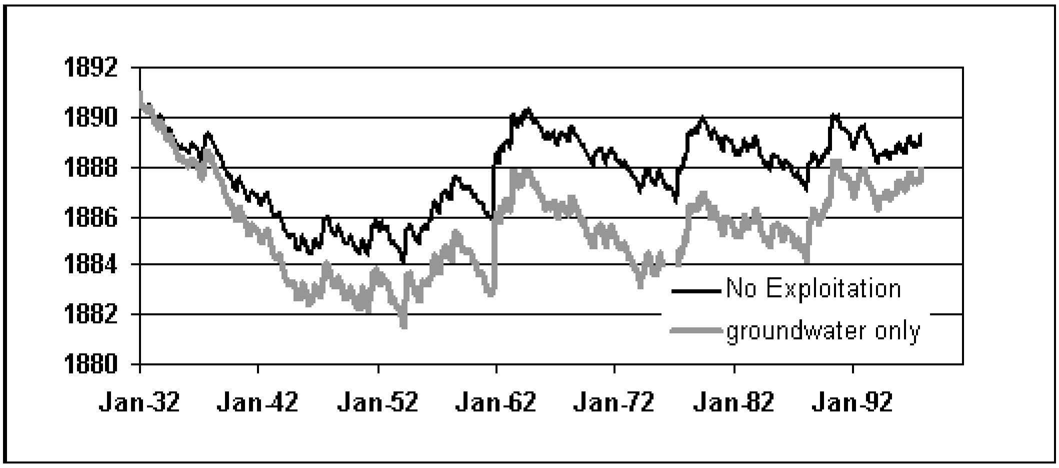


Figure 5. Difference between levels in figure 6.

Figure 6

The equilibrium drawdown of the lake level can be calculated from the long-term water balance and is 3.5 meter. Figure 7 shows that this level is reached after a period of 35 year. Even after the moment the exploitation of the aquifer has stopped in 1970 it takes 6 years before the lake level reacts and start to rise. At the end of the simulation period in 1997 the difference the lake levels of an exploited lake and un-exploited lake is 1.5 meter. This means that 25 years after the termination of the exploitation the lake has only recovered 50% towards its natural equilibrium state.

Water management

The very slow reaction time of the lake groundwater system demonstrated by the simulations has important repercussions for the management of Lake Naivasha.

The drawdown effect of groundwater abstraction on the lake level takes decades and the recovery of the lake takes a period of similar length. This effect should be considered for the management of the lake. The groundwater abstractions taking now place in the area North of lake Naivasha will affect the lake levels for the coming years even if groundwater exploitation would be fully banned right now.

At the other hand the presence of good aquifer around the lake constitutes a large source of water and could be used for optimized water management. Especially after a wet period with a rise of the lake levels the water stored in the lake could be used to artificially recharge the aquifers. The techniques, also know as Managed Aquifer Recharge Systems (MARS) could play an important role in the rational use of water. Water that would naturally discharge into the lake like runoff from greenhouses, runoff from the perennial and ephemeral rivers or even water from the lake could be injected in the aquifer to be used in periods of water shortage.

Conclusions

The analysis of the flow direction around Lake Naivasha shows that 3 different groundwater zones can be distinguished: (1) a zone were over irrigation leads to a waste of water, (2) a zone were over-irrigation leads to pollution of the lake and (3) a zone were water is recycled and the water and solute balance of the lake is not affected.

With the lake water balance model including a lumped groundwater node the effect of groundwater can be clearly shown. If the groundwater node is disabled the simulation is less accurate especially after a periods of recovery or recession. A more realistic simulation using Modflow with the lake Module shows that the response and recovery time of the system after exploitation of the aquifer is tens of years.

The presence of good aquifers around the lake pose a problem since the water management is more complex; at the other hand they constitute an important opportunity as a buffer of water allowing a more rational management of the available resources.

Acknowledgements

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References

Anderson, M.P. and X. Cheng. 1993. Long- and short-term transience in a groundwater/lake system in Wisconsin, USA. *Journal of Hydrology*. 145:1-18

Jones, R. N. McMahon T. A. and. Bowler J. M 2001.Modelling historical lake levels and recent climate change at three closed lakes, Western Victoria, Australia. *Journal of Hydrology*, Volume 246:159-18

Almendinger J.E, 1990. Groundwater control of closed-basin lake levels under steady-state conditions. *Journal of Hydrology*, Volume 112, Issues 3-4

Council, G.A. 1998. Lake Package for MODFLOW. *In.* E. Poeter, C. Zheng, and M. Hill eds.*MODFLOW '98: Proceedings of the 3rd International Conference of the International Ground Water Modeling Center*,. 675-682. Golden, Colorado: Colorado School of Mines.

Becht R & Harper D.M. Towards an understanding of human impact upon the hydrology of lake Naivasha. *Hydrobiologia,* 2002: 488, 1-11.

Owor, M., 2000, *The long term interaction of groundwater with lake Naivasha, Kenya : a numerical simulation of the relationship between groundwater and lake allowing for fluctuating lake levels*. MSc Thesis, ITC, The Netherlands

Sekela Rodrick Kibona, 2000. *Temporal and spatial variation of groundwater level north of lake Naivasha, Kenya - analysed using modflow*. MSc Thesis, ITC, The Netherlands

Nabide, I.K., 2002. D*evelopment of 3-D conceptual hydrogeological model for Lake naivasha area : based on the integration of geology, hydrochemistry, isotopic analysis and boundary conditions*. MSc Thesis, ITC, The Netherlands

Yihdego, Y., 2005. *A three dimensional ground water model of the aquifer around lake Naivasha area, Kenya*. MSc Thesis, ITC, The Netherlands

Clarke, M. C. G., *et al.,*1990. *Geological, volcanological and hydogeological controls of the occurrence of geothermal activity in the area surrounding Lake Naivasha, Kenya*. Min. of Energy, Kenya and British Geological Survey.

Sediment loading on inland lakes/wetlands: A case study of lake Nakuru, Kenya

Maina-Gichaba C.[1], W. A. Shivoga[1], E. M. Enanga[1], S. Kibichii[2] & Scott N. Miller[3].

[1]Egerton University, P.O. Box 536, Njoro
[2]Department of Fisheries, Moi University P.O. Box 1125, Eldoret
[3]University of Wyoming, Department of Renewable Resources, P.0. Box 3354, Laramie, Wyoming, 82071, US.
Email: maina_gichaba@yahoo.co.uk

Abstract

Total suspended sediments and discharge were studied on the river mouths and sewage drain that empty into Lake Nakuru, Kenya. Total suspended sediment loading into Lake Nakuru is a function of the concentration of total suspended solids and discharge at each mouth. The study was conducted at the mouths of rivers Njoro, Makalia, Nderit, Baharini springbrook and Sewage drain. *In situ* measurements of discharge were done, at each mouth, and 500ml water samples were taken and determination of total suspended solids done in the laboratory. Historical data was also used to provide typical discharge and total suspended solids values for each month. Loading was then calculated for each river mouth and the sewage drain. Although River Nderit had the highest concentration of total suspended solids, than rivers Njoro and Makalia, it delivered a lower amount of total suspended solids due to its lower discharge volume. River Njoro was found to deliver the most loads of total suspended solids (70%) despite having a lower concentration of total suspended solids followed by Makalia (21%), Nderit (4%), Sewage drain (4%) and Baharini (1%)in that order. Baharini is fed by a number of clean water springs emanating from the Lake Nakuru National Park which is a protected area. All the other rivers, especially Njoro and Makalia drain rural and urban watersheds with intensively cultivated easily eroded landscapes. This study shows that rivers emanating from outside protected areas under agriculture and urbanization contribute the highest to lake sediment loading and management of sedimentation of Lake Nakuru should focus mainly on soil management in the landscape.

Key words: discharge, loading, total suspended solids

Introduction

The increasing human populations in tropical catchments imply that more intensive agricultural activities have to be adopted to satisfy food requirements. Usually, this is achieved by increasing area under crop production and livestock grazing. In tropical Africa, another trend towards food satisfaction is the conversion of forests to agricultural land. In both cases, there is a net increase in eroded particles to surface waters. Rivers mobilize dissolved substances, suspended particles and organic matter from terrestrial systems to lakes (Allan, 1995).

In a drainage basin scale, downstream movements of water from high elevation mountains, through settled and agricultural land, link the terrestrial and aquatic ecosystems resulting in a net downstream movement of materials (Allan, 1995). Adverse development in catchments, often remote from the receiving water body itself, are widely recognized as a major contributor to degradation in lake quality (Everard *et al*., 2002). This is a major concern in a lake like Lake Nakuru whose simple ecosystem Vareschi (1979); can easily be shifted into a new equilibrium that may not be sustainable

The importance of the catchment area to inputs of both particulate and especially dissolved organic and inorganic matter to lakes, and of their regulatory influence on the metabolism and productivity of lakes are gaining appreciation and understanding world wide Wetzel (2001). For example, suspended solids from terrestrial systems are known to be good scavengers for heavy metals Arruda *et al.,* (1988) and are likely to be transported together from industrial sources and solid waste dumping sites. For small tropical lakes, rivers form complex interfaces with terrestrial systems and their effects on recipient lakes are usually fast (Kitaka *et al*., 2002a). Indeed sediments have caused the lake to decrease in size. The lake, which covered 48km^2, has shrunk to 38km^2 in 30years (Kuloba, 2004, pers. comm.).

Lake Nakuru is bordered by the fast growing Nakuru town to its northern most shore and its catchments are located in high potential agricultural areas which are prone to high rates of soil erosion Karanja *et al*., (1995). In their study along River Njoro, Karanja *et al.* (1995) observed that there was greater suspended sediment concentration on the rainy day than on the dry day.

Cultivation on fragmented land in Njoro River catchment appeared to lower the flow rate and increase the suspended-sediment concentration in the river Karanja *et al.,* (1995). Similarly, higher turbidities at mouths of rivers, draining into Lake Victoria, were mainly due to suspended solids, which were a consequence of increased soil erosion and other watershed disturbances Lung'ayia *et al., (2001*). The different rivers and the sewage drain studied represent different land uses and hence a good comparison of loading of suspended solids to the lake. It is against this background that the proposed study was conceived.

Materials and methods

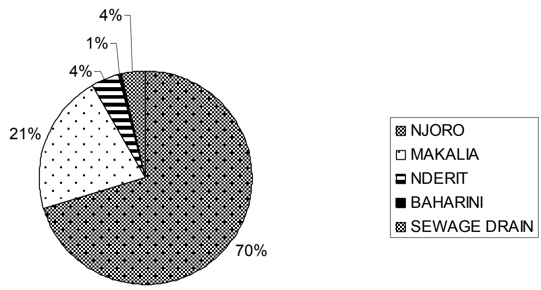
Rivers Njoro, Makalia, Nderit, Baharini spring brook and the Sewage drain were compared in this study. Sampling sites were located at the river mouth of each river, just before it enters the lake. Water samples were collected on a monthly basis for analyses of total suspended solids in the laboratory and velocity measurements taken using a Valeport (BFM001) current meter in addition to the width and the depth of the water. These were the parameters that were used to calculate discharge (m^3/s). Laboratory analyses produced total suspended solids (mg/l). Discharge measurements were converted to correspond to a days (m^3/day) loading while total suspended solids were converted to g/m^3. Multiplication of the two resulted to loading per day which was later converted to per year for each river.

Data used was for the years 2003 and 2004. The data was summarized and a pie chart showing percentage contribution of each river's loading of total suspended solids per year resulted.

Results

River Njoro was found to load most suspended solids (70%) to Lake Nakuru followed by Makalia (21%), Nderit (4%) and Sewage Drain (4%) and finally Baharini spring brook (1%) in that order as shown in Figure 1.

Figure 1. Pie chart showing percentage of total suspended solids loading to Lake Nakuru per year



Discussions

River Njoro may be delivering a higher percentage of suspended solids to Lake Nakuru than the other water channels because first, it flows almost throughout the year although there have been cases when it has failed to deliver water to the lake. Secondly it drains a large catchment area that is overly dominated by agricultural activities and livestock grazing and these are major contributors of suspended solids due to erosion associated with these land uses. Mau forest from which the river originates has had trees cut down so much and settlement and agricultural activities adopted further increasing erosion and hence suspended solids to the river and consequently the lake.

River Makalia has the second highest percentage of suspended solids to the lake since it also carries a high volume of water especially when it rains and with it comes a great load of suspended solids due to erosion and the soil that will have accumulated in the channel during the dry spell. It also traverses areas of small-scale agriculture and therefore carries a high sediment load.

River Nderit and sewage drain are the third largest contributors of total suspended solids per year. In the case of River Nderit this is due to its intermittent flow and when it flows, very little water gets to the lake despite the fact that it has a high concentration of sediments. The sewage drain has low contribution of total suspended solids because the water passes through the treatment plant where most of the suspended materials settle during treatment hence a lower load of solids although during storms it delivers slightly more and that is the reason why it equals River Nderit in contribution. River Nderit does not flow throughout the year. If both the sewage drain and River Nderit were to flow throughout the year then River Nderit would contribute more suspended solids than the sewage drain.

Baharini spring brook contributed the least proportion of the suspended solids mainly because it originates from within Lake Nakuru National Park and therefore does not pass through disturbed areas prone to erosion and hence has very little chance of collecting a high load of suspended solids. Baharini is a permanent, clean, low gradient spring brook with a steady discharge arising from slightly alkaline Baharini Springs on the northern shoreline of Lake Nakuru.

We can therefore conclude that more effort in soil conservation measures should be concentrated in the River Njoro watershed since this is the biggest threat to Lake Nakuru in terms of total suspended solids and consequently its decrease in size.

Acknowledgements

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References

Allan, J. D.,1995. Stream Ecology: Structure and Function of Running Waters. Chapman and Hall, London.

Arruda, J.A., Cringan, M.S., Layher, W.G., Kersh, G. & Bever, C., 1988 Pesticides in fish tissues and water from Turtle Creek Lake, Kansa. *Bull. Environ. Contamin. Toxicology.* 41: 617

Everard, M. Vale, J.A., Harper, D.M. & Tarras-Wahlberg, 2002. The physical attributes of the Lake Naivasha Catchment Rivers. *Hydrobiologia* 488: 13-25.

Lung'ayia, H., Sitoki, L. & Kenyanya, M., 2001. The nutrient enrichment of Lake Victoria (Kenyan Waters) *Hydrobiologia* 458: 75-82.

Karanja A.K., China S.S. & Kundu P., 1995 The influence of land use on Njoro River catchment between 1975 and 1985, pp 20-29. *Proceedings of a workshop on Use of Research Findings in the Management and Conservation of Biodiversity: A case of Lake Nakuru National Park, Kenya, 4-8 December, 1994.*

Kitaka, N., Harper, D.M., Mavuti, K.M. & Pacini, N., 2002a. Chemical characteristics, with particular reference to phosphorus, of rivers draining into Lake Naivasha, Kenya. *Hydrobiologia* 488: 57-71

Vareschi, E., 1979. The ecology of Lake Nakuru (Kenya) II. Biomass and spatial distribution of fish (Tilapia grahami Boulenger=Sarotherodon alcalicus grahami Boulenger). *Oecologia,* 37: 321-335.

Wetzel, R.G., 2001. *Limnology: Lakes and Rivers Ecosystems*. Academic Press, London.

Lake level and area variations 1960 to 2002 in Lake Kyoga, Uganda

Thomas Gumbricht

Independent researcher. Birger Jarlsgatan 102 b, 3tr, 11420 Stockholm, Sweden
Email:thomas_gumbricht@yahoo.se

Abstract

Lake Kyoga is a shallow lake in central Uganda of 3 to 5 m depth forming part of the Equatorial lake system. The lake is around 3000 km^2 and the local basin (57,000 km^2) drains large parts of Uganda. However, more than 90 % of the inflow originates from the Victoria Nile. The 1997/98 El Niño flood displaced hundreds of families and dislocated sudds that formed a plug at the outlet which rose the lake level almost 2 meters. Subsequent high flood levels in 2000 destroyed several settlements and led to the drowning of many people. The increase of the lake surface has meant loss of substantial areas of fertile soil, but is also reported by the local communities to have led to higher fish production. In this study, time series of Landsat MSS and (E)TM satellite images has been used to trace recent changes in the area of Lake Kyoga, and compare the flooding data with water levels.

Key words: Lake Kyoga

Introduction

Lake Kyoga is a shallow lake of 3 to 5 m depth forming part of the Equatorial lake system (Figure 1). The lake is around 3000 km^2 and drains large parts of Uganda. The local basin draining directly into Lake Kyoga is 57,000 km^2. However, more than 90 % of the inflow originates from Lake Victoria and the Victoria Nile. The massive swamps at the mouth of the tributary rivers provide a natural regulation of tributary inflows.

The 1997/98 El Niño flood displaced hundreds of families and dislocated sudds (floating mats of papyrus) that formed a plug at the outlet which rose the lake level almost 2 meters. Subsequent high flood levels in 2000 destroyed several settlements and led to the drowning of many people (Bird and Shinyekwa, 2003). The increase of the lake surface has meant loss of substantial areas of fertile soil, but is also reported by the local communities to have led to higher fish production. Bottom dwelling mud or lung fish species (e.g. *Protopterus aethiopicus*) of both commercial and subsistence importance has increased as reported by local fishers.

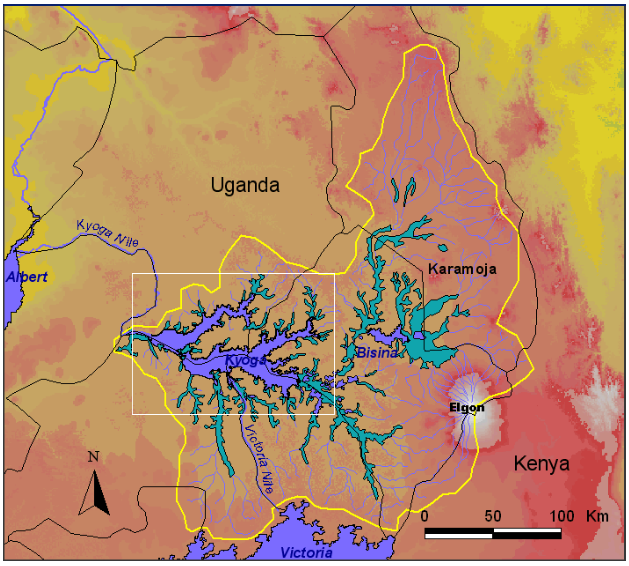


Figure 1. Lake Kyoga and its basin showing the lakes, major streams and wetlands. White box indicate area of Figure 2.

In this study time series of satellite images, mainly Landsat Multi Spectral Scanner (MSS), Themat Mapper (TM) and Enhanced TM (ETM) were used to trace recent changes in the area of Lake Kyoga, and compare this flooding data with historical water levels.

Materials and methods

48 medium to high resolution satellite images from the period 1972 to 2003 were used for mapping the area of Lake Kyoga and its transient changes over this period (table 1) The lion's share of the data are freely available quicklook images from the Landsat

program. These images are available for download as color composites, where water and cloud are easy to detect. This data was complemented with 2 radar composite scenes (JERS) from 1996. Additionally one older Corona ("spy") image was used to confirm the lake area in 1963. No radiometric correction was undertaken. All images were resampled to 250 m resolution and geocorrected to Universal Transverse Mercator (UTM) zone 36. The geocorrection was manually done using ground control points collected by handheld GPS as reference. To create precise geocorrections images were combined to animations, and iteratively adjusted until the animations contained no "jumps".

Table 1. Quicklook satellite images and radar data used in this study.

Source	Dates	Comments
Corona	1963-10-29	Not used in analysis
Landsat MSS	1972-12-29, 1973-03-03, 1974-01-29, 1976-01-27	All images combined to one estimate
Landsat TM	1984-04-26,1984-06-13,1984-07-31, 1984-09-01, 1985-04-13, 1986-01-10, 1986-11-10, 1986-12-12, 1989-02-11, 1994-10-31, 1994-12-02, 1995-01-19, 1995-03-08, 1995-04-09	
Landsat ETM	1999-10-05, 1999-12-05, 2000-01-25, 2000-04-14, 2000-05-16, 2000-08-04, 2000-10-23, 2000-11-24, 2001-04-17, 2001-05-19, 2001-07-06, 2001-09-08, 2001-11-27, 2002-01-30, 2002-03-19, 2002-04-04, 2002-05-22, 2002-06-07, 2002-07-08, 2002-08-10, 2002-09-27, 2002-10-13, 2003-01-17, 2003-03-22,2003-05-25,2003-08-07	
JERS-SAR	January to March 1996, October to November 1996.	Composites Undisturbed by clouds

For all images a statistical unsupervised classification technique was used to identify clouds, water, wetland and land areas (McCarthy et al., 2003). The derived classes were manually categorized into cloud, water, wetland or land by comparing the classified images with the downloaded color composite images (Figure 2). After identification the clouds were expanded by 2.5 km in all directions to accommodate for both thin clouds and cloud shadows. All images were then stacked and water occurrence in cloud free areas was estimated. If this occurrence was more than 95 % *and* the radar images (which are not disturbed by clouds) showed water, then that picture element (pixel) was considered to always be water. Clouds over those areas thus identified were hence changed into water in all images. Using the stacked information on water occurrence the maximum extent of the Lake and its riparian mixed land-water pixels and adjacent wetlands was estimated by using the identified core water area and a growth routine to identify all "wet" areas directly linked to the core area. For each individual image, the occurrence of water under clouds in the fringe area between the lake core and maximum extent area was resolved by using adjacent dates.

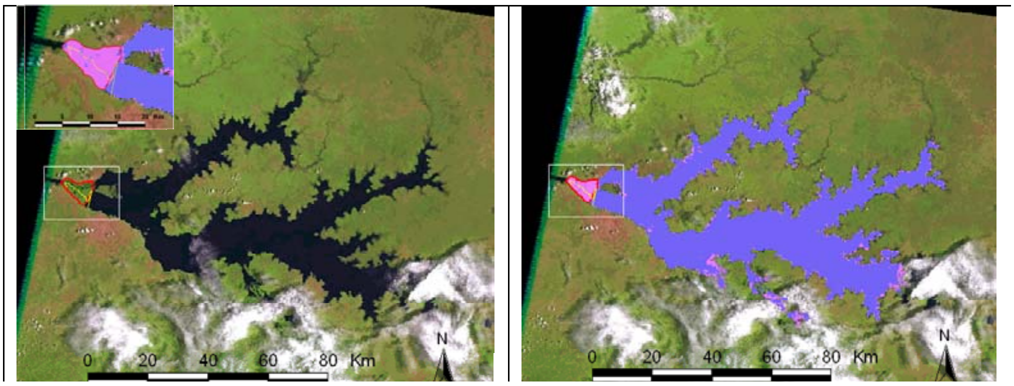


Figure 2. Image classification method exemplified by Landsat ETM from October 2002; the red area indicates the plug that formed after the 1997/98 El Niño. The yellow line is the survey line from November 2002 (see text).

Available "ground truth" data included GPS tracks and waypoints from six field visits in the period November 2002 to May 2004. A boat and flight survey conducted 18-19 November 2002 showed that the accuracy for a 5 km transect along different shorelines was 100 % when compared to water and wetlands classified from the Landsat ETM scene from October 2002 (See Figure 2). Also the tracks and waypoints from other visits indicate a high classification accuracy.

Two Landsat MSS, two Landsat TM and one Landsat ETM scene were acquired in full resolutions. The classification accuracy of the reduced resolution images from the same scenes was evaluated (table 2).

Table 2. Classification accuracy in 250 m resolution images as compared to full resolution images (30 to 60 m).

Date and sensor	Full resolution Water area (km^2)	Low resolution Water area (km^2)
1973-02-02/1974-01-29 (MSS)	2687	2702
1986 (TM)	2628	2574
1995 (TM)	2491	2492
2001 (ETM)	2744	2655

Data on relative water stages in Lake Kyoga was taken from the Department of Water Development recorder in the Lake. This dataset has large gaps, and in cases where the mismatch between available recorded water stages and dates of satellite images was less than 1 month, the recorded data was interpolated, or in a few cases extrapolated if stable conditions prevailed in the adjacent record.

Results

The area of lake Kyoga has varied between 2400 and 2850 km^2, or almost 20 % over the last 3 decades. From the water level recordings it is evident that the Lake level has varied around 3 meters, and that the lowest recording is from 1961, before the well known rise in the Lake level of Lake Victoria. This rise in 1961 is also clearly reflected in the levels at Lake Kyoga (Figure 3). The Corona satellite image over Lake Kyoga confirms the Lake shore, but large parts of the Lake in covered in clouds in the images.

The recorded changes in Lake levels and the open water area as interpreted from satellite images show a high degree of correlation (Figure 3), with an R^2 of 77 %.

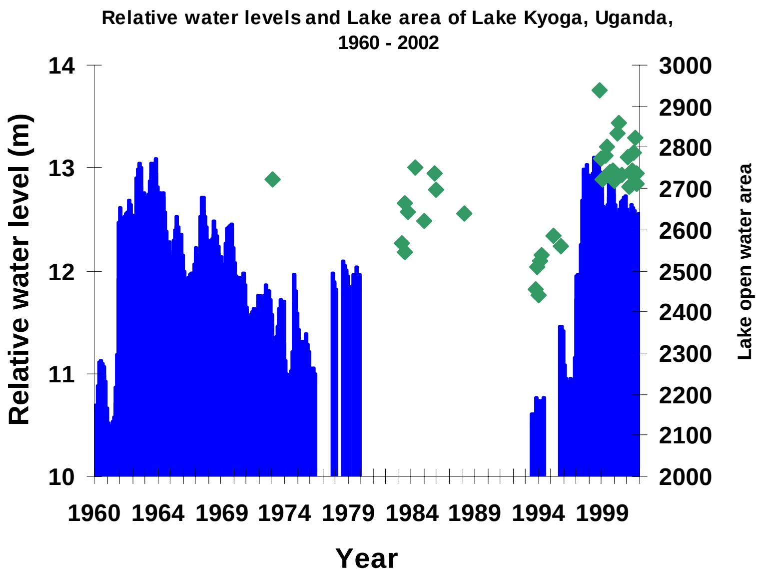


Figure 3, Lake Kyoga recorded water levels 1960 to 2002 (blue columns), and lake open water area (green rectangles) as interpreted from 48 satellite images 1973 to 2003.

Discussion

Lake Kyoga has experienced large variations in water levels as well as in surface water area over the last 40 years. An abrupt rise in 1961 coincides with the well-known rise in Lake Victoria water levels. The 1997/98 rise and prolonged high water level was caused by a plug of papyrus formed following a high precipitation period related to an El Niño event.

Following the rise in water level the local fishermen report improved catches, This increase in yield is, however not reflected in the official statistics (Allison et al., 2003). It has been suggested that a long term decrease in fish yields from Lake Kyoga over the last 20 years is due to over-fishing, wetland conversion and aquatic weed invasion (see Ogutu-Ohwayo, 1990; Allison, 2003). Local fishers tend to favor the latter explanation (weed invasion e.g. entangling and suffocating fish). They also attribute a collapse of fish yield in nearby Lake Bisina (see Figure 1) to invasive submersed weeds (personal communication with officials in several villages).

Conclusion

Lake Kyoga in central Uganda has experienced large variations and a general increasing trend in water levels and water area over the last decades. Compared to the early 1960's the Lake water volume is now almost double.

References

Allison, E., 2003. *Linking national fisheries policy to livelihoods on the shores of Lake Kyoga, Uganda*. LADDER working paper No. 9. 43 pp.

Bird, K., and I. Shinyekwa, 2003. *Chronic Poverty in rural Uganda: Initial Findings*. Nairobi, Kenya: Rural Livelihoods and Poverty Reduction Policies Conference, 13-14 January 2003.

McCarthy, J., T. Gumbricht, T. McCarthy, P. Frost, K. Wessels and F. Seidl, 2003. *Flooding dynamics of the Okavango wetland in Botswana between 1972 and 2000.* Ambio, 32: 453-457.

Ogutu-Ohwayo, R., 1990. The decline of the native fish species of Lake Kyoga and Kyoga (East Africa) and the impact of the introduced species, especially the Nile perch, Lates niloticus, and the Nile Tilapia, *Oreochromis niloticus*, *Environ. Biol Fish.*, 27: 81-96.

ECOHYDROLOGY - needs and opportunities in Africa. The Lake Naivasha Demonstration Site

David Harper[1], Maciej Zalewski[2], Kenneth Mavuti[3]

[1]Department of Biology, University of Leicester, LE1 7RH, United Kingdom
[2]International Centre for Ecology, Polish Academy of Sciences, Lodz and Department of Applied Ecology , University of Lodz, Poland
[3]Department of Zoology, University of Nairobi, Box 30197, Kenya

Abstract

Ecohydrology is a growing paradigm that is integrating the physical and biological aquatic sciences at catchment scale. Many African lakes and basins are severely degraded by dam construction, over-abstraction of water, erosion from inappropriate land uses or direct pollution. The worst effects are now recognised and mitigation measures are being proposed and implemented. These can be strengthened by an ecohydrological approach. Several case studies will be examined from this perspective.

Introduction

It is now well-understood that water quantity and quality decline is occurring on a global scale, severely reducing the biodiversity, cultural and socio-economic potential of many countries, especially in the tropical world. The time that mankind has to rectify this situation is becoming increasingly limited.

Four main hazards exist in freshwater and coastal ecosystems: 1) inorganic and non-natural organic chemical point-source pollution, 2) naturally-derived organic (human and livestock) point-source pollution 3) diffuse nutrient enrichment and 4) physical degradation. The first two are well defined, so their elimination has been and is, to a great extent, dependent upon financial potential and proper hydro-technical management, and the problem solved in more developed countries. The latter two hazards usually occur at a catchment scale and were (are) not seen as problems where rivers lakes and coastal waters were (are) badly polluted by the first two.

The third and fourth hazards are of increasing significance worldwide where serious pollution has been eliminated or did not exist, because they are proportional to human population density and the aspirations of societies. In many countries, particularly settled lowlands, the two hazards combine to lower the potential of river basins to effectively provide ecosystem services to mankind.

The direct causes of the loss of ecosystem services have been multidimensional factors such as loss of water retention in catchments due to forest cover degradation, improper river channelisation for flood control and land drainage, and soil erosion. A crucial question for the beginning of the 21st Century is how to solve the problem of such multidimensional and large-scale processes. This paper outlines a new paradigm – or way of analysis of freshwater resource problems – which has been developed in the temperate zones and is now being increasingly used in tropical countries.

Principles and values of Ecohydrology

The key to the achievement of sustainable water management is to use as a starting point our understanding of bio-geochemical processes, particularly the catchment-scale cycles of water and nutrients with special emphasis on their interactions with living communities. The understanding of interactions between living and non-living components of an ecosystem and between separate biological components is now referred to as “the Ecosystem Approach”, in aquatic ecosystems it can be termed the Ecohydrological Approach. Ecohydrology recognises the critical importance of interactions within the natural aquatic system and extends this concept to interactions between humans and the ecosystem. The ecohydrological approach goes beyond a mere understanding of the interactions, to use this understanding to enhance the ecosystem’s resistance and resilience against human impact. Aquatic ecosystems are situated in depressions in the landscape; for this reason all human activity within the catchment is reflected in basin water quality and quantity.

The degradation process of catchment ecosystems has a long history - initiated in Europe between 5000 and 10000 years ago, as deforestation by "slash and burn" techniques. About 2-1000 years ago the next intensive phase of catchment degradation started to appear, as river channels and floodplains were modified by the development of human settlement and agriculture. However the most dramatic acceleration occurred from about 200 years ago, when sewage systems generated high point-source pollution and surface run off occurred from large impermeable surfaces of human aggregations. That became exacerbated in the past 50 years by intensive agricultural methods which have both added diffuse nutrients to river basins and at the same time resulted in extensive river engineering to restrict the passage of flood waters onto floodplains.

Ecohydrology can be considered to be the integration of many strategies that had appeared as individual ecological management technologies since the 1970s, such as biomanipulation in lakes; ecotone enhancement and artificial wetland development along rivers and their floodplains (Fig 1). Thus, it is the logical successor to a number of ecological paradigms that have altered and deepened our understanding of river basin ecology (Figure 2.)

Collectively these individual techniques, and hence Ecohydrology, may be referred to as "low cost – high technology" solutions for sustainable river basin management, to minimise threats for human health and biodiversity, to maximise water availability and other ecosystem services for society. This is because the costs of elimination of a unit of damage, for example phosphorus, is one order of magnitude lower by ecohydrological methods than it is by technical – such as elimination in a sewage treatment plant in this example. One kilogramme of phosphorus provided to a reservoir would easily be transferred to 2 - 3 tons of algal biomass; on the other hand if 400 gm of phosphorus annually were converted to biomass of meadow or wetland plants and another 600 grams blocked in soil formation processes, the 1 kg of P could be eliminated annually by about 100 m^2 of trapping wetland (Koch 2001). In other words, it is enough to convert a 10 x 10 m plot of land in the floodplain as an artificial wetland to form a bio-filtration area that would reduce 2-3 tons of algae at a downstream lake or reservoir.

Technical water management in the 20th Century has been focused upon the elimination of dangers such as pollution or flood and drought. A successful strategy however, is composed of two elements – elimination of danger and the amplification of opportunities. The latter opportunities arise from the basin-scale use of ecohydrology, which is being demonstrated around the world (http://portal.unesco.org/en/ev.php-URL_ID=11157&URL_DO=DO_TOPIC&URL_SECTION=201.html).

The science of Ecohydrology

The principles of Ecohydrology are expressed in three sequential components:-

HYDROLOGICAL: The quantification of the hydrological cycle of a basin, should be a template for functional integration of hydrological and biological processes.

ECOLOGICAL: The integrated processes at river basin scale can be steered in such a way as to enhance the basin's carrying capacity and its ecosystem services.

ECOLOGICAL ENGINEERING: The regulation of hydrological and ecological processes, based on an integrative system approach, is thus a new tool for IWRM.

Their expression as testable hypotheses (Zalewski *et al.,* 1997) may be seen as:-

H1: Hydrological processes generally regulate biota

H2: Biota can be shaped as a tool to regulate hydrological processes

H3: These two types of regulations (H1&H2) can be integrated with hydro-technical infrastructure to achieve sustainable water and ecosystem services

Demonstration sites for Ecohydrology

In order to demonstrate the suitability of Ecohydrology for addressing many of the problems of sustainable basin management, information dissemination has been carried out thoroughly, an academic journal Ecohydrology & Hydrobiology has been established and a series of Technical Handbooks published by UNESCO, such as Harper & Zalewski (2001). Two practical publications for users have been published *Guidelines for the Integrated Management of the Watershed Phytotechnology* and *Ecohydrology Integrated Watershed Management - Ecohydrology and Phytotechnology Manual.* Both are available from http://www.unep.or.jp/ietc/Publications/

The Pilica River and Sulejow Reservoir, Poland have been declared the first Ecohydrology Demonstration sites by UNESCO, as a result of the research work over some two decades quantifying the measures. This programme of demonstration sites has been extended in 2005 to a further 15 sites around the World. The present paper outlines the only African lake site – Lake Naivasha, Kenya (see Mavuti and Harper, this volume, for a full description of the site).

Lake Naivasha originally – until the 1970s – had a most effective natural purification filter around its shores. This was a belt of *Cyperus papyrus* (papyrus), tens of metres thick along the lake edge and many square kilometres at the delta of the two permanent inflowing rivers. It was very well described in a series of papers at that time by the scientist who quantified the ecosystem services which papyrus provides – John Gaudet (Gaudet, 1975, 1976, 1977). These services were based upon the vegetation zonation from the upper historical water level to the level of the day, as follows:-

1. The highest former lake level, characterised by Fever tree (*Acacia xanthophloea).* Underneath that woody shrubs such as *Sesbania sesban* and *Cassia didyobotrys.*

2. On the lake-ward edge of the shrubs, grasses & sedges down to recently-exposed mud.

3. At the water fringe, rooted papyrus and other *Cyperus* species

4. On the lake floating papyrus islands

5. Between the islands and the lake edge, sheltered lagoons, with extensive beds of the blue water lily *Nymphaea nouchallii*.

6. Beds of various species of submerged plants down to about 5m depth in the lagoons.

7. The whole maintained a clear water (oligotrophic) lake by the physical and chemical recycling-release processes of these aquatic plant zones combined.

Naivasha became so well known that its virtues as an exemplary natural purification system have been described, as if still intact, up to the present time (Davis & Hirji, 2003), even though it has become so degraded over the past 20 years that the ecosystem functions of papyrus have been totally lost (Harper & Mavuti 2004, Mavuti & Harper, this volume)!

The degradation that has happened to these natural functions of the lake's riparian zone (and in a parallel fashion, the natural functions of the riparian zone of rivers in Naivasha'a catchment) has been caused by a number of human factors that have had an additive effect.

The worst is the extraction of water from the whole catchment, at higher levels than are naturally replenished by the hydrological cycle. Abstractions occur for five purposes:-

1. settled agriculture in the upper catchment;

2. water supply to the city of Nakuru from a small dam on the Turasha, main tributary of the Malewa;

3. the horticulture industry centred around Naivasha lakeshore;

4. One third of a million or so people associated with the hortucultural industry in the town of Naivasha and along the south lakeshore;

5. the geothermal power station at Olkaria, to the immediate south of the lake.

6. The combined abstractions have caused the lake to be about 3 vertical m lower than it would have been naturally (Becht & Harper, 2002). The consequences for this are shown by Mavuti & Harper (in this volume).

The second factor is that, within the lake, the native plant communities have been eliminated by the introduced Louisiana crayfish, *Procambarus clarkii* (Harper et al, this volume). There is now an absence of any biological or physical complexity below the water line, which formerly assisted the riparian plant zones in maintaining the lake's oligotrophic status.

These two factors combined mean the lake now (2005) has no natural buffer on its edge or in its shallows against the sediment and nutrients which enter both from the permanent river inflows and the many temporary streams which flow in the wet season, especially those from the southern lake edges, where the greatest density of both horticultural enterprises and workers' settlements are located.

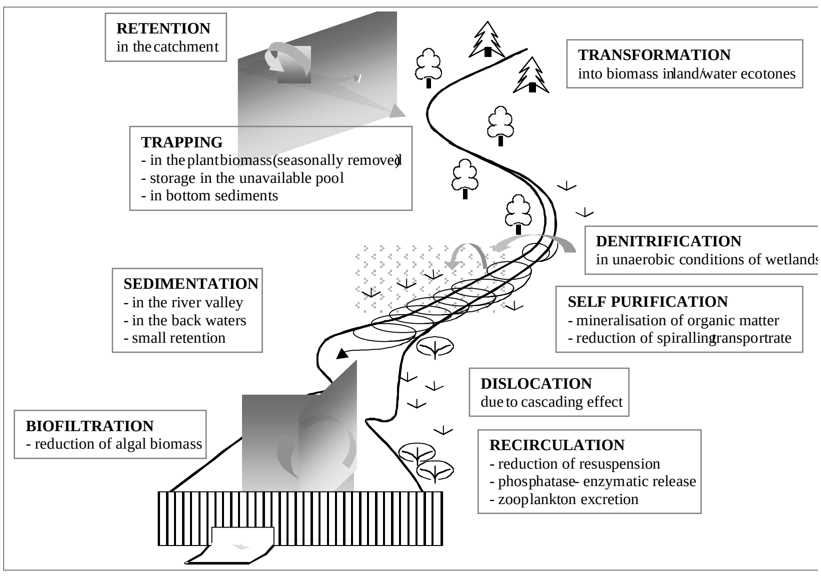


Figure 1. Examples of ecohydrological techniques in an hypothetical river basin (Zalewski, 2004).

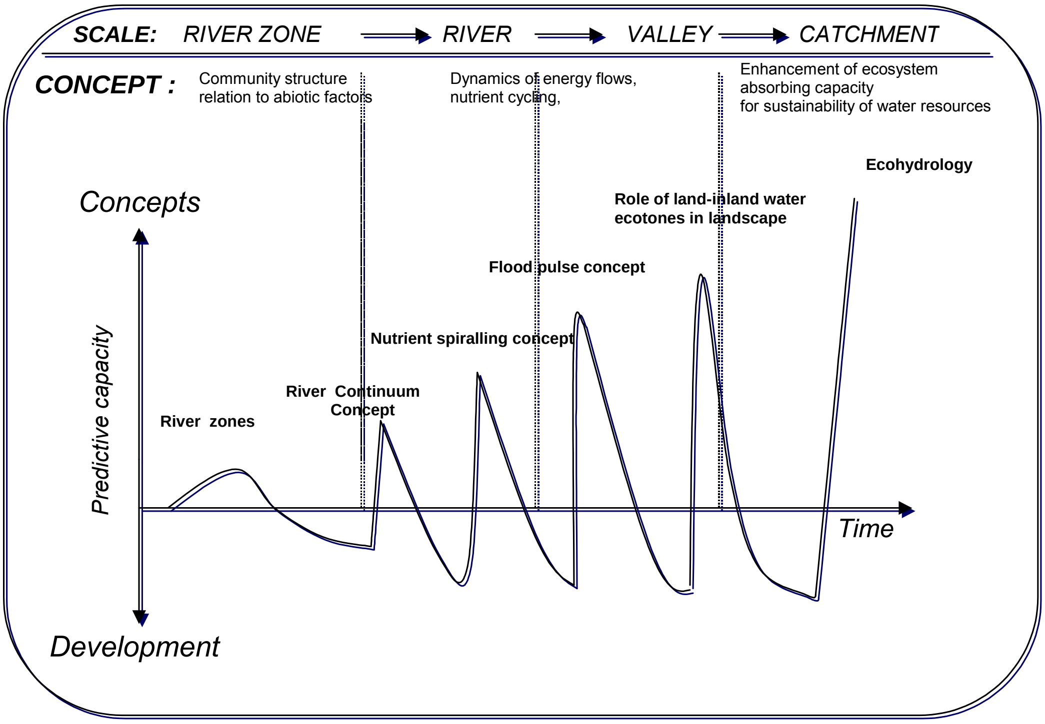


Figure 2. The concept of Ecohydrology showing its debt to preceding paradigms about river basins.

The selection of Lake Naivasha as an Ecohydrology Demonstration site, is based upon the scientific understanding of its ecosystem functioning and deterioration from naturalness, coupled with a belief that ecosystem functioning could me restored using this "low-cost, high-technology" method. The case for restoration of the ecological functioning of the lake is based upon three factors:-

1. A quantitative understanding of the differences between the water flowing through a papyrus fringe (Gaudet, 1977) and that not (Mavuti & Harper, this volume) and their consequences.

2. The visual impact of new deltas in the lake of gravel and sand sediment brought down by the temporary streams flowing through settlements and quarries entering the lake without vegetation buffer.

3. The visual impact of the eutrophication of the lake as severely reduced transparency.

4. Such restoration can only be an essential partner of the move towards abstraction of only the renewable water volume, rather than the present 'mining' of lake and groundwater.

5. This first proposal for restoring the lake is based upon of the feasibility of Ecohydrological techniques at three different scales:-

6. Restoration of an intact vegetated ecotone that will buffer the lake against sheet surface runoff during storm events (south-west side of the lake).

7. Restoration of an intact papyrus fringe and minor reconstruction of temporary stream channels, which will buffer the lake against intense sediment input during storm events (south-west side of the lake).

8. Creation of a sequential constructed wetland at the delta of the main inflowing river, the Malewa (north end of the lake), spilling water at three different discharge patterns. The lowest, at normal discharges, will spill more widely over riparian land to try to recreate a band of *C. papyrus* naturally at the lake edge. The second, at intermediate discharges, will seek to provide physical settlement for eroded sediment from the upper catchment, through riparian wet grassland. The third, at maximum discharges, will seek to recycle sediment and nutrients into riparian woodland, which could then be used for agro-forestry.

The second phase of developments of Lake Naivasha as an Ecohydrology Demonstration site will be to promote the message of ecohydrology through education at all levels from schools to universities. Already it is promoted in schools by short, locally-made videos and DVDs with the message '"Maji ni uhai" – Water is Life, in Swahili (see www.brockinitiative.org). Then it is to encourage wise use among adults – all of them, from illiterate farm workers to PhD horticultural experts, so that they understand the principles of the demonstration sites.

The third phase will be then to spread this educational campaign into the catchment, concurrently with extending demonstration sites up the riparian zone of the river Malewa, further incorporating principles of ecohydrology such as use of floodplain plantations of indigenous trees as fuel plots and new wetlands as human waste treatment facilities below rural towns.

References

Davis, R & R. Hirji, 2003. Water Resources and the Environment, *Technical Note G3 Wetlands Management.* World Bank, Washington USA, 28pp.

Gaudet J. J., 1975. Mineral concentrations in papyrus in various African swamps. *Journal of Ecology* **63,**483-491.

Gaudet J. J., 1976. Nutrient relationships in the Detritus of a tropical swamp. *Arch. Hydrobiol.* **78,** 213-239.

Gaudet, J. J., 1977. Natural drawdown on Lake Naivasha, Kenya and the formation of Papyrus swamps. *Aquatuc Botany* **3**: 1-47.

Harper, D.M. & K.M. Mavuti, 2004. Lake Naivasha, Kenya: Ecohydrology to guide the management of a tropical protected area. *Ecohydrology & Hydrobiology* **4**; 287-305

Harper, M. Zalewski ed.. 2001. Ecohydrology. Science and the sustainable management of tropical waters. A summary of the projects presented to the Conference Naivasha, Kenya, 11-16 April 1999. IHP UNESCO. *Tech. Doc. on Hydrology No 46,* Paris 2001

Koch E. 2001. *Tempo asymilacji fosforu przez roslinnosci zroznicowanych typologicznie obszarów zalewowych doliny Pilicy [Comparison of Nutrient De-position Rate in the Pilica River Floodplain of Different Typologies]*. Msc. Thesis. Department of Applied Ecology University of Łódź, Poland.

Zalewski, M. (2002) (Ed). *Guidelines for the Integrated Management of the Watershed – Phytotechnology & Ecohydrology.* United Nations Environment Programme Freshwater Management Series No. 5. 188pp, ISBN 92-807-2059-7.

Zalewski, M., Janauer, G.A. & Jolankai, G. 1997. Ecohydrology. A new paradigm for the sustainable use of aquatic resources. UNESCO IHP Technical Document in Hydrology No. 7.; IHP - V Projects 2.3/2.4, UNESCO Paris, 60 pp.

The question of urban environmental sustainability within lake Victoria basin: The case of Kisumu municipality, Kenya

Hayombe P. O.[1] V.A.O Odenyo[1]. G.P.Opata[1], Caleb Mireri[2]

[1]School of Environmental Studies, Moi University, Box 3900 Eldoret Kenya
[1]School of Environmental Studies (SES), Moi University, Box 3900 Eldoret
[2]Faculty Environmental Science, Kenyatta University

Abstract

City-Lake Interface as a concept exemplifies urban environmental sustainability concerns, where two ecosystems, the city and lake interact. Kisumu Municipality represents such an interface at the tip of Lake Victoria in Kenya. This paper discusses urban environmental sustainability, using the indicator of spatial distribution of uncollected solid wastes (garbage). The application of Global Positioning System (GPS) helped in locating and mapping of informal dumping sites and their weight in tones. F-test indicated that there are no significant differences among the neighbourhoods, especially informal settlements, which represents over 60% of the population. While household survey, indicated that 98% of respondents, in a sample of 376, confirmed that garbage was a major environmental concern. There are no appropriate solid waste management strategies. This scenario therefore is critical to a city-lake interface, as it aggravates lake water pollution hence environmental crisis. The paper elucidates solid waste monitoring, planning and management concerns.

Key words: City-Lake Interface, Kisumu Municipality, Urban Environmental Sustainability, Solid wastes, Garbage

Introduction

Environmental sustainability, especially in the rapidly urbanizing developing countries is a major concern. There is increasing need to find better ways of balancing the requirements and pressures of urban growth and change with the opportunities and constraints of local environmental resource base (UN-HABITAT/UNEP, 2003). Kisumu Municipality, the third largest town and one of the faster growing secondary towns in the Republic of Kenya, stands strategically placed on the gulf of Lake Victoria. Urban environmental sustainability of this town is in question. The City-Lake Interface as a concept of urban development helps to explore the sustainability concerns. This concept is embodied in the theoretical perspective of green social theories further explained by systems theory as an integrated development approach. Urban environmental sustainability is herein referred to as a balance of human activities in urban systems, with their environmental resource base (Ravetz, 2000).

The systems theory as applied in ecosystem approach in development elucidates the spatial interaction between two discernible ecosystems, namely, urban and the lake herein referred to us the City-Lake interface. The interface as exemplified in Kisumu Municipality demonstrates the urban environment crisis resulting from urban decay and environmental degradation as some of the challenges. Secondly, the interface displays some of the lost opportunities of urban development, eco-tourism and urban investment concerns.

City-Lake interfaces in developed world have received considerable planning and development interventions as they attract more investment opportunities (Hoyle and Pinder, 1992). However, in developing world due to inadequate development innovations and thus impulses, only pedestrian lifestyle and activities are attracted creating informal settlements and their environmental implications (Hayombe, 2004). Urban centres located within fragile ecosystems definitely pose great danger to the environment both in space and time. The urbanization process adjacent to the lake environment in the East African Community region has significant environmental impacts to Lake Victoria (LVI, 2002). These significant environmental impacts therefore raise the question of urban environmental sustainability among researchers and practitioners in the field of environmental planning and management.

Urban environmental sustainability is consequently compromised as environmental problems increase with no adequate intervention. These environmental sustainability indicators include massive uncollected solid waste (garbage), poor and inadequate environmental sanitation infrastructure (water and sanitation), water pollution, wetland encroachments and loss of Biodiversity. In addition, the ineffective institutional framework for environmental planning and management further aggravates the unsustainability of the city-lake interface.

This paper explores the urban environmental sustainability indicator using spatial distribution of uncollected solid wastes both at household level and informal dumping sites. The household survey comprised a sample of 376 respondents using questionnaire-interview method in the neighbourhoods of Kisumu Municipality. The spatial distributions of informal dumping sites were done through ground truthing using a Global Positioning System (GPS) in 200 sites in the five (5) main neighbourhood of Kisumu Municipality. Using triangulation method combining observation and GPS recording of uncollected tonnage of garbage was estimated as lorry load of seven (7) tonnes, in each informal dumping site.

The result discerns the urban environmental sustainability concerns, where huge uncollected garbage is present even within the Central Business District (CBD), as well as local authority managed houses. However, the problem is more significant in the informal settlements of Nyalenda, Manyatta and Obunga. The scenario is critical as over sixty percent (60%) of the urban population reside in these informal settlements. The urban environmental sustainability is in question as piles of garbage end up being washed into the lake ecosystem. The environmental imperatives are persistent lake water pollution resulting in loss of Biodiversity as well as deteriorating quality and quantity of water resources.

The challenge to have urban environmental sustainability of a city-lake interface within Lake Victoria, the second largest fresh water lake is undisputed. Rapid urbanization is real with increasing population in Kisumu and other urban centres within the interface gaining momentum. Without appropriate responses as drivers and pressures increase, the sustainability of the two ecosystems, the city and lake, is in question. Concerted efforts must be put in place to enhance urban environmental sustainability. The following research questions are addressed in this study: Is the spatial distribution of solid waste a significant environmental sustainability indicator in Kisumu Municipality? What is the extent and magnitude of solid waste problem? What solid waste coping strategies exist and how effective in management? What are the likely cumulative effects and their planning and management implications? And lastly what is the role of actors in solid waste management?

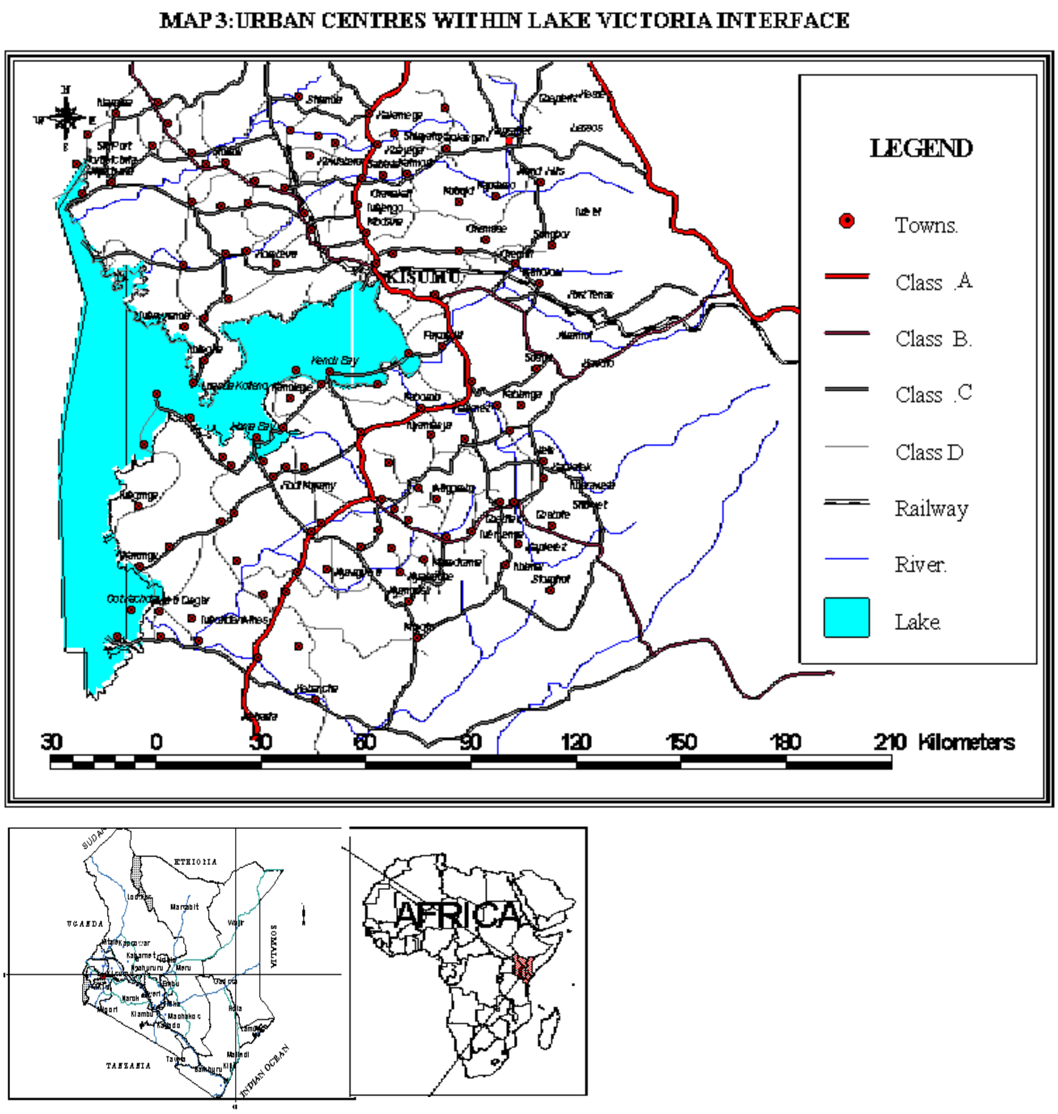


Figure 1.

The case study

This study focuses on urban development and their environmental planning and management implications in sustainable city-lake development. Kisumu Municipality is a case study used to demonstrate the city-lake ecosystem relationship. It is located on Winam Gulf of Lake Victoria, the second largest fresh water-lake in the world. It also represents a medium sized African city that befits the strategy, as if forms an interface with the lake. Evolution of Kisumu is mainly underpinned by its interaction with the lake. Apart from Kisumu there are a number of towns of equal status within the lake environs, comprising of Kampala, Entebbe and Jinja in the Republic of Uganda, while Musoma and Tabora are in the Republic of Tanzania. However, Kisumu is located on a unique configuration of Lake Victoria, the tip of the narrow Winam Gulf (Map1). The gulf presents a closed ecosystem upon which water pollution dispersion is minimal unlike rest of major urban centres with wider configuration. The Winam Gulf and Lake Victoria at large are currently facing environmental threats, in terms of water quality and ecosystems depletion, from the rapid growth of human settlements within the Lake Victoria at large region. Kisumu being a massive urban centre in the region contributes greater portion of these water pollutants. There are a number of other lower rank towns and fishing landing beaches also abutting the Lake Victoria, which with time will have critical environmental concerns to the lake. The spontaneous growth of these towns without proper land use plans and/or environmental management plans (EMP) is now a bigger challenge and a case to worry.

Administratively, Kisumu City is governed by the Municipal Council with the Major as the head of the council ad it is divided into eleven (11) locations and thirty-two (32) sub locations. The administrative boundary was covering an area of 20 sq.Km in 1971, which was then extended to the current 417 sq.km of which 157.km (35.5%) is covered by water. This extension has made it possible to divide Kisumu into three distinct physical morphologies, the old town, peri-urban and rural Kisumu. The current unofficial urban and peri-urban boundary stretches for about 600 sq.km (KCC, 2004), and resident population of over 400,000 and daytime of 800,000.

Methodology

This paper attempts to elucidate the concept of sustainable city-lake interface as an integrated ecosystem assessment approach in environmental planning and management. Secondly, the paper applied two research methodology namely, data collection and data analysis. The study of urban environmental sustainability of a city-lake interface is confined to spatial distribution of solid wastes in Kisumu Municipality.

Data collection

Data collection as alluded was done through the use of GPS recording informal dumping site and the magnitude of waste estimated. Secondly, household surveys, where a questionnaire-interview was conducted to explore the problem of garbage in the neighbourhood of Kisumu Municipality.

Global Position System (GPS) Survey

The spatial distribution of solid wastes was collected using triangulation method. Through a traverse, observation and estimation of magnitude of data was done during the month of August 2004. The GPS identified each site location parameters (latitude, longitude and altitude), for which magnitude of uncollected garbage was estimated in each informal dumping site. The estimation was calculated using a lorry load as seven (7) tonnes being the most convenient measurement. A total of 205 sites were systematically sampled along the main traverses in the entire Kisumu Municipality neighbourhood (Table 1, Map 2.)

Table 1: Global Positioning Systems (GPS) Informal Dumping Sites Sampled.

Neighbourhood	No. of Sites Observed	Percentage Representation	Category of Settlement
Nyalenda	56	27	INFORMAL
Bandani	42	20	INFORMAL
Manyatta	39	19	INFORMAL
Arina	39	19	FORMAL
Central Business District (CBD)	29	14	FORMAL
Total	205	100	

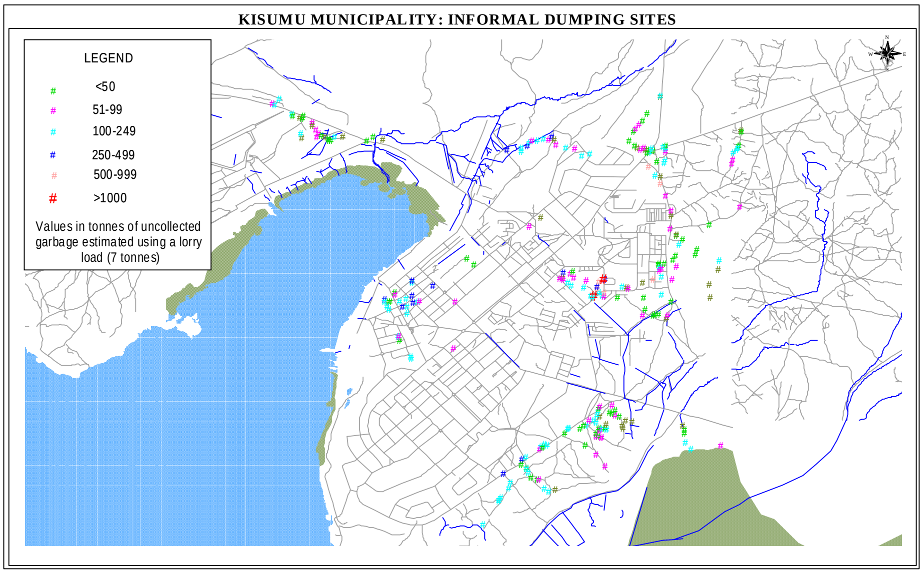


Figure 2.

Household questionnaire survey

The household survey was done during the month of June, July and August, 2004 where 376 respondents were interviewed using questionnaires in the Kisumu Municipality. The sample area is shown in Map 2. The core areas sampled include sixty percent (60%) informal settlements and 40% formal settlements (Table 2).

Table 2. Neighbourhood Household Survey.

Neighourhood (Sublocation)	No. of Respondents	Category of Settlement	Percentage Representation
Kaloleni	13	Informal	3.45
Kongony	7	Peri-urban	1.80
Manyatta A	90 (45/45)	Informal/ formal	24.0
Manyatta B	33	Informal/formal	9.00
Migosi	70	Formal	18.6
Nyalenda A	37	Informal	9.80
Nyalenda B	49	Informal	13.0
Nyawita	49	Informal	13.0
Southern	25	Formal	6.40
Total	**376**		**100**

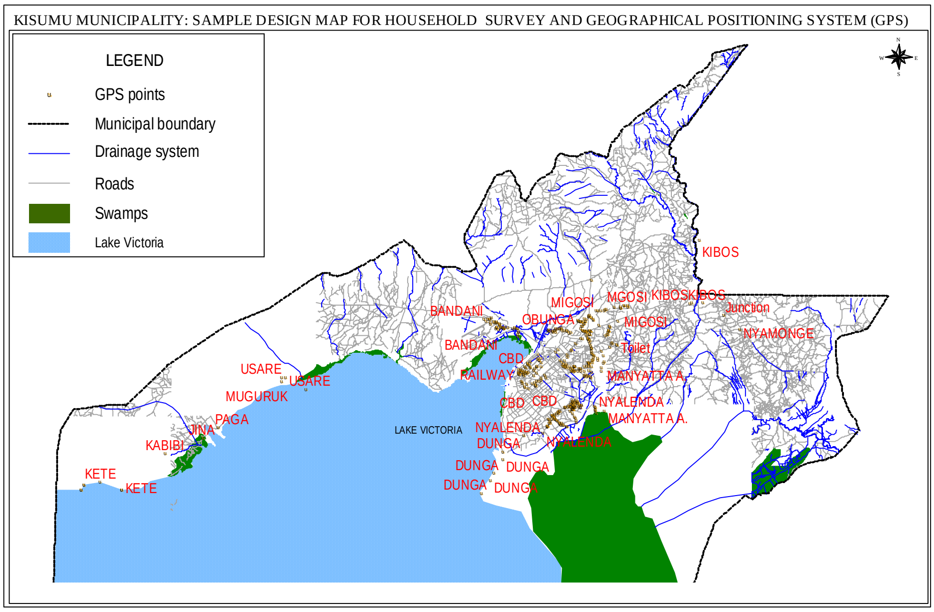


Figure 3.

Data analysis and presentation

Data analysis and presentation included spatial and graphical interpretation. The result was also mapped and presented as graphs (bar graphs and pie charts). In addition, the F-test was used as a statistical analysis to provide information on variation and differences within the selected representative neighbourhoods. The F-test was used to indicate if there are significant differences in variance between the means of the neighbourhoods (Table 3 and Table 4).

Conceptual framework for sustainable city-lake interface

Sustainable city-lake interface concept recognizes the fact that there is need for symbiotic relationship between the city and the lake ecosystem (Figure 1). This conceptual model is based on Green Social Theory, Systems Theory, and the Driver-pressure-State-Impact response (DIPSIR) model. Secondly, the concept incorporates the principles of eco-development postulated in the Rio-de Janerio Earth's Summit and later on in the World Summit on Sustainable Development in Johannesburg (2002). This strategy therefore is premised on Localizing Agenda 21 and Millennium Development Goals (2000). The strategy recognizes eco-development approach where two ecosystems, city and lake, should uphold ecological sustainability. Secondly, the notion that human beings are part and parcel of ecosystems and depend on the same for their survival, therefore the need for wise use of environmental resources while promoting social sustainability.

The Post-modernism theory reflects on a re-thinking of the development process, which considers both social and ecological sustainability. Green Social theory, as an emerging theory of postmodernism reorients the philosophical values in decision making where must be sought between society and the environment in the development process. Green social theory is therefore embedded in the emerging paradigm of sustainable development, basically referred to as eco-development embracing both ecological sustainability and social sustainability. However, the dynamics of urban development and environment nexus is realistically explained through a holistic approach in Systems theory. Recognizing two systems as the city and lake within an interface that co-exist and with varied linkages provides an appropriate integrated assessment. Environmental sustainability using indicators such as solid waste provide this integrated system assessment in systems theory, herein modeled as DIPSIR. Where drivers factors (D) of environmental degradation creates pressure (P) to the environment state (S) and promotes environmental impacts (I) as no appropriate responses (R) are put in place. This model explains the dynamics complex system of urban development and environment nexus and or referred to as the city-lake interface.

The city-lake interface is within two interacting ecosystems co-existing and abutting to each other, namely the city and the lake. Natural environment (land) and marine environment (lake) have co-existed abutting to each other for years in a sustainable and natural stable equilibrium. Minor disturbance and modification due to natural environment fluctuations have always reverted to the natural stable equilibrium. However, the introduction of human activities, namely; residential, commercial, industrial, recreational, agricultural starts to destabilize the natural equilibrium. More so the massive concentration of human pollution within city ecosystem accelerates the destabilization and degradation of fragile ecosystems, namely marine lake ecosystems.

The marine environments, namely the lake environment are fragile ecosystem, with variety of species of fauna and flora. The marine environment, however, has evolved and existed over the years as human beings have always converted and utilize its valuable resources both at subsistence and cash economy level. The marine environment, the lake, have attracted human settlement activities due to its resource value of water, fish, recreation and transport facilities, which in a small scale does minimal transformation to lake ecosystem. However, as massive concentration of human activity increase in the city coupled with unprecedented population growth, environmental degradation becomes a major concern. The human activities require land, energy, food, and shelter to be provided by the natural environment in time and space. The resulting interactions and linkages are the use of products and release of by-products to the environment. However, in absence of appropriate responses and institutional framework the outcome is depletion of environmental resources. This question of ecological and social sustainability within the city-lake interface, as a system, provides a basis for the theoretical and policy formulation of sustainable city-lake development strategy (Figure 1).

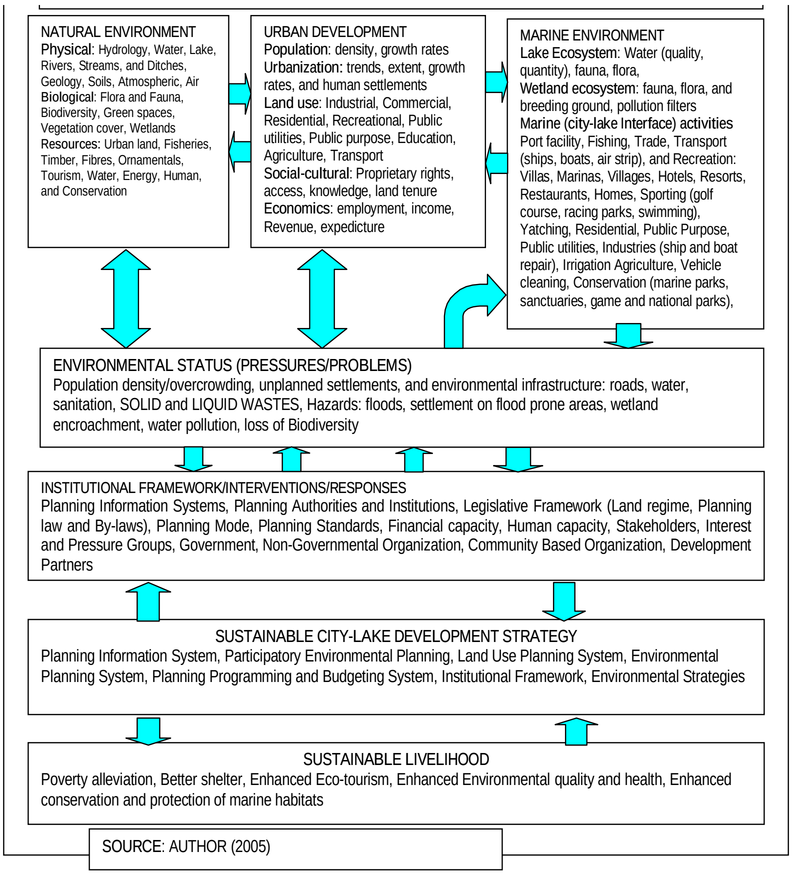


Figure 4 : Conceptual framework for sustainanble city-lake interface development.

Results

The result for garbage problem analysis is presented in twofold as GPS and household survey. Further analysis includes solid waste management illustrating agencies collecting garbage and coping strategies.

Global positioning system

The results include means of tonnage in the neighbourhood (Table 3). The F-Test of one tailed probability result is also presented (Table 4). The neighbourhood means indicate 88 tonnes and 135 tonnes as lowest (Nyalenda) and highest (CBD) respectively. The F-test indicates that there are no significant differences at 5% significance level between the neighbourhoods of Nyalenda and Bandani, Manyatta and Arina, Manyatta and CBD, Bandani and CBD, Arina and CBD. While there are significant differences between Nyalenda and Manyatta, Nyalenda and Arina, Nyalenda and Bandani, Manyatta and Bandani, Bandani and Arina. The significant differences exemplified in the other neighbourhoods could be due to sample fluctuations. The overall results of informal dumping sites are shown in Figure 1. The map discerns the spatial distribution with higher concentration found in areas with high density of population as Manyatta and Nyalenda.

Table 3. Global Positioning System (GPS) Informal Dumping Site Tonnage Mean and Standard Deviation.

Neighbourhood	*Mean*	*Standard Deviation*
Nyalenda	88	139
Bandani	98	146
Manyatta	120	196
Arina	124	158
CBD	135	144

Table 4. F-Test Results between Neighbourhoods.

Neigbourhood 1	Neighbourhood 2	F-Test	5%	Remarks
Nyalenda	Manyatta	4.86525	1.39	S
Nyalenda	Bandani	0.02268	1.51	NS
Nyalenda	Arina	2.7188	1.51	S
Nyalenda	CBD	1.5165	1.51	S
Manyatta	Bandani	9.95824	1.51	S
Manyatta	Arina	0.61419	1.62	NS
Manyatta	CBD	0.01275	1.62	NS
Bandani	Arina	3.8216	1.51	S
Bandani	CBD	0.0517	1.51	NS
Arina	CBD	0.00586	1.62	NS

*Note: S= Significant and NS= Not Significant (Significance at 5 %)

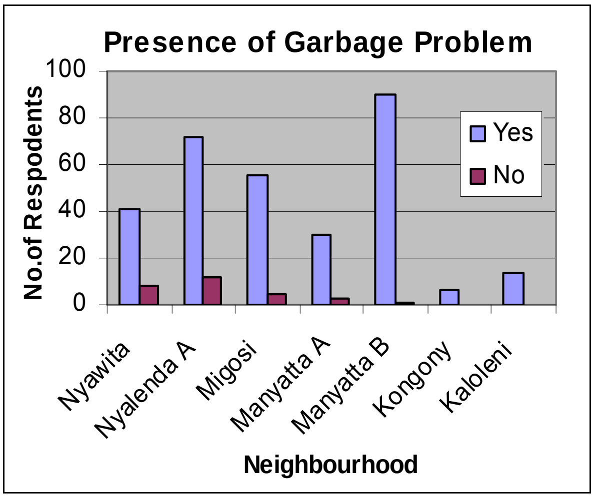
Household survey

The household survey results indicate that only 3.19% of respondents did not report the problem of solid wastes in their neighbourhood. This indicates that 96.8% of the respondents experience garbage problem. This result concurs with that of spatial informal dumping sites presented in Table 3 and Figure 1.

Further analysis for each neighbouhood illustrates the similar results, where garbage problem was reported as high. Except for southern area which is mainly high income and low-density comprising of Milimani Estate, the rest of neighbourhoods reported garbage problem. The problem is prevalent in the neighbourhoods of Nyawita, Nyalenda, Migosi, Manyatta A, Manyatta B, Kaloleni and Kongony (Table 5 and Figure 1). Over 80% of respondents reported garbage problem in their neighbourhood, an indication of non collection and/or inadequate appropriate solid waste management. This necessitated further examination of solid waste management.

Table 5 and Figure 5 (Bargragh): Garbage Problem: Percentage of Respondents Per Neighbourhood.

Location	Garbage Problem	No Problem	Percentage (%)
Nyawita	41	8	83
Nyalenda A	72	12	85
Migosi	55	5	91
Manyatta A	30	3	90
Manyatta B	90	1	98.8
Kongony	6	0	100
Kaloleni	14	0	100

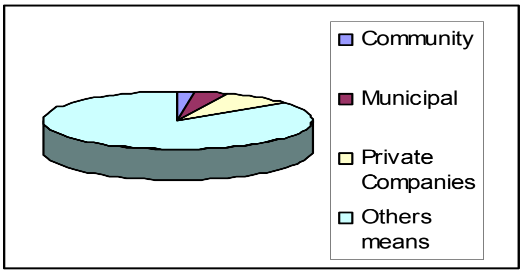


Solid waste management

Results of solid waste management indicate that very little garbage is collected from the neighbourhoods of Kisumu Municipality. Agencies that collect garbage, the community, municipal and private companies, only account for 15%, with other management options representing 85% (Table 6 and Figure 6). Examination of these management options indicate that most of the garbage is either dumped in the undefined (informal) sites or burned accounting for 46% and 42% respectively (Table 6). While the sale, recycling and use of solid waste as manure account for 10.4%.

Table 6. Collection of Garbage by Agencies and Coping Strategies for Solid Waste Management.

Who Collects	Respondents	Percentage	Coping Strategies	Respondents	Percentage
Community	7	1.8	Sale	12	3.18
Municipal	17	4.5	Recycle	13	3.44
Private Companies	33	8.7	Manure	14	3.71
Others means	330	88.0	Burning	163	42.0
			Dumping in undefined sited	174	46.0



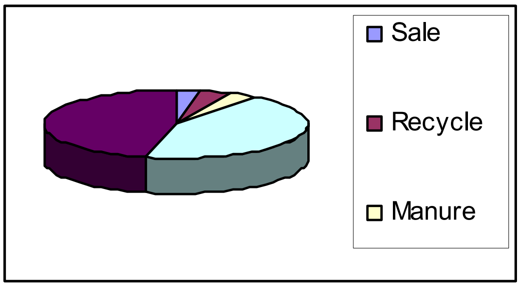


Figure 6. Pie Charts: Collection of Garbage by Agencies and Coping Strategies in Solid Waste Management.

Discussion

This section further explores the problem of solid wastes and brings on board the magnitude, factors and its implications to institutional management and environmental sustainable development of a city-lake interface.

GPS: Solid wastes

The F-test indicate that there are both no significant differences both at 5% significance level between five (5) neighbourhoods and significant differences in the other (5) neighbourhoods. This scenario nevertheless illustrates the problem of garbage in Kisumu Municipality despite the differences. Of importance to note is that the average tonnage in each neighbourhood is of great magnitude indicating uncollected solid wastes (Photographs A-F). The prevalence of informal dumping sites is a reality. And inadequate appropriate solid waste management put in place by the local authority and/or community initiatives is a major concern. Though solid wastes is now highly valued as secondary raw material, which can be recovered and reused (The East African, Weekly Paper, May 2-8, 2005), this initiative are still inadequate in Kisumu Municipality.

Critical analysis indicates that in the informal settlements of Bandani, Manyatta, Nyalenda, Nyawita, there are no significant differences in variation of uncollected garbage in terms of distribution. Noted is that formal settlements, except Arina, display no significant differences with the informal settlements. Explanation here is that the once planned housing settlements have degenerated into conditions of informal settlements and hence suffer the same predicaments. The Central Business District (CBD), although display results of uncollected garbage with a number of informal dumping sites has significant differences with other neighbourhood apart from Nyalenda. The CBD has low density of population, however informal dumping sites were observed on the service lanes, behind shops, though the magnitude and spread are not comparable to those of other neighbourhoods. The no significance difference between CBD and Nyalenda can be explained by sampling fluctuations.

The most critical conclusion is that informal dumping site is a phenomenon in Kisumu Municipality, with tonnage of uncollected garbage, lying up to one year and over without collection. The Daily Nation, Wednesday, May 11, 2005 reports: "Kisumu Bus Park has a real eyesore not far from where touts and operators frequently huddle, and next to a series of eateries, are heaps of uncollected garbage." This phenomenon of uncollected garbage and no appropriate mechanism of management are not only in the informal settlements but also in the formal settlements, which include the Central Business District (CBD). The CBD also comprise of the light industries, locally known as the "Jua Kali Sites", which has tremendously added a lot of uncollected garbage. This garbage is mainly non biodegradable including heavy metals, inorganic chemicals and oil spills. The expansion of unplanned Jua Kali industry is a major challenge as more unemployed persons continue to join the sector as the only viable alternative. Suffice is to say that informal settlements are rapidly expanding with no appropriate responses put towards solid waste management. The environmental implications of this phenomenon can not be gainsaid, especially within the city-lake interface. Inadequate intervention or response will definitely results in the cumulative environmental effects aggravating the environmental crisis.

Household survey: Solid wastes

Results indicate that higher percentage of respondent per neighbourhood reported environmental problem of solid wastes. Solid waste management in this neighbourhood hardly includes collection but other coping strategies, such as burning and dumping in undefined sites. Most of the formal and informal neighbourhoods experience the problem apart from the high income and low density areas, such as Southern (Milimani) area which mainly rely on private companies for garbage collection. In most towns in Kenya over 80% of garbage is not collected (Kibwage, 1996; Hayombe, 1997). KCC (2004) report that most of the solid waste generated in the city remains uncollected with collection efficiency estimated at 20%. Other means of solid waste handling represents 85%, an indication that garbage is either dumped in the neighbourhood or recycled. The Kisumu Municipal Council (KMC) as an institution has failed to cope with garbage menace. Explanations for this institutional failure may include inadequate capacity, technology and low revenue base to facilitate service delivery. Suffice it say that the "I don not care attitude" also cuts across both the community to be served and the institution that offer service. Attitude change is one first approach, through environmental education in order to realize meaningful solid waste management. Professor Ratemo Michieka reckons that "Ignorance of environmental issues is the main cause of environmental problems, poverty and unsustainable living" (Buigutt, 2004). However, the introduction of Local Authority Transfer Fund (LATF) by the Central Government in Kenya has not yielded much towards solid waste management. Focusing on domestic solid waste, one would expect more community initiatives to manage the garbage menace. However, the rate of informal dumping show that most resident do not see the problem as their own even from environmental health, aesthetic and psychological perspective. Environmental implications and urban environmental sustainability are compromised as local actions from actors, victims and managers are inadequate.

Environmental planning and management implications

This uncollected solid waste, both domestic and industrial wastes, gets washed by runoff to the, open drainage systems, both in the informal and formal neighbourhoods. These systems, however, include rivers Kisat and Kibos that traverse the town and drain into Lake Victoria. Both biodegradable and unbiodegradable substances, which include oil spills and heavy metals from 'jua kali' garages and domestic wastes from residential neighbourhoods pollute the lake water. This results in unsustainable development between the city and lake ecosystems. The city-lake interface development compromises urban environmental sustainability. The interface attracts more people as it offers development opportunities and gainful employment. These opportunities lead to expansion of human settlements, which is no exception in Kisumu-lake interface. Heavy pollution from these urban settlements due to increased garbage waste has been documented in various literatures. Water hyacinths formed as result of eutrophication and choking Lake Victoria are attributed to pollution from these urban solid wastes among others.

Urban environmental planning and management response should focus on these informal dumping sites, which residents now perceive as formal sites. The first response would be to spatially map these sites as demonstrated by GPS data-set (Map 3). Secondly, to isolate the sites for appropriate management measures, where accessibility should be provided and guaranteed. Third is to convert some of the sites as refuse collection interceptor sites, in order to tap the residents' perceived psychology. Fourth is to plan, establish and construct these sites as garbage collection during upgrading schemes in the informal settlement areas. The last intervention would be to identify the residents within each neighbourhood informal dumping site. And through a participatory process and civic engagement create a way forward, channeled through the sites focal local points. The application of Geographical Information Systems (GIS), using Global Positioning Systems (GPS) is therefore considered handy in environmental monitoring, planning and management of solid wastes. The GPS supplements the traditional questionnaire-interview method in identifying and offering solutions to reduce impacts of urban environmental sustainability indicators, such as solid wastes. This tool provides a mechanism for monitoring cumulative environmental impacts in City-Lake Interface.

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Kisumu City Development Strategy (CDS) 2004-2009 recognized some of positive solid waste management initiatives that exist in Kisumu as reuse and recycling. Industries and other players are being encouraged to adopt the 4Rs concept- recover, re-use, reduce and re-cycle (Kituyi and Wakhungu 2004; NEMA 2004). These small-scale initiatives include reuse and recycling of paper, plastic, organic waste, metals and water hyacinth, all providing micro-enterprise engagement for a significant number of inhabitants. An umbrella body, the Kisumu Recyclers and Collectors Association (KCORE) has been established to provide a common platform for negotiation and market access (KCC, 2004). Environmental awareness campaigns have also been enhanced to promote community and stakeholder participation in solid waste management, the year 2005 World Environmental Day was celebrated in Kisumu City, on 5th June, 2005.

The there are several actors in solid waste management in any urban centre; however, always the challenge is enormous with limited outcomes. These actors include the central government, the local government, the non-governmental organization (NGO), community based organization, private sectors, the residents among others. The central government, here represented by lead agencies such public health and lately, National Environment Management Authority (NEMA) are suppose to provide policies, guidelines and standards to regulate solid waste management operations. These broad guidelines are provided in the Public Health Act Cap 412 and the Environmental Management and Coordination Act (EMCA) no.8 of 1999. The local government, herein represented by Kisumu Municipal Council should provide more detailed operational regulative framework. This framework at the local neighbourhood level is referred to as Council By-Laws, which are supposed to be enforced to better outcome.

The failure of the two arms government both central and local is being filled by the uncoordinated structures of NGOs, mainly development partner driven. The NGOs strength is in mobilizing the local and attitude change through environmental education and awareness creation. Mobilizing local community is handy when existing CBO are ready to take up the challenge in collaboration with the government, central and local, and NGOs. Private Sector Partnership as a new concept, has now introduced emerging initiatives in solid waste management, where waste are now considered as raw material and input in various income generating activities. Environmental planning and management focusing on solid waste as an indicator of environmental sustainability must also consider the role the stakeholders outlined.

Conclusion

The urban environmental sustainability as exemplified by the indicator of solid waste is in question in Kisumu Municipality. This concern is compromising the city-lake interface development, as environmental implications pose challenges to urban planners and managers. The results provided critical concern, where the problem is not only confined to informal settlements but also formal settlements including the Central Business District. Local actions, both at community and institutional level, to promote sound solid waste management are inadequate. Concerted efforts to promote sustainable urban development within the city-lake interface would nevertheless require appropriate solid waste management.

References

Buigut K.S.A (2004): Universities Should Lead in Sustainable Development. NEMA News Magazine Vol.III- December, 2004 A quaterly publication of National Environmental Management Authority

Hardoy, J.E; D. Satterwaite (2001); Environmental Problems in an Urbanizing World. London. Earthscan Publications Ltd.

Hayombe P.O. (1997): Some Environmental Problems Related to Urban Sprawl in Southern Kasarani Division, Nairobi City: A Planning Perspective. Unpublished M.Phil. Thesis in Environmental Studies (Environmental Planning and Management Division), Moi University, Eldoret, Kenya

Hayombe P.O. (2004): Environmental Planning and Management Implications of a City-Lake Interface: A Case Study of Kisumu Municipality, Kenya. Unpublished Ph.D Proposal presented to School of Environmental Studies (Environmental Planning and Management Division), Moi University, Eldoret, Kenya

Hoyle and Pinder (1992): European Port Cities in Transition, Belhavev Press London, John Wiley & Sons, Inc. New York, pp.238-320

Kibwage C. (1996): Solid Waste Management in Nairobi City. Unpublished M.Phil.Thesis, School of Environmental Studies, Moi University, Kenya.

Kisumu City Council (KCC, 2004): Kisumu City Development Strategies (2004-2009): Lake Victoria Region City Development Strategies for Improved Urban Environment and Poverty Reduction. Courtesy of UN-HABITAT and SIDA.

Kisumu City Council (KCC, 2004): Kisumu City Development Strategies (2004-2009): Lake Region City Development Strategies for Improved Urban Environment and Poverty Reduction, Courtesy of UN-HABITAT and SIDA pp 25-26

Kituyi E. and Wakhungu J.w (2004): Harnessing Life Cycle Approaches for Africa's Development: Institutional and Policy Challenges in Ecopolicy 12. Africa Centre for technology Studies, Nairobi, Kenya pp 4-5

Lake Victoria Initiatives (LVI, 2002): Kisumu City council: An Indicative Situational Analysis for the Lake Victoria region/Municipal Development Strategies (CDS) for Improved Urban Environment and Poverty Reduction.

National Environment Management Authority (NEMA 2004): State of Environment Report 2003, Republic of Kenya pp 68-69

National Environment Management Authority (NEMA 2005): State of Environment Report, 2004 (Draft), Republic of Kenya

The Daily Nation, Wednesday 11, 2005

The East African, Weekly Magazine, May 2-8, 2005

UN-HABITAT/UNEP (2003): Environmentally Sustainable Urbanization. *Developing Environmental Planning and Management Capacities for Poverty Reduction*. Meeting Report of the 2003 Global Meeting of the Sustainable Cities Programme (SCP) and the Localizing Agenda 21 Programme (LA21), Alexandria, Egypt 29 September to 02 October 2003

Physicochemical factors and community structure of benthic macroinvertebrates along a gradient of changing land uses in the watershed in the upper reaches of river Njoro, Kenya

Samuel Kibichii[1], W.A. Shivoga[2], Mucai Muchiri[1], & Scot N. Miller[3]

[1.] Department of Fisheries, Moi University P.O. Box 1125, Eldoret, Kenya
[2.] Department of Environmental Science, Egerton University P.O. Box 536, Njoro, Kenya
[3] Department of Renewable Resources University of Wyoming, Department 3354, 1000 University Dr. Laramie, WY 82071

Abstract

Living organisms have for a long time been known and effectively used as indicators of ecosystem health. In this study, population structures of benthic macroinvertebrate taxa were used to asses impacts of land-use in the upper reaches of the River Njoro watershed on the ecological quality of the river. Benthic macroinvertebrates were sampled in triplicate at 11 sites in the river between April and August 2003. The sites corresponded to different watershed land uses and were: Natural forests (Runguma and Salt Spring); grazing (Logoman and Tiriytagoi); small-scale agriculture (Sigotik Upstream, Bridge and Downstream and Nessuit Bridge); intensive agriculture including floriculture (Forest camp, Highland Flowers and Confluence Upstream). From the samples, taxa abundance (%) and community structure were estimated for each sample. Physicochemical variables were also analyzed at site and from water samples obtained from each site. *Beatis* and simulidae composed 65% by number of all invertebrates collected. High and low mean diversities and richness were recorded at Runguma and Highland Flowers respectively. Among the upper most sites, high mean temperatures were recorded at grazed sites. Highest mean total P (0.24±0.01 mg/L) and total N (7.09±2.52 mg/L) were recorded at Confluence Upstream and Sigotik Downstream respectively. The two sites are in settled and cultivated areas of the watershed. Therefore, agriculture and grazing have modified the physico-chemical environment of River Njoro leading to a reduction in the diversity and evenness of distribution of the invertebrates. Consequently, seemingly “sensitive” taxa like the Fresh water crab (*Potamon sp.*), Helotidae, Lepidostomatidae and *Leptophlebia sp***.** are restricted to least impacted sites as compared to *Baetis sp*. and Simulidae which dominate in grazed and cultivated sites.

Key words: aquatic ecosystems, human influences, nutrients

Introduction

In recent years there have been calls for a broader, more holistic approach to aquatic ecosystem management (Yoder and Rankin, 1989) for the conservation of biodiversity, with emphasis on maintaining the ecological integrity of streams (Wells *et al.,* 2002). The character of a stream is determined by the character and magnitude of the basin it drains (Hesse *et al.*, 1989). Indeed, running water ecosystems are intimately tied to physical, chemical and biological processes that occur throughout the catchment (Gore and Shields, 1995).

An important aspect of this holistic approach is the need to use physical, chemical and biological measurements to characterize and monitor change in stream ecosystems (Wells *et al.,* 2002; Simon *et al.*, 2003; Yoder and DeShon, 2003). This is because physical and chemical measurements (used extensively in the past) are no longer considered sufficient for the protection of aquatic ecosystems (Carlisle *et al.*, 2003). Since streams integrate watershed characteristics and receive pollutants from land and the air, water quality measurements in affected streams are good indicators of changing environmental and ecosystem conditions in a watershed.

Biota is abundant in all types of aquatic systems, living on or in the substrates or in the water column, and therefore can be used to evaluate environmental health in all types of aquatic habitats (Jones *et al.*, 2002). The natural diversity within a system however, precludes the possibility of using every component of the biota. The most commonly used bio-indicators in streams are benthic macroinvertebrates, benthic biofilms (e.g. periphyton) and fish. These organisms have differing strengths as indicators, and periphyton and benthic invertebrates can be used as early warning systems for managing fisheries (Jones *et al.*, 2002). They can also be used in smaller systems that generally support diverse communities but only a limited or no fish fauna as is the case in River Njoro.

The ubiquitous and sedentary nature of benthic macroinvertebrates (Rosenberg and Resh, 1999), as well as their measurable responses to ambient conditions and exposure over time (Wells *et al.*, 2002) favour their use as important indicators in stream ecosystem monitoring. Because of the restricted mobility and habitat preferences associated with the benthos, they are subjected to the full rigour of their local environments (Jones *et al.*, 2002). Subsequently, they change in composition and distribution as environmental conditions change (Carlistle *et al.*, 2003).

Analytical procedures for biological assessments in streams fall into two main categories, univariate and multivariate approaches. Univariate approaches are characterized by the use of indices that express ecosystem health status as a single value (Jones *et al.*, 2002). Examples of common univariate techniques are simple abundance, richness, or biomass measures as well as diversity and biotic indices (Jones *et al.*, 2002; Simon *et al.*, 2003). Multimetric approaches involve the use of such measures as the index of biotic integrity, the reference condition approach, habitat survey indices and the index of community integrity (Simon *et al.*, 2003). These measures require stringent taxonomic resolution as well as knowledge on the sensitivity of identified taxa to pollutants. There is a paucity of publications on the taxonomy of stream biota in most tropical Africa. As such the need to identify benthic organisms to species level is not achievable in many cases. In addition, sensitivity values for Kenyan stream benthic invertebrates have never been estimated. Univariate approaches therefore are more feasible and were used in this study.

When observed patterns of macroinvertebrate community assemblages are related to offstream activities, e.g. land use and instream habitat conditions (using physico-chemical parameters), ecosystem effects of ecological stress can be discerned and remedial measures put in place. Regardless of how a range of human influences is selected among study sites, sampling at sites with different intensities and types of human activity is essential to detect and understand human influences (Simon, 2003). In this case sampling and analysis should concentrate on multiple sites within the same environmental setting (e.g within the same reach in a catchment) across a range of conditions from least impacted to severely disturbed as a result of human disturbance (Emery and Thomas, 2003). Sampling sites in this study were concentrated in the upper reaches of River Njoro only.

Land use in the River Njoro watershed has been changing since the beginning of the last century following large scale settlement on the middle reaches of the watershed by colonial farmers and later in the 1970's and 1980's by Kenyan farmers on former white settler farms and adjacent forest reserves in the upper reaches. The effects of these land uses on water quality and the ecosystem have never been investigated although an introduced trout has already disappeared from the river following the changes. This study sought to achieve 3 objectives along a gradient of changing land uses in the upper reaches of River Njoro:

Determine the physico-chemical conditions of water

Determine the community assemblages of benthic macroinvertebrates

Assess relationships between physicochemical parameters and the community assemblages.

Materials and methods

River Njoro (0^0 15', 0^0 25 S and 35^0 50', 35^0 05 E) drains a watershed of approximately 250 sq. km in Nakuru District, Kenya. It's 50 km long and consists mainly of Enjoro and Little Shuru tributaries. The river itself is structured in a typical pool-riffle sequence with predominantly soft substrata in pool areas and bedrock in riffle sections. six sampling sites were established on the main stem River, three on Little Shuru, and one each on Logoman and Runguma tributaries each corresponding to some particular offstream land use during a reconnaissance survey on 19th and 20th March 2003 (Figure 1; Table 1). Runguma and Salt Spring were located in forested parts of the watershed. Logoman and Tiriytagoi were established along grazing fields. Sigotik Upstream, Bridge and Downstream, and Nessuit were in areas experiencing small-scale agriculture. Forest Camp, Highland Flowers and Confluence Upstream were in downstream, intensively cultivated parts of the watershed.

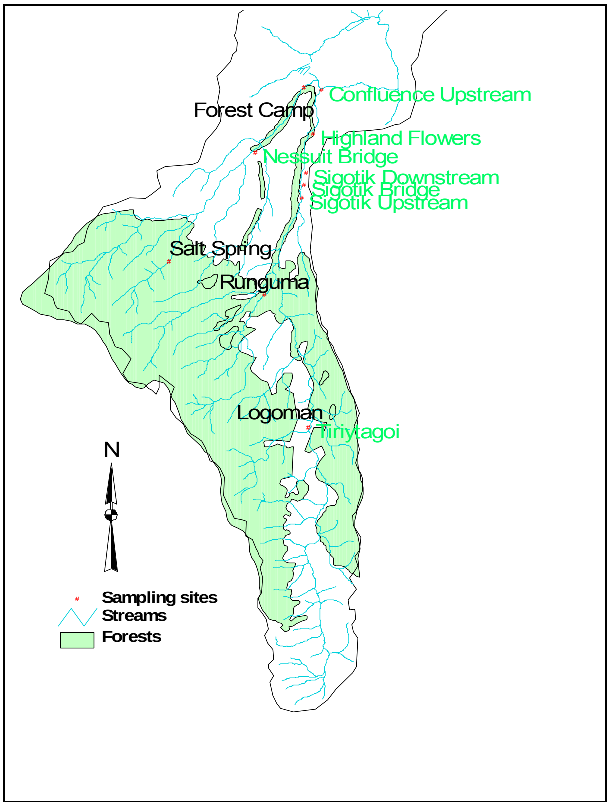

Figure 1. Map of the upper reaches of River Njoro watershed showing the location of sampling sites.

Table 1. Descriptions of sampling sites in the upper reaches of River Njoro.

Tributary	Sampling site	Altitude (metres above sea level)	Predominant Land Use	Condition of riparian zone
Enjoro	Tiriytagoi	2701	Intensive grazing	Cleared to the edge
	Sigotik Upstream	2417	Small-scale agriculture	Shaded but charcoal burning practised
	Sigotik Bridge	2408	Small-scale agriculture	Severely disturbed by people and livestock
	Sigotik Downstream	2402	Small-scale agriculture	Shaded
	Highland Flowers	2337	Intensive agriculture	Heavily shaded
	Confluence Upstream	2320	Intensive agriculture	Severely disturbed
Little Shuru	Salt Spring	2600	Indigenous bamboo and woody forests	Heavily shaded with a variety of woody vegetation and semi aquatic macrophytes
	Nessuit Bridge	2420	Small-scale agriculture	Shading interrupted by a watering point
	Forest Camp	2335	Intensive agriculture	Shaded
Runguma	Runguma	2660	indigenous forest	Heavily shaded with a variety of woody vegetation and semi aquatic macrophytes
Logoman	Logoman	2685	Intensive grazing	Severely disturbed

Sampling was done twice in April and May but once in June, July and August 2003. At each sampling site, temperature (^{o} C), pH, conductivity (µS/cm), dissolved oxygen (DO) saturation (%), and DO concentration (mg/L) were measured *in situ* using a JENWAY 3405 electrochemical analyzer after calibration. Total Phosphorus (mg/L), total Nitrogen (mg/L), Ammonium Nitrogen (NH_3–N; mg/L) nitrite Nitrogen (NO_2-N; mg/L), nitrate Nitrogen (NO_3^{-}N; mg/L), total suspended solids (TSS; mg/L) and total organic carbon (TOC; mg/L) were determined at the water quality testing laboratory in Lake Nakuru, using refrigerated water samples obtained from each site. All analyses were done following standard methods (APHA, 1998). Means (± standard errors) were calculated for each parameter per station. Analysis of variance (one way-ANOVA; Zar, 1996) was used to test for difference between sites for each physicochemical parameter. Duncan's multiple range tests were further used to determine the specific sites that were differing.

Benthic macroinvertebrate samples were obtained immediately after taking water samples using a modified Hess sampler (sampling area = 2.7 dm^2) in triplicate at each site. Pools, riffles and their transition areas were sampled to increase probability of encountering as many taxa as possible. Samples were fixed in 10% formaldehyde solution on site. They were later sorted in the laboratory under a stereomicroscope (LEICA ZOOM model 2000) and preserved in 70% ethanol. Observed organisms were later identified to the lowest taxonomic level within which they could be placed with certainty. To characterize sites, community attributes of total counts per sample at a site, density (individuals per dm^2) taxa numbers; richness (using margalef's index), diversity (using Shannon diversity index), evenness and dominance (Brower *et al.*, 1990) were estimated for each sample at a site. These measures were then correlated (Spearman Rank Order correlations; Zar, 1996) with physicochemical variables. All analyses were done using statistica for windows package (statsoft, 2001).

Results

Physico-chemical variables

Table 2 shows the physicochemical characteristics measured *in situ* at sampling sites in the upper reaches of River Njoro. On looking at spatial variations, there is an increase in temperature with decreasing altitude downstream. However, the sites on the main Enjoro tributary and Logoman had higher temperatures than those on Little Shuru and Runguma tributaries. The sites sampled at Tiriytagoi and Logoman had open canopies and sparsely vegetated banks occasioned by riparian grazing and logging to the water edge. The lowest and the highest mean water temperatures were recorded at Tiriytagoi, the uppermost sampling site and Confluence Upstream, the lowermost site respectively. Likewise, sites on Little Shuru showed a gradual increase in water temperatures downstream.

Table 2: Mean physico-chemical variables measured *in situ* at the sampled sites in the upper reaches of River Njoro between April and August 2004.

Tributary	Sampling site	Water Temperature (^{0}C)	Dissoved Oxygen (mg/l)	% oxygen saturation	Conductivity (µS/cm)	pH
Enjoro	Tiriytagoi	14.4±0.02	11.65±0.26	109.3 3.3	81.5±6.46	7.59±0.17
	Sigotik Upstream	15.7±0.4	12.07±0.18	121.4±2.7	105.3±12.09	7.0±0.06
	Sigotik Bridge	16.0±0.57	11.75±0.50	130.5±1.1	103.6±11.1	7.12±0.13
	Sigotik Downstream	15.8±0.6	11.6±0.51	129.2±1.3	108.7±13.65	7.2±0.13
	Highland Flowers	15.6±0.2	10.8±0.48	107.7±2.6	106.9±14.89	7.52±0.11
	Confluence Upstream	16.53±0.37	10.55±0.2	110.5±2.2	107.1±15.1	7.42±0.08
Little Shuru	Salt Spring	12.07±0.26	9.5±0.32	103.4±3.6	277±57.9	7.6±0.05
	Nessuit Bridge	12.2±0.1	11.8±0.54	118.1±2.1	125.7±23.3	7.2±0.06
	Forest Camp	14.1±0.6	11.49±0.61	118.2±2.5	136.5±25.1	7.7±0.5
Runguma	Runguma	13.12±0.12	10.97±0.29	110±5.5	109±15.2	7.1±0.06
Logoman	Logoman	15.1±0.4	9.33±0.3	105±6.5	95.6±5.8	7.1±0.1

The lowest mean temperature (12.07±0.26°C) was recorded at Salt Spring sampling site. This was significantly lower than that of Forest Camp ($p<0.05$). Mean temperature for Logoman tributary was significantly higher than those recorded at sites in Little Shuru tributary ($p<0.05$). Significantly higher mean temperatures were recorded during the dry season ($p=0.001$) for all sites during the study period except Runguma and Salt Spring where no differences were observed ($p>0.05$). Considerably higher mean temperatures were recorded at all sites during the second week of April at the end of the dry season. All sites on the four tributaries of River Njoro had high concentrations of oxygen (all above 100% saturation; Table 2) throughout the study period. However, DO concentration showed an inverse trend with temperature. Consequently, the mean dissolved oxygen (DO) concentration was lower at Tiriytagoi (11.65±0.26 mg/l), peaked at Sigotik Upstream (12.07±0.18 mg/l) but gradually dropped to its lowest at the Confluence Upstream site (10.55±0.2 mg/l) in the Enjoro tributary. One-way ANOVA test showed a significant difference in oxygen between the sampling sites along the Enjoro stream. However, Duncan's Multiple Range test showed that only Confluence Upstream and Highland Flowers had significantly lower oxygen values than all the other sites ($p=0.01$).

A similar trend in oxygen concentration levels was observed at sampling sites along the Little Shuru tributary. Mean dissolved oxygen concentrations for Logoman and Runguma were 9.33±0.3 mg/l and 10.97±0.29 mg/l, respectively. This indicates that dissolved oxygen is related to temperature and not land use in the upper reaches of River Njoro. There was no significant difference in dissolved oxygen concentration and percentage saturation between sampling occasions for all sites ($p>0.05$).

All sites along the Enjoro main tributary had lower conductivity values than the Little Shuru tributary on all sampling occasions. Conductivity increased downstream on the Enjoro stream with the lowest mean value (81.48±6.46 µS/cm) at the Tiriytagoi site and the highest (108.69 µS/cm) at Sigotik Downstream (Table 3), but there was no significant differences between sites on Enjoro ($p>0.05$).

Table 3: Summary of mean (±.S.E.) concentrations of physico-chemical variables from water samples collected from sampling sites in the upper reaches of River Njoro.

Tributary	Sampling site	NH_4-N (mg/l)	NO_2-N (mg/l)	NO_3-N (mg/l)	Organic-N (mg/l)	TN (mg/l)	PO_4-P (mg/l)	TP (mg/l)	TOC (mg/l)	TSS (mg/l)
Enjoro	Tiriytagoi	0.07±0.03	0.01 ±0.0	0.13 ±0.01	5.87 ±2.53	5.93 ±2.93	0.04 ±0.02	0.156 ±0.05	47.5 ±8.42	74.5 ±2.9
	Sigotik Upstream	0.09±0.04	0.02 ±0.0	0.13 ±0.01	0.90 ±0.13	1.05 ±0.14	0.06 ±0.04	0.157 ±0.05	35.83 ±11.0	41.5 ±6.71
	Sigotik Bridge	0.03±0.01	0.03 ±0.0	0.08 ±0.00	1.60 ±0.04	2.21 ±0.27	0.04 ±0.03	0.14 ±0.05	41.5 ±7.19	36.0 ±6.71
	Sigotik Downstream	0.07±0.02	0.01 ±0.0	0.09 ±0.07	7.01 ±2.55	7.09 ±2.52	0.06 ±0.04	0.15 ±0.06	32.62 ±3.73	41.0 ±10.3
	Highland Flowers	0.05±0.0	0.02 ±0.0	0.14 ±0.01	1.20 ±0.27	1.37 ±0.28	0.05 ±0.04	0.155 ±0.07	46.09 ±6.1	43.0 ±2.68
	Confluence Upstream	0.05±0.01	0.01 ±0.0	0.11 ±0.01	2.20 ±0.18	2.33 ±0.17	0.06 ±0.03	0.24 ±0.02	43.76 ±1.93	45.0 ±9.39
Little Shuru	Salt Spring	0.05±0.02	0.01 ±0.0	0.08 ±0.00	0.04 ±0.00	0.13 ±0.00	0.015 ±0.00	0.13 ±0.02	18.3 ±0.22	30.5 ±8.72
	Nessuit Bridge	0.07±0.03	0.01 ±0.0	0.12 ±0.00	0.17 ±0.22	0.85 ±0.21	0.045 ±0.02	0.23 ±0.08	72.49 ±11.4	43.5 ±11.2
	Forest Camp	0.12±0.08	0.02 ±0.0	0.12 ±0.01	1.70 ±0.17	1.85 ±0.17	0.075 ±0.01	0.19 ±0.1	41.65 ±6.1	34.0 ±0.90
Runguma	Runguma	0.02±0.0	0.01 ±0.0	0.10 ±0.00	0.20 ±0.04	0.32 ±0.04	0.09 ±0.08	0.148 ±0.025	82.63 ±7.28	57.0 ±8.27
Logoman	Logoman	0.15±0.02	0.02 ±0.0	0.09 ±0.01	2.80 ±0.12	2.92 ±1.13	0.085 ±0.056	0.18 ±0.01	26.53 ±3.1	37.0 ±2.56

Among the sampling sites on the Little Shuru tributary, the highest and most variable mean conductivity (277.67±57.92 µS/cm) was recorded at the Salt Spring site, which is located near an inflowing salty spring. The lowest (126.56±19.73 µS/cm) was observed at the Nessuit Bridge site. There are a number of small freshwater tributaries and springs which add fresh water to the Little Shuru as it flows downstream leading to the rapid reduction of conductivity downstream. Despite this, the lowest conductivity values on Little Shuru are higher than the highest values recorded on the Enjoro main channel. For Runguma and Logoman, their mean conductivity values were not significantly different from sites on Enjoro ($p>0.05$) but were significantly lower than those of sites on Little Shuru ($p=0.001$). Sampling sites on Enjoro, Logoman and Runguma tributaries had low mean pH as compared to those on Little Shuru (Table 2). Sigotik Upstream (7.11±0.07) and Tiriytagoi (7.59±0.12) recorded the lowest and highest mean pH, respectively on the Enjoro main stream. Mean pH was significantly lower at the Sigotik Upstream site in comparison to Tiriytagoi, Highland Flowers and Confluence Upstream sites as well as sites on Little Shuru tributary ($p=0.0002$). There were no significant differences between sampling occasions for this variable at all sites ($p>0.05$). In general, higher pH values were recorded on Little Shuru than all the other tributaries but a down or upstream pattern in pH did not exist. Also, no relations were observed between spatial variations in mean pH and land use.

Nitrate was particularly lower at Salt Spring, Runguma and Logoman and elevated at Highland Flowers, Nessuit Bridge, Sigotik Upstream and Tiriytagoi. Since these latter sites are located in cultivated and grazed parts of the watershed this may be related to both crop production and livestock concentration especially upstream of Tiriytagoi. Nitrite and ammonium concentrations were low (mean < 0.1 mg/l) at all sampling sites except Logoman which recorded a mean of 0.15±0.04 mg/l NH_4-N (Table 3). This site was heavily grazed because of its open glade grasslands favoured by Maasai herdsmen. The stream valley here was also nearly flat unlike the steep one observed at Tiriytagoi although both sites were grazed. Therefore, the elevated ammonia concentration may have been caused by dung and other cattle excreta deposited on the stream at this site. On Enjoro, the highest (7.01±2.55 mg/l) and the lowest (0.90±0.15 mg/l) mean concentrations of organic nitrogen were recorded at Sigotik Downstream and Sigotik Upstream, respectively. There was a significant difference between sites on Enjoro for this variable ($p=0.0001$). Particularly, Sigotik Downstream had significantly higher mean total organic nitrogen concentration than all other sites on Enjoro ($p=0.0000$) except Tiriytagoi ($p>0.05$). In Little Shuru, there was a gradual increase in the concentration of this variable downstream. The highest and lowest mean total organic nitrogen concentrations were recorded at Forest Camp and Salt Spring respectively. Logoman (2.8±0.12 mg/l) and Runguma (0.20±0.04 mg/l) recorded significantly lower means than Sigotik Downstream and Tiriytagoi ($p=0.0000$). Total Nitrogen followed same trends with total organic nitrogen (Table 3).

Very low levels of orthophosphate phosphorus were observed at all sites in the upper reaches of River Njoro (Table 3). The highest mean (0.087±0.03 mg/l) was recorded at the Logoman site. However, total phosphorus recorded the highest and lowest on Enjoro at Confluence Upstream and Sigotik Bridge respectively (Table 3). In the Little Shuru tributary, the highest mean concentration of total phosphorus (0.235±0.0 mg/l) were recorded at Nessuit Bridge and the lowest (0.136±0.01 mg/l) at Salt Spring respectively. Runguma and Salt Spring had significantly lower (p=0.000) mean total phosphorus concentrations than Confluence Upstream and Nessuit Bridge. It is important to remember that Confluence Upstream and Nessuit Upstream sites are located in cultivated parts of the watershed unlike Salt Spring and Runguma which are in densely forested parts of the watershed.

Runguma had the highest total organic carbon concentration than all other sites sampled (mean=82.63±14.03 mg/l; Table 3). This was expected because of its intact riparian zone combined with the contributions from the forested sub-watershed it drains. In the Enjoro main stream, the highest (47.07±8.42 mg/l) and lowest (32.62±11.44 mg/l) mean total organic carbon concentration were recorded at Tiriytagoi and Sigotik Downstream, respectively. This shows that the concentration of organic matter at the Runguma site is almost double those recorded at each site in the main Enjoro tributary. In Little Shuru, Nessuit Bridge had organic matter content close to that of Runguma (i.e. mean=72.49±0.22 mg/l TOC). This is likely to have been caused by the contributions of the relatively intact natural forest and riparian zone along this tributary in comparison to the main Enjoro stream whose sub-watershed and riparian vegetation has been degraded to some extent by logging, grazing and charcoal burning. In general therefore, there appears to be a close relationship between organic matter content of the water and the type of land-use of the watershed in the upper reaches of River Njoro.

Overall, water was clearer of suspended solids in Little Shuru than all the other tributaries. The lowest mean concentration of TSS (30.5±8.72 mg/l) was recorded at Salt Spring followed by Forest Camp (Table 3). The slightly elevated level observed at Nessuit Bridge, along Little Shuru, is likely to be a direct consequence of disturbance of streambed by cattle and people who frequent this watering point. On the Enjoro main stream, mean TSS concentration was higher initially at Tiriytagoi (74.5±2.9 mg/l) reduced gradually to its lowest at Sigotik Bridge but rose again to 45.0±9.39 mg/l at Confluence Upstream (Table 3). It is interesting to note that there was an abrupt rise at Sigotik Downstream as compared to Sigotik Bridge despite the fact that the two sites are hardly 300 metres apart. TSS concentrations at Runguma were elevated as compared to those recorded at both Salt Spring and Logoman for sites in the relatively undisturbed parts of the watershed. This is likely to have been caused by the high concentrations of suspended organic matter observed here.

Benthic macroinvertebrates

A total of 22,892 macroinvertebrates belonging to 70 taxa (Appendix 2) were identified from the samples collected in the upper reaches of River Njoro during this study. The nymphs of the mayfly (Ephemeroptera) *Baetis sp.* was the most abundant taxon, accounting for 38% in numbers followed by the blackfly (Diptera) larvae Simulidae (31%). Together with larval Chironomidae (Tanytarsini, Tanypodinae and Chironomini), *Caenis sp.* , Ostracoda, Sphaeridae, Lumbriculidae and *Hymanella sp.* they composed the ten most abundant taxa accounting for ≈95% by number of all invertebrates collected. These 10 taxa also occurred at all sites sampled. In the main Enjoro stream, the highest macroinvertebrate density (69,600 individuals/m^2) was observed at Highland Flowers in the dry season, while the lowest (200 individuals/m^2) were recorded at Confluence Upstream in May during the wettest month of the study period. The highest abundance recorded in the Little Shuru tributary was 12,800 individuals/m^2 at Salt Spring sampling site in August. The lowest (100 individual/m^2) was observed at the Nessuit Bridge site during the wettest month of the year. The highest (11,900 individuals/m^2) and lowest densities (200 individuals/m^2) at Runguma were recorded in the last weeks of April and May, respectively. In Logoman, the highest density of benthic macroinvertebates (19,400 individuals/m^2) was recorded in June while the lowest (100 individual/dm^2) was recorded in the last week of May immediately after heavy rains.

In general therefore, the highest abundance of macroinvertebrates in the upper watershed of River Njoro was recorded during the dry season while the lowest was observed during the wettest month of the year. This suggests that high water levels in the river reduce the density of organisms inhabiting the benthic substrates. It can also be deduced that at sites occurring in least impacted parts of the watershed, the variation between the highest and lowest densities is smaller. Cumulatively, the highest number of macroinvertebrate taxa (40) was recorded at Highland Flowers. However, eight out of this were adult beetles (Coleoptera) that were observed only once during the first sampling occasion and were never encountered again. This reduces the number of actual taxa found inhabiting this site to 32. In the main Enjoro channel, the lowest number of taxa was recorded at Sigotik Bridge (Table 4). Since this was the most frequented site by livestock and people, it is likely that the lower number of taxa recorded here as compared to both of its upstream and downstream “references”, was caused by the frequent disturbance of the sediments during water drawing and watering of cattle.

Table 4: Summary of taxa composition, density (individuals per square metre) and distribution of main taxa of benthic macroinvertebrates at sites in the upper reaches of Njoro River during the study period.

Tributary	Site	Highest Density	Lowest Density	Number of Taxa	Ephemeroptera taxa	Trichoptera Taxa	Diptera Taxa	Coleoptera Taxa
Enjoro	Tiriytagoi	89	3	26	2	4	7	2
	Sigotik Upstream	178	5	36	4	7	6	7
	Sigotik Bridge	59	2	29	3	4	6	2
	Sigotik Downstream	357	6	25	3	3	8	2
	Highland Flowers	696	3	40	3	4	10	8
	Confluence Upstream	65	1	30	3	6	6	4
Little Shuru	Salt Spring	128	4	36	6	8	9	4
	Nessuit Bridge	118	1	20	3	3	9	1
	Forest Camp	50	2	29	3	6	8	5
Runguma	Runguma	50	2	32	3	7	8	2
Logoman	Logoman	119	1	33	2	3	12	3

Overall, the highest number of taxa belonging to the insect order Ephemeroptera were observed at the Salt Spring sampling site, with *Oligoneuriella sp*. and *Ephemera sp.* occurring exclusively there. This site also recorded the highest number of taxa of the insect order Trichoptera. In general, Little Shuru recorded the lowest number of Coleoptera taxa unlike Enjoro where up to 8 families were recorded (Table 4). Curculionidae, Dyopidae and Laccophilidae occurred exclusively at Highland Flowers. Logoman had the highest number of taxa belonging to the insect order Diptera. Non-insect taxa like Turbellaria (*Hymanella sp*., *Mesostoma sp*. and *Catenulla* sp.), Oligochaeta (Lumbriculidae, Tubificidae and Naididae), Nematoda, Mollusca (Gastropoda and Sphaeridae), Crustacea (Cyclopoidae and Ostracoda), and Hydracarina occurred in low abundances and did not show any preferences for specific sites.

The freshwater crab (*Potamon sp.*) was however observed in relatively large numbers at the upper most sites (Tiriytagoi, Runguma and Salt Spring). Particularly, they occurred in abundance at the Salt Spring. Also, the mayfly nymph, *Leptophlebia sp.* occurred almost exclusively in Little Shuru as it was present at all its three sites and only rarely on the main Enjoro stream. The larval caddisfly (Trichoptera) Lepidostomatidae occurred in large numbers in the forested upstream sites (Runguma and Salt Spring) but rarely at the sites along cultivated parts of the watershed. The Coleoptera family, Helodidae, also occurred almost exclusively at Runguma and Salt Spring. This implies that these taxa are sensitive to land use change as they occurred rarely at grazed and cultivated parts of the watershed.

There was a significant difference in median abundance of *Baetis sp.* between sampling sites (Kruskal-Walis ANOVA by ranks test, $n=231$; $H=81.61$; $p=0.000$). However, sites on unsettled parts of the watershed (Tiriytagoi, Logoman, Runguma, and Salt Spring) recorded low median relative abundances of *Baetis sp.* compared to Sigotik Upstream, Highland Flowers, Nessuit Bridge and Confluence Upstream that recorded upto 75% dominance by *Baetis sp.* The latter sites are in cultivated and settled areas of the River Njoro watershed along both Little Shuru and Enjoro tributaries. The relative abundance of Simulidae was significantly higher at Sigotik Downstream compared to all other sites ($p=0.0000$). There were also significant differences between sites in the median relative abundances of Tanypodinae ($p=0.004$), Tanytarsini ($p=0.002$), *Hymanella sp.* ($p=0.002$), Lumbriculidae ($p=0.0000$) and *Caenis sp.* ($p=0.0068$). However, the median relative abundances of these latter taxa did not show any discrimination of sites based on settlement or forestry unlike *Baetis sp.* There were no significant differences between sampling occasions in the relative abundances of taxa at all sites except for *Baetis sp.* and Simulidae which were significantly abundant and composed upto 88% of all invertebrates collected at Highland Flowers and Sigotik Downstream in July respectively.

In the Enjoro stream, the highest mean taxon richness was recorded at the Highland Flowers site, while the lowest was recorded at the Confluence Upstream site (Table 5). In the Little Shuru tributary, the highest (9.4±0.79) and lowest (4.27±0.8) mean taxon richness were recorded at Salt Spring and Nessuit Bridge sites, respectively (Table 5). There was a significant difference in macroinvertebrate taxon richness between sampling sites on Little Shuru ($p=0.002$). However, the Salt Spring site was not significantly different from the Logoman and Runguma sites ($p>0.05$). Since these sites were in the less impacted parts of the watershed this suggests that change in land use from forestry to agriculture and settlement has caused a reduction in the number of taxa inhabiting the benthic habitats in the upper parts of the watershed.

Table 5: Summary of mean (±S.E.) community structure attributes for sites sampled in the upper reaches of River Njoro.

Tributary	Sampling Site	Taxon richness	Diversity	Evenness	Dominance
Little Shuru	Salt Spring	9.44±0.79	0.71±0.04	0.74±0.25	0.25±0.03
	Nessuit Bridge	4.27±0.80	0.40±0.05	0.63±0.25	0.31±0.06
	Forest Camp	6.81±0.56	0.59±0.05	0.70±0.25	0.30±0.04
Logoman	Logoman	7.57±0.99	0.58±0.05	0.67±0.25	0.28±0.04
Runguma	Runguma	10.06±0.74	0.75±0.03	0.76±0.25	0.24±0.04
Enjoro	Tiriytagoi	6.35±0.80	0.84±0.29	0.99±0.25	0.01±0.29
	Sigotik Upstream	6.9±0.74	0.38±0.04	0.46±0.25	0.54±0.05
	Sigotik Bridge	6.0± 0.79	0.51±0.05	0.68±0.25	0.32±0.05
	Sigotik Downstream	6.38±0.60	0.43±0.04	0.54±0.25	0.46±0.05
	Highland Flowers	7.18±0.76	0.41±0.05	0.50±0.25	0.50±0.07
	Confluence Upstream	5.5±0.72	0.50±0.05	0.65±0.25	0.35±0.05

In similarity to the taxon richness, benthic macroinvertebrate diversity did not show a particular downstream trend on both Little Shuru and Enjoro tributaries. However, sites in the forested unsettled areas of the watershed, for instance, Tiriytagoi, Runguma and Salt Spring recorded higher mean diversities of benthic macroinvertebrates in comparison to the settled and cultivated sites like Sigotik Upstream, Nessuit Bridge and Highland Flowers (Table 5). Diversities in the unsettled sites were significantly higher than those of settled sites ($p=0.001$). This indicates that land use type has an influence on the diversity of benthic macroinveretebrates in the upper River Njoro watershed. There was a significant difference between sampling occasions for this attribute at sites on Little Shuru ($p=0.0000$). Particularly, Salt Spring had significantly higher diversities during all sampling occasions than Nessuit Bridge ($p=0.001$), and Forest Camp which had reduced diversities over the dry than wet seasons ($p>0.05$).

Macroinvertebrate evenness showed contrasting patterns with dominance. High mean dominance was associated with very low mean evenness for the same samples and sites (Table 5). In Enjoro, the highest and lowest mean evenness were recorded at Tiriytagoi and Sigotik Upstream respectively. Subsequently, the highest dominance was recorded at Sigotik Upstream. Tiriytagoi recorded low densities and relatively high taxon richness. Consequently, high evenness values were observed here in the main Enjoro stream. In contrast, Sigotik Upstream and Highland Flowers had high densities of benthic macroinvertebrates dominated by *Baetis sp*. Tiriytagoi (0.76±0.25) recorded the highest mean evenness of benthic macroinvertebrates in the entire watershed followed by Salt Spring, Runguma and Logoman, and these sites did not differ significantly in mean evenness between each other ($p>0.05$). In general therefore, higher evenness of distribution of benthic invertebrates were recorded at the least impacted sites unlike sites on settled and cultivated parts of the watershed where the communities were dominated by a single taxon usually *Baetis sp*. or Simulidae.

Relationships between physico-chemical factors and benthic macroinvertebrate composition and community structure

Based on Spearman rank order correlations, several significant relationships were observed. For example, pH was positively correlated with conductivity ($r=0.71$; Table 6). This can be explained by the high values of pH and conductivity recorded in the Little Shuru tributary. Macroinvertebrate taxon richness showed positive and negative correlations with temperature and total phosphorus, respectively. Similarly, macroinvertebrate taxa diversity showed positive and negative correlations with conductivity and total suspended solids respectively. Dominance showed significant positive correlations with total phosphorus. The relative abundances (%) of several individual taxa also gave significant associations with physico-chemical variables. For example, the relative abundance of *Baetis sp.* was positively and negatively correlated to TSS and conductivity. Similarly, Simulidae gave significant associations with organic nitrogen and pH. Also, positive and negative correlations were observed between Tanypodinae and Sphaeridae with conductivity and temperature respectively (Table 6). Helodidae, *Leptophlebia sp.,* Lepdostomatidae and *Potamon sp.* were positively correlated to D.O., pH and TOC but negatively correlated to temperature, total phosphorus, total nitrogen, TSS and nitrites.

Table 6: Spearman rank order correlation coefficients observed among physico-chemical parameters and between physico-chemical variables and taxa and community structure attributes. (** Correlation is significant at α=0.01. * Correlation is significant at α=0.05). All units remain as they are shown in previous tables.

	Temperature	TP	TN	TSS	NO_2-N	DO (mg/l)	O-N	Condu-ctivity	pH	TOC
TP	0.12		0.42*	0.58*	0.13	0.08	-0.01	0.58*	0.57	-0.32
Ph	0.12	0.57*	0.01	0.30	0.01	0.07	-0.12	0.09		0.21
Richness	0.96**	-0.93**	-0.14	-0.34	-0.51*	0.32	-0.13	0.22	0.37	0.41
Diversity	-0.32	-0.22	-0.34	0.97**	0.01	-0.20	-0.34	-0.96**	-0.01	0.38
Dominance	0.15	0.86**	0.28	-0.48*	-0.29	0.27	0.27	-0.09	0.19	-0.35
Baetis sp.	-0.37	0.10	-0.24	0.88**	-0.36	0.38	-0.23	-0.86**	0.45*	0.11
Simulidae	0.35	-0.17	0.79*	-0.24	-0.14	-0.48*	0.96**	-0.15	0.64*	0.01
Tanypodinae	-0.04	0.21	-0.12	-0.06	-0.55*	-0.02	-0.10	0.79*	0.25	0.21
Sphaeridae	0.71*	-0.17	-0.03	0.33.	-0.13	-0.22	-0.02	0.10	-0.12	0.11
Potamon sp.	-0.76*	-0.25	-0.31	-0.47*	-0.39	0.83**	-0.24	0.47	0.44*	0.49*
Helotidae	-0.27	-0.38*	-0.48*	-0.18	-0.11	0.04	0.04	-0.14	0.08	-0.11
Lepidostomatidae	-0.22	-0.22	-0.32	-0.13	-0.38*	-0.14	0.11	-0.14	-0.17	0.14

Discussion

Spatio-temporal variations in physico-chemical factors in upper river njoro

Water temperature showed two distinct spatial patterns in the upper reaches of River Njoro. Firstly, the downstream increase in water temperature on both Enjoro and Little Shuru tributaries. Secondly, those reaches of the river with open canopy exhibited high temperatures than shaded ones. Temperatures of streams and rivers vary in relation to air temperatures (Wetzel, 2001) and therefore a strong linear relationship exists between air and river water temperature (Allan, 1995). Air temperatures often increase progressively with decreasing altitude, with high temperatures in lower plains and low temperature in the mountains. Hence, temperature often increase progressively from the headwaters to the mouth of rivers (Vannote *et al*., 1980; Wetzel, 2001).This observation explains the first pattern observed in this study.

Within particular sections, or streams at similar altitudes, however, temperatures vary in response to the intensity of vegetation cover, groundwater seepage, channel depth and shape, and stream orientation (Walling and Webb, 1992). In comparing Runguma, Salt Spring, Logoman and Tiriytagoi sampling sites it was surprising to find that the later two sites had elevated temperatures despite occurring at a relatively higher altitude. Tiriytagoi was logged to the stream bank in 1998 and has not been replanted to date (Personal communication with the members of the local community). It is also heavily grazed just as is Logoman. As a result, there is little canopy cover along the two sites. In contrast, Salt Spring and Runguma are heavily shaded by both marginal macrophytes and woody plants offering canopy cover. Differences in their water temperatures therefore may be as a result of differences in canopy cover. Open canopies expose the stream to direct insolation. This may have also been responsible for the slightly elevated temperature at Sigotik Bridge compared to both Sigotik up and downstream sites. As first order streams, they are shallow and therefore have a high surface area to volume ratio which allows for fast heating of water. In contrast, small and heavily canopied streams exhibit only very small temporal changes in temperature often reflecting temperature of forest soil (Wetzel, 2001). The high temperature recorded in the dry season was expected because at this time aerial temperature is high.

Percentage saturation of oxygen in the upper reaches of River Njoro was generally high (slightly above 100% saturation). This indicates that this part of the river is well aerated. This can be attributed to characteristic steep slope typical of a stream in headwater reaches. Furthermore, this part of the river has a high number of riffles allowing for turbulent mixing of the water hence the high dissolution of oxygen. The sampling period covered mainly the rainy season when the inputs of organic matter from stream side vegetation is lower compared to the dry season (Magana, 2001). Allan (1995) reported that oxygen content within small, turbulent streams like River Njoro, is always approximately at, or somewhat above, saturation.

Overall, there is a decrease in dissolved oxygen downstream. This is corroborated with the increasing water temperature at almost similar magnitudes moving down the river. These observations concur with the observations of Wetzel (2001) that there is a decrease in dissolved oxygen solubility with increasing temperature in all aquatic systems. Another mechanism for decreasing dissolved oxygen concentration downstream River Njoro is possible if an increase in temperature induces an increase in aerobic respiration involving decomposition of organic materials derived from upper reaches of the river and the stream bank. This is a universal observation in rivers, especially those draining modified catchments (Hynes, 1975).

The close association observed between conductivity and pH, especially in the Little Shuru tributary, indicates possible influence of underlying sedimentary geology along this tributary. Sedimentary rocks easily dissolve in water (Allan, 1995) and therefore associated ions get leached to surface waters through springs. This probably explains the elevated conductivity and pH recorded at the Salt Spring site, which is fed by a number of small trickles from streamside springs. The high pH indicates carbonate enrichment of the stream, as carbonates buffer water against acidity by increasing water alkalinity. This enrichment is however reduced as the stream meets other fresh water tributaries that dilute the salt content. Water conductivity is lower by more than half by the time it gets to Nessuit Bridge. The low conductivity recorded in the rainy season is a consequence of dilution of the stream by surface runoff resulting from rains over the watershed above Salt Spring.

The elevated level of ammonia recorded at the Logoman is likely to be due to comparatively higher inputs of decomposing animal dung visible in the water at the site. Concentration of animals on land causes direct and indirect effects on aquatic ecosystems, the most obvious being additions of ammonia and nitrite as a consequence of increased runoff of animal waste into streams from grazed fields (Gammon *et al*., 2003). In their study, Gammon *et al*. (2003), investigated sites downstream of animal feed lots and found that they had elevated levels of ammonia, pH and turbidity. Apparently, this observation seems to be contrasted by that the lower levels of ammonia concentration at Tiriytagoi though this site is nearby and also heavily grazed by livestock. This can be partly explained by the length of the grazed stretch of stream at Tiriytagoi. Field surveys upstream of Tiriytagoi indicated that concentration of cattle increased up to 10 km upstream of the site. Given the high level of oxygen in the colder upstream stretch of the river, it is possible that ammonia is actually generated from decomposing animal waste deposited on the stream, or washed into the stream from the grasslands; but gets oxidized as the water flows downstream. This explanation is supported by the elevated level of nitrates recorded at Tiriytagoi, which results from oxidation of ammonia (Wetzel, 2001). The pH recorded at Tiriytagoi was nearly neutral. Neutral pH increases mineralization of ammonia (Allan, 1995: Wetzel, 2001).

All the grazed sampling sites had ammonium values much greater than the 0.018 mg/l mean calculated by Meybeck (1992) for unpolluted rivers. This shows that the grazed stretches of River Njoro are polluted by inputs of livestock manure. Livestock grazing has negative impacts on the water quality of the river, especially the small first order streams as exemplified by the Logoman tributary. Further research needs to be done to ascertain the apparent significant contribution of livestock to water quality in the River Njoro Watershed.

The spatial variations of other nutrients, especially nitrates and phosphates seem to be associated more with crop farming land use rather than grazing and forestry in upper River Njoro Watershed. For example, the mean nitrate concentration recorded at the Highland Flowers sampling site is almost double that recorded at the Salt Spring site in the densely forested part of the watershed. Similarly, significantly higher concentrations of total phosphorus were recorded at the Confluence Upstream site, with small scale crop farming of maize and beans. Lower values of phosphorus were recorded at the forested sampling sites (Salt Spring, Runguma and Logoman). Dissolved inorganic phosphorus often increases to levels of 0.05 and 0.1 mg/l from agricultural runoff (Wetzel, 2001). Ormanik (1977) recorded an increase in phosphorus and nitrogen from agricultural runoff as compared to runoff from the landscape under indigenous forests. Ideally, nutrients in runoff from agricultural land should be detectable in the dissolved forms, especially if measured during or immediately after storm events. However, as dissolved nutrients get to streams, their concentrations decline downstream as a result of physical adsorption to suspended solids that get precipitated, and to particulate seston as well as assimilation (Ormanik, 1977). When the river level rises again, the nutrients, now in adsorped form get mobilized downstream and are detected in total phosphorus. This mechanism is likely to have been responsible for the elevated phosphorus and nitrogen concentrations at the downstream cultivated sites in the upper reaches of River Njoro Watershed. It has been reported in experimental watersheds in the USA (Allan, 1995).

This study shows a significant correlation between total phosphorus and total suspended solids. This implies a common origin. It is likely that the implied common origin of phosphorus and TSS is agricultural land. In comparison to forest and grasslands, cultivated land is easily erodible because of reduced surface roughness and organic matter content (Wetzel, 2001). It is usually rich in nutrients because of artificial fertilization to increase crop production per unit acre of land. In general, the topography of the catchments influences the extent of erosion and the subsequent export of nutrients (Wetzel, 2001, Allan, 1995). The upper reaches of River Njoro, being of a high slope, typically is more susceptible to erosion. To counter negative influences of nutrient pollution of River Njoro, preservation of a vegetated riparian strip could be considered as a strategy. For example, a land-water "buffer zone" of wetland and littoral vegetation of 20-

30m in width can remove nearly all nitrates by denitrification (Wetzel, 2001) and phosphorus by direct assimilation (Allan, 1995; Wetzel, 2001).

High total organic carbon concentrations were recorded at the densely forested sampling sites (e.g. Runguma) while low values were recorded in cultivated, grazed and clear felled areas of the River Njoro Watershed (e.g. Tiriytagoi, Logoman and Sigotik Upstream) during the study. This finding suggests that grazing and logging affect the level of organic matter supply, retention, transformation and release in streams. Terrestrial plants form much of the allochthonous organic matter inputs to aquatic ecosystems (Bretschko, 1995b). These materials are transformed by microbial activity then leached to the stream through groundwater and runoff (Wetzel, 2001). Removal of terrestrial vegetation, such as by clear-cutting of forests of a drainage basin or removal of riparian vegetation for agriculture or grazing, reduces the inputs of organic matter to the soil, and this reduces the amount of total organic carbon in runoff or groundwater inputs to streams (Allan, 1995). Consequently, streams running through parts of the watershed that have a sparse vegetation cover, are expected to have low concentrations of organic matter suspended and dissolved in water. This is likely to have been responsible for the reduced levels observed in stream water from areas of the watershed experiencing grazing, deforestation and agriculture.

It was surprising, however, that low levels of TOC were recorded at the forested Salt Spring site. This site is steep and the flushy nature of the river allows for fast downstream delivery of allochthonous organic matter. High gradient streams are known to be less retentive of organic matter (Dudgeon, 1994).As the gradient becomes gentle, organic matter e.g. leave litter and woody debris are processed and released as fine particulate organic matter which caused the high levels of TOC recorded at Nessuit Bridge.

Total suspended solids in this study were used as a measure of sediment inputs into the river from the upstream landscape of each sampling site. The highest total suspended solids concentration was recorded at Tiriytagoi, the uppermost site on the main Enjoro tributary, while the lowest was recorded at the Salt Spring sampling site, the upper most site on the Little Shuru tributary. It was surprising to find elevated levels of total suspended solids at the Tiriytagoi site in comparison to the highly disturbed downstream sites where intensive agriculture is practised and especially Logoman a similarly intensively grazed site. A combination of intensive upstream grazing and sloughing-off of stream banks following heavy rains in logged areas may have been responsible for this. Tiriytagoi is a livestock watering point which has steep river banks with loose soil overhanging the river. It is possible that logging which occurred at Tiriytagoi in 1998 removed roots which otherwise were holding the soil to the river banks. This resulted into the river banks becoming unstable especially when trampled by livestock that frequent the site for watering. Unstable banks are a major source of suspended soil particles from agricultural (Weaver and Garman, 1994), logged (Hartman *et al.*, 1996) and grazed (Wohl and Carline, 1996) watersheds.

Yankey *et al.* (1991) after conducting a 10-year study on non point-source pollution in Rock Creek, Idaho, concluded that eroded stream banks in grazed areas were a major source of sediments. They estimated that sediment from grazed areas was 2-5 times greater than that from croplands. It is important to note that no attempt was made to estimate the contributions of total suspended solids by grazing and agricultural lands during this study. However, the rise in the concentration of total suspended solids in water from the Sigotik Downstream sampling site to the Confluence Upstream site is likely to have been caused by surface runoff from surrounding agricultural land as well as erosion of the many foot and livestock paths connecting homesteads and the watering points along the river in this settled part of the watershed.

Spatio-temporal variations in the distribution and community structure of benthic macroinvertebrates

The 70 taxa, a majority of which were identified only to family level, encountered in samples from the upper reaches of River Njoro, show that it is rich in benthic macroinvertebrates. The composition of these taxa, however, is similar to those identified from similar streams elsewhere. For example, *Baetis sp.* and Simulidae formed 69% by number of all benthic taxa identified. Van Someren (1952), from his investigations of Sagana, Naro Moru, Sirimon and Barguret streams of Mount Kenya, found that at high altitudes, the river bed communities were completely dominated by the Mayfly *Baetis sp.* and the Blackfly *Simulium sp.* and that all other taxa occurred in much less numbers. Repeat visits to these streams by Mathooko and Mavuti (1992) also recorded similar findings. Dudgeon (1994b) while investigating six Guinean streams found that in overall terms, Baetidae dominated the benthos. However, he found that Simulidae did not form a large percentage of the benthos. The present findings however, seem to differ from those of Ahmel-Abdallah *et al.* (2004) on upland streams of Tanzania who observed that freshwater crabs dominated the benthos.

It is interesting to note that a clear downstream longitudinal distribution pattern of macroinvertebrate taxa does not exist in the upper reaches of River Njoro. This contradicts the River Continuum Concept (Vannote *et al.*, 1980) which predicts an increase in taxa from headwaters to mid reaches as water temperature increase in a stream. The observed reduction in diversity of benthic macroinvertebrate may be attributable to change in habitat quality as more land in the watershed is converted to agriculture and the riparian zone is eroded due to frequent trampling by livestock. Although Lepidostomatidae, Seriscostomatidae, Helotidae and *Potamon sp.*, which are well known shredders (Ahmel-Abdallah *et al.*, 2004), were present in many sites, high abundances of these taxa occurred only at Salt Spring, Runguma and reduced abundances were recorded at Logoman and Forest Camp. Presence at other sites may have been caused by downstream drift especially over the wet season. This is supported by the fact that only a few or a single specimens were encountered, especially in sites along cultivated parts of the watershed.

The above observation suggests that only at least-impacted sites are food items (litter fall, wood debris) of high quality found. Actually, a majority of the downstream sites have canopy interruptions from deforestation, riparian grazing or direct river use as watering points. In settled areas, every farm adjacent to the river must create an access path to the stream through which they pass with livestock to the only source of water.

As farms in the upper reaches of River Njoro are small, usually 5 acres or below, foot paths on both sites of the river are numerous, and this increases the cumulative areas of open canopy that allow sunlight to reach the stream. This increases temperature variability that can expand beyond evolutionary histories of a majority of shredder taxa because of their association with heavily canopied streams whose temperatures vary less over time. Also frequent disturbance of the streambed, mean that only a few taxa tolerant to shifting sediments and bedrock can proliferate in large numbers while a majority of the taxa occur only rarely. This impact is increased during rare storm events when large amounts of sediments are eroded from cultivated farms and foot paths, and then washed into streams. Suspended solids reaching streams can smoother the riverbed, flush away substrates on the river and associated invertebrates and increase the amount of fine sediments in pools which can be colonized only by a few specialized taxa (Bretschko, 1995a). This may have been responsible for large variability in the density of invertebrates between stretches of the river in cultivated and forested parts of the watershed.

Forests, especially indigenous ones, serve to delay the rate of delivery of water from the landscape, trap eroded sediments and therefore tend to maintain relatively stable flows (Carlisle *et al.*, 2004). This allows for development of a community of invertebrates that is diverse and rich (Wohl and Carline, 1996). Environmental stability is listed as a major factor controlling the structure tropical ecosystems (Jacobsen and Encalada, 1998). Also, the sharp decreases in density of invertebrates following the onset of the heavy rains, in the upper reaches of River Njoro suggests that reductions in numbers is a stochastic process in which invertebrates are flushed out of the streambed during flood events. This finding is in agreement with those of Jacobsen and Encalada (1998), Hynes (1975) and Dudgeon (1993) from other tropical rivers.

Diversity is a structural character of an ecosystem (Bretschko, 1995a). High species diversity allows for complex population interactions involving energy transfer, competition and niche apportionment (Brower *et al*., 1990). It is generally expected that unimpacted sites contain diverse benthic communities (Jones *et al.*, 2002). Anthropogenic disturbance from changes in land use practice is known to affect invertebrate diversity patterns in streams draining modified catchments (Death, 2000), often leading to a decrease in total number of taxa as well as a shift to a more unevenly distributed community where one or two taxa are numerically dominant (Jones *et al.*, 2002; Carlisle *et al.*, 2003). The high level of dominance by a few benthic taxa at sites experiencing agricultural impacts (e.g. Sigotik Upstream, Sigotik Downstream and Highland Flowers) compared to evenly distributed taxa in pre-settlement areas (Runguma, Logoman and Salt Spring) suggests that human activities are already shaping benthic communities in the upper reaches of River Njoro.

Dudgeon (1994b) recorded a decrease in zoo-benthic taxa colonizing artificial substrates placed in an impacted site compared to a control. In that particular study, he found that in comparisons of impacted and unimpacted sites, reductions in zoobenthos densities and diversities were related to changes in stream sediments and increased suspended solid loads associated with agricultural activities. In a related experiment, he observed rapid reductions in density and diversity when macroinvertebrates were transferred to the impacted site, thereby providing strong support for the detrimental effects of high TSS loads. Immediate reductions in density of this type had earlier been attributed to catastrophic drift caused by shifting sediments (Culp *et al.*, (1986). Erosion of land is likely to increase the amount of fine sediments deposited in pools of impacted sections of the stream. Fine sediments are easily mobilized downstream (Wohl and Carline, 1996). Bretschko (1995a) recorded reductions in invertebrate densities in pools with finer sediments in River Njoro. He observed that the shifting nature of fine sediments in streams makes them less attractive for colonization by invertebrates and therefore would always have a fewer number of invertebrates as compared stable larger particles. In constantly disturbed streambeds, even larger particles are less attractive to colonization because their surfaces are covered by silt (Allan, 1995). Silt-covered cobbles have less potential for colonization by periphyton whose microfilm is fed on by scrapping macroinvertebrates. At Sigotik Bridge larger particles at riffles were brown in colour and less slippery in nature. This provides evidence that microbial colonization is low. Typically, such substrates are less attractive to macroinvertebrate colonization (Allan, 1995). This explains why invertebrate density was lower at Sigotik Bridge. Smoothering of larger particles by soil particles is actually caused by the shifting soil particles caused by constant stirring of the streambed especially in the pools where livestock (especially cattle, sheep and donkeys) drink from.

The significant correlations observed in this study between TSS and dominance of macroinvertebrates by a few taxa shows clearly that mobilization of sediments from agricultural lands was responsible for reductions in numbers of seemingly sensitive taxa and the proliferation of *Baetis sp*. and Simulidae, the two taxa responsible for dominace. Coldwater organisms are generally known to be more sensitive to environmental change (Griffiths, 1996), and are therefore expected to respond quickly to disturbances.

Relationships between physico-chemical factors and benthic macroinvertebrates

More recent analyses of stream benthic communities, have shifted toward analyses of changes in population and community dynamics in response to environmental variability (e.g. Townsend *et al.*, 1987; Richards and Host, 1993; Mathooko, 2001; Wells *et al.*, 2002; Carlisle *et al.*, 2003 and Ahmel-Abdallah *et al.*, 2004). As such, an essential step in stream restoration is the identification of the suite of environmental factors that strongly influence the composition and abundance of stream biological communities. This study shows that TSS, total phosphorus, organic nitrogen and conductivity are the main physico-chemical factors controlling community structure as well as the distribution of specific taxa in the upper reaches of River Njoro.

Direct effects of phosphorus and nitrogen compounds on invertebrate communities are difficult to determine because no direct relationships have been reported. They are however likely to influence benthic communities through indirect pathways like primary production (Richards and Host, 1993). Although primary production was not measured in this study, it is likely that it was positively correlated to rises in nutrients and exposure of the river arising from agricultural activities. This is highly likely because agriculture, as the principle economic activity in rural areas, causes reduced canopy cover in streams as farmers extend tillable to the stream edge, cut down trees in riparian zones for charcoal burning, and acquire building materials for farm development from woody riparian vegetation that otherwise would reduce sunlight reaching streams. Increased sunlight intensity coupled with nutrient enrichment from surface runoff favours primary production (Vannote *et al.*, 1980). These changes the availability of food resources to those produced from within the stream besides increasing the demand for oxygen at night. Therefore, species that do not tolerate high environmental variability would be reduced in numbers or otherwise disappear from affected sections of the stream (Allan, 1995).

The dominant taxa at impacted sites i.e. *Baetis sp.* and Simulidae recorded in this study are known to have short regeneration times and rapid colonization rates (Hynes, 1975) enabling them to cope with fluctuating environments and build up large populations opportunistically (Newbold *et al.*, 1980). On the hand, the significant correlations between *Potamon sp.*, Lepidostomatidae and Helodidae with TOC, conductivity and pH show that these taxa, especially Helodidae and *Potamon sp*. can be used to monitor stream recovery, especially if riparian zone reclamation can be adopted as a conservation strategy to achieve a higher watershed health. The negative correlations observed between numbers of these taxa with total nitrogen, total phosphorus and suspended solids imply that they occur in low numbers in areas affected by agricultural impacts. Ideally, as watershed health improves, the riparian strip along the river is restored. Consequently, more allochthonous materials are generated to the stream. In addition, shading will intensify; as such stream temperatures will become less variable. Since stream temperature variability is a strong factor controlling cold water species (Wetzel, 2001), watershed restoration at the riparian strip and landscape level will favour proliferation of sensitive taxa which will extend lower on the river than now. Particularly, shredders will be favoured because more food items, and perhaps of a higher quality, become available following restoration.

Similarly, dominance, as a community structure index, as well as the abundance of *Baetis sp.* and Simulidae can be used to monitor deterioration of stream habitats and water quality in the upper reaches of River Njoro. This is supported by the significant positive correlation between these two taxa with total suspended solids and total nitrogen. These variables are associated with impact from grazing (Wohl and Carline, 1996) and cultivation (Allan, 1995). As such, their dominance supports the idea that *Baetis sp.* and Simulidae are tolerant to agricultural impacts (Hynes, 1960; Newbold *et al.* 1980). The significant positive correlations observed between the diversity of benthic invertebrates and total suspended solids in this study, implies that diversity of benthic invertebrates is limited by suspended solid generating activities. For River Njoro, this is grazing and land cultivation.

Conclusions

Results from this study show that the river is warmer in grazed and cultivated parts of the watershed. Suspended solids and nutrient variables are also elevated at these areas. Consequently, a few seemingly sensitive taxa (e.g. *Potamon sp.*, Lepidostomatidae and Helotidae) are restricted to the forested and undisturbed upper stretches of the river where higher richness, diversity and evenness of invertebrates were recorded. Therefore, land use transition from forestry to grazing and agriculture has affected the physicochemical conditions of the river as well as the structure of benthic communities.

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References

Ahmel-Abdallah *et al.* 2004. Comparative Studies on the Structure of an Upland African Stream Ecosystem. *Freshwater forum* 21: 27-47

Allan, J.D. 1995. *Stream Ecology: Structure and Function of Running Waters*. Chapman & Hall, London. 388 p

American Public Health Association (APHA). 1998. *Standard Methods for Examination of Water and Wastewater*, 20th ed., APHA, Washington, D.C. 1037 p.

Bretschko, G. 1995a. Tropical River Ecology Initiative. Proceedings of the first workshop on the limnology of Njoro River, Egerton University, Kenya. 92 p.

Bretschko, G. 1995b. River/land Ecotones: Scales and Patterns. *Hydrobiologia* 303: 83-91

Bretschko, G. 1996. Tropical River Ecology Initiative. Proceedings of the 2nd workshop on the limnology of Njoro River, Egerton University, Kenya. 96p.

Brower, J.E., J.H. Zar and C.N. von Ende. 1990. *Field and Laboratory Methods for General Ecology*. WM. C. Brown Publishers, New York. 237 p.

Cairns, J.Jr.. 2003. Biotic Community Response to Stress. Pages 13-22 in T.P. Simon (Ed). *Biological Response Signatures: Indicator Patterns using Aquatic Communities.* CRC Press, Boca Raton. 576 p.

Carlisle, D.M., P.M. Stewart, and J.T. Butcher. 2003. Macroinvertebrate Assemblages Associated with Patterns in Land Use and Water Quality. Pages 271-286 in T.P. Simon (Ed). *Biological Response Signatures: Indicator Patterns using Aquatic Communities.* CRC Press, Boca Raton. 576 p

Clarke, G.M. 1969. *Statistics and Experimental Design: Contemporary Biology*. Edward Arnold Ltd, London. 161 p.

Culp, J.M., F.J. Wrona, and R.W. Davies. 1986. Response of Stream Benthos and Drift to Fine Sediment Deposition versus Transport. *Can. J. Zool.* 64: 1345-1351

Dance, K.W., and H.B.W. Hynes.1980. Some Effects of Agricultural Land Use on Stream Insect Communities. *Environ. Pollut*. 22:19-28.

Death, R.G. 2000. The Effect of Land Use on the Species-area Relationships in Benthic Stream Invertebrates. *Verh. Internat. Verein. Limnol*. 27: 2519-2522

Death, R.G. and M.J. Winterbourn. 1994. The Measurement of Environmental Stability in Streams: a Multivariate Approach. *J. N. Am. Benthol. Soc.* 13: 125-139

de Moor, F.C., H.M. Barber-Jones, A.D. Harrison, C.R. Logo-Ortiz. 2000. The Macroinvertebrates of the Cunene River from the Rucana Falls to the River Mouth and Assessment of the Conservation Status of the River. *African Journal of Aquatic Science* 25: 105-122.

Dudgeon, D. 1993. Thhe effects of spate-induced disturbance, Predation and Environmental Complexity on Macroinvertebrates in a tropical Stream. *Freshwat. Biol*. 30: 189-197

Dudgeon, D. 1994a. The Influence of Riparian Vegetation on Macroinvertebrate Community Strucure and Functional Organization in six New Guinea Streams. *Hydrobiolgia* 294: 65-85

Dudgeon, D. 1994b. Functional Assessment of the Effects of Increased Sediment Loads Resulting from Riparian-zone Modification of a Hong Kong Stream. *Verh. Internat. Verein. Limnol* 25: 1790-1792

Edington, J.M. and A.G. Hildrew. 1995. Caseless Caddis Larvae of the British Isles: A Revised Key with Ecological Notes. Freshwater Biological Association, *Scientific publication* No. 53. 136 p.

Fore, L.S. 2003. Biological Assessment on Mining Disturbance of Stream Inveretebrates in Mineralized Areas of Colorado. Pages 347-370 in T.P. Simon (Ed). *Biological Response Signatures: Indicator Patterns using Aquatic Communities.* CRC Press, Boca Raton. 576p.

Gammon, J.R., C.F. Wayne and T.P. Simon. 2003. Patterns in Water Quality and Fish Assemblages in three Central Indiana Streams with Emphasis on Animal Feed Lot Operations. Pages 373-419 in T.P. Simon (Ed). *Biological Response Signatures: Indicator Patterns using Aquatic Communities.* CRC Press, Boca Raton. 576 p.

Gordon, N.D., T.A. McMahon, and B.L. Finlayson. 1993. *Stream Hydrology: An Introduction for Ecologists*. John Wiley & Sons. 526 pp.

Gore, J.A. and D. Shields Jr. 1995. Can Large Rivers be Restored? *BioScience* 45: 142-152

Graynoth, E. 1979. Effects of Logging on Stream Environments and Faunas in Nelson.
Newfoundland Journal of Marine and Freshwater Research, 13 (1): 79-109.

Griffiths, R.W. 1996. Biomap: Concepts, Protocols and Sampling procedures for Southwestern Region of Ontario. Biomap Report SWR-1. Ministry of Environment and Energy, Southern Region, London, Ontario.

Haach corporation. 1987. A Method for Total Kjeldahl Nitrogen Determination. *Journal of Association of Official Analytical Chemists.* 70 (5): 783-787

Hall, J.D., R.L. Lantz. 1969. Effects of Logging on the Habitat of Coho Salmon and Cutthroat Trout in Coastal Streams. Pages 353-375 in North-Cote, T.G. (Ed.), proceedings of a symposium-Salmon and Trout in Streams. Fisheries institute, university of British Columbia. 621 p.

Hansaker, C.T. and D.A. Levine. 1995. Hierarchical Approaches to the Study of Water Quality in rivers. *BioScience* 45: 193-203

Harrison, A.D. and H.B.N. Hynes. 1988. Benthic Fauna of Ethiopian Streams and Rivers. *Arch. Hybiol. Suppl* 81: 1-36

Hartman, G.F., J.C. Scrievener, M.J. Miles. 1996. Impacts of Logging in Carnation Creek, a High-Energy Coastal Stream in British Columbia, and aheir Implications for Restoring Fish Habitat. *Can. J. Fish. Aquat. Sci*. 53 (suppl. 1): 237-251.

Hauer, F.R. and V.H. Resh. 1996. Benthic Macroinvertebrates. Pages 339-365 in F.R. Hauer and G.A. Lamberti (eds.). *Methods in Stream Ecology*. Academic Press, San Diego. 499p.

Hesse, L.W., G.R. Chaffin and J. Brabander. 1989. Missouri River mitigation: A sytem Approach. *Fisheries (Bethesda)* 3: 11-15.

Hynes, H.B.N. 1960. *The Biology of Polluted Waters*. Liverpool University Press, Liverpool. 202 p.

Hynes, H.B.N. 1975. The Stream and its Valley. *Verh. Int. Ver. Limnol.* 19: 1-15

Jacobsen, D. and A. Encalada. 1998. The Macroinvertebrate Fauna of Ecuadorain Highland Streams in the Wet and Dry Season. *Arch. Hydrobiol.* 142: 53-70

Jones, C., R.M. Palmer, S. Matkaluk and M. Walters. 2002. *Watershed health Monitoring: Emerging technologies*. C.R.C. Press, Boca Raton, Florida. 463p.

Kammerbauer, J. and C. Ardon. 1999. Land use Dynamics and Landscape Change Pattern in a Typical Watershed in the Hillside Region of Central Hondorus. *Agriculture, Ecosystems and Environment.* 75: 93-100

Karanja, A.K., S.S. China and P. Kundu. 1986. The Influence of Land use on Njoro River Catchment between 1975 to 1985. In proceedings of the third national workshop on soil and water conservation in Kenya, Nairobi. pp 355-370.

Karr, J.R. 1999. Defining and Measuring River Health. *Freshwater Biology* 41: 221-234

Karr, J.R. and D.R. Dudley. 1981. Ecological Perspectives on Water Quality Goals. *Environ. Manage*. 5:55.

Leichtfried, M. and W.A. Shivoga. 1995. The Njoro River-Lake Nakuru Ecotonal System in Kenya. *Jber. Biol. Stn Lunz* 15*: 67-77*

Magana, A.E.M. 2001. Litter Input from Riparian Vegetation to Streams: a Case Study of the Njoro River, Kenya. *Hydrobiologia* 458: 141-149.

Manly, B.F. 1994. *Multivariate Statistical Methods: A primer*. 2nd Ed. Chapman and Hall, London. 200 p.

Mathooko, J.M. 1995. Retention of Plant Coarse Particulate Organic Matter at the Surface of the Wet-store and Dry-store Zones of the Njoro River, Kenya. *Afri. J. Ecol.* 33:151-159.

Mathooko, J.M. 2001. Disturbance of a Kenya Rift Valley Stream by the Daily Activities of the Local People and their Livestock. *Hydrobiologia* 458: 131-139.

Mathooko, J.M. and K.M. Mavuti. 1992. Composition and Seasonality of Benthic Invertebrates and Drift in the Naro Moru River, Kenya. *Hydrobiologia* 232: 47-56

Mathooko, J.M. and T. Kariuki. 2000. Disturbance and Species Distribution of Riparian Vegetation of a Rift Valley Stream. *Afr. J. Ecol.* 38:123-129.

Mathooko, J.M., C.M. M'Erimba and M. Leichtfried. 2000. Decomposition of Leaf Litter of *Dombeya goetzenii* in the Njoro River, Kenya. *Hydrobiologia* 418: 147-152.

McDowall, R.M., E. Graynoth, and G.A. Eldon. 1977. The Occurrence and Distribution of Fishes in Streams Draining the Beech Forests of West Coast and Southland, South Island, Newzealand. *Journal of the Royal Society of NewZealand*. 7:405- 24.

Meybeck, M. 1992. Carbon, Nitrogen and Phosphorus Transport by World Rivers. *Am. J. Sci*. 282: 401-450

Minshall, G.W. 1981. Structure and Temporal Variations of the Benthic Macroinvertebrate Community Inhabiting Mink Creek, Idaho, USA., a Third Order Rocky Mountain Stream*. J. Freshwat*. *Ecol*.: 1: 13-26.

Morris, C.C., J.A. Sawyer IV, H.H. Bennet and C.D. Robinson. 2003. Effects of Sediment Quality on the Health of Aquatic Ecosystems; a Case Study on the Depth of fines in Coastal Plain Streams in Alabama. Pages 113-124 in T.P. Simon (Ed).*Biological Response Signatures: Indicator Patterns using Aquatic Communities.* CRC Press, Boca Raton. 576 p.

Newbold, J.D., D.C. Erman, and K.B. Roby. 1980. Effects of Logging on Macroinvertebrates in Streams with and Without Buffer Strips. *Can. J. Fish. Aquat. Sci*. 37:1076-1085.

Ormanik, J.M. 1977. Nonepoint Source-Stream Nutrient Level Relationships: A Nationwide Study. EPA-600/3-77-105. US Enviromental Protection Agency, Corvallis, OR. 87 p.

Richards, C. and G.E. Host. 1993. Identification of Predominant Environmental Factors Structuring Stream Macroinvertebrate Communities within a large Agricultural Catchment. *Freshwater Biology* 29: 285-294

Rosenbereg, D. and V. Resh. (Eds). 1992. *Feshwater Biomonitoring and Benthic Macroinvertebrates*. Chapman and Hall, New York. 488 pp.

Roth, N.R., J.D. David and D.L. Erickson. 1996. Landscape Influences on Stream Biotic Integrity Assessed at Multiple Spatial Scales. *Landscape Ecol.* 11: 141-156

Savage, A.A. 1989. Adults of the British Aquatic Hemiptera Heteroptera: A key With Ecological Notes. Freshwater Biological Association-*Scientific publication* No. 50. 176p.

Scruton, D.A. and R.J. Gibson. 1995. Quantitative Electrofishing in Newfoundland and Labrador: Result of Workshops to Review Current Methods and Recommends Standardization of Techniques. *Can. Manuscr. Rep. Fish. Aquat. Sci. 2308*: vii+145pp.

Shivoga, W.A. 1999. Spatio-temporal Variations in Physico-chemical Factors and Stream Fauna Abundance and Composition in the Zones of Transition between Baharini Sprinkbrook, Njoro River and Lake Nakuru, Kenya. Ph.D. Thesis, Vienna University, Austria. 102 pp.

Shivoga, W.A. 2001. The Influence of Hydrology on the Structure of Invertebrate Communities in two Streams Flowing into Lake Nakuru, Kenya. *Hydrobiologia* 458: 212-130.
Simon, T.P. 2003. Biological Response Signatures: Toward the Detection of Course-and- effect and Diagnosis in Environmental disturbance. Pages 3-12 in T.P. Simon (Ed). *Biological Response Signatures: Indicator Patterns using Aquatic Communities.* CRC Press, Boca Raton. 576 p.
Southwood, T.R.E. 1978*. Ecological Methods*. Chapman and Hall, London. 305 p.
Statsoft *Inc.* 2001. STATISTICA for Windows. Tulsa, OK: Statsoft inc., http://www.statistica.com
Stewart, P.M., E.L. Garza and J.T. Butcher. 2003. Effects of Contaminated Dredge Spoils on Wetland Plant Communities: a Literature Review. Pages 99-112 in T.P. Simon (Ed). *Biological Response Signatures: Indicator Patterns using Aquatic Communities.* CRC Press, Boca Raton. 576 p
Townsend, C.R., A.G. Hildrew and K. Schofield. 1987. Persistence of Stream Invertebrate Communities in Relation to Environmental Variability*. Journal of Animal Ecology*. 56:597-613.
Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, C.E. Cushing. 1980. The River Continuum Concept. *Can. J. Fish. Aquati. Sci.* 37:130-137.
Van Someren. 1952. The Biology of Trout in Kenya Colony. River Research and Development Centre, Game Department, Kenya Colony.
Wallace, I.D., B. Wallace and G.N. Philipson. 1990. A Key to the Case-bearing Caddis Larvae of Britain and Ireland. Freshwater Biological Association, *Scientific Publication* No. 51. 240 p.
Walling, D.E. and B.W. Webb. 1992. Solutes in River Systems. Pages 251-327. In: S.T. Trudgill (ed.). *Solute Processes*. John Wiley & Sons, Chichester.
Wang, L. and J. Lyons. 2003. Fish and Benthic Macroinvertebrate Assemblages as Indicators of Stream Degradation in Urbanizing Watersheds. Pages 227-250 in T.P. Simon (Ed). *Biological Response Signatures: Indicator Patterns using Aquatic Communities.* CRC Press, Boca Raton. 576 p
Wang, L. J. Lyons, P. Kanehl, and R. Gatti. 1997. Influences of Watershed Land Use on Habitat Quality and Biotic Integrity in Wisconsin Streams. *Fisheries* 22: 6- 12.
Weaver, L.A. and G.C. Garman. 1994. Urbanization of a Watershed and Historical Changes in Fish Assemblages. *Trans. Am. Fish. Soc.* 123: 1962-1972
Wells, F., L. Metzeling and P. Newall. 2002. Macroinvertebrate Regionalisation for Use in the Management of Aquatic Ecosystems in Victoria, Australia. *Environ. Monit. Assess*. 74: 271-294.
Wetzel, R.G. 2001. *Limnology: Lake and River Ecosystems.* 3rd. Academic Press, San Diego. 1006 p.
Wohl, N.E. and R.F. Carline. 1996. Relations Among Riparian Grazing, Sediment Loads, Macroinvertebrates in Streams With and Without Buffer Strips*. Can. J. Fish. Aqaut. Sci*. 37: 1076-1085.
Yankey, R., S. Potter, T. Maret, J. McLaughlin, D. Carter and C. Brockway. 1991. Rock Creek Rural Clean Water Program: Ten Year Report. U.S. Department of Agriculture and Twin Falls and Snake River Soil Conservation Districts, Twin Falls, Idaho.
Yoder, C.D. and E.Y. Rankin. 1998. The Role of Biological Indicators in a State Water Quality Management Process. *Environ. Monit. Assess*. 51: 61-88.
Zar, J.H. 1996. *Biostatistical Analysis*. 2nd Ed. Prentice-Hall, Englewood Cliffs, N.J. xiv, 718p.

Significance of land use changes on African Lakes

Kigamba Judy Njoki

Ministry of Aater and Irrigation, Provincial Water Office, P.O.Box 1343 – 10100, Nyeri, Kenya.
Email: judykigamba@yahoo.com

Abstract

There are several lakes in the African continent whose sizes vary. The quality of their water range from saline to fresh waters. The geographical location of a lake has determined the benefits it offers to the community in its proximity. The lakes have social and economic benefits. In most African countries, the population has been increasing inversely to both economic growth and general development of their respective countries thus subjecting the bigger percentage of the population to a vicious cycle of poverty. Under these circumstances, the natural resources come in handy for the survival of the poor and needy with some of them resulting to the lakes and their basins. Overtime the traditional land uses have degenerated evolving into an unsustainable use and poor management of the lakes. Resultantly, the lakes have been exposed to a number of problems including loss of biodiversity, sedimentation, pollution, eutrophication, over abstraction of water and invasion by exotic species. This situation is raising concern regarding the restorative and ultimately sustainable management of the lakes and their basins. For these management interventions to bear fruit, the populations benefiting from them would have to actively participate. Institutional support, sound policies and relevant technical knowledge are also important. Discussed in the paper is change of land uses, their impact on the lakes, restorative and sustainability management of lakes. In particular there is a case study on lake Ol'Bolossat.

Key words: community, management, sustainable.

Introduction

The total amount of fresh water available to human beings on earth (in lakes and reservoirs) is only 0.26 percent of the planet's water (UNEP, 1994), hence the great importance attached to this comparatively scarce water resource. Water supply is therefore one of the limiting factors necessary for advanced economic development, as it plays an important role in various sectors of society. The African continent is endowed with quite a good number of lakes, e.g. the Great Lakes of the African Rift Valley that include among others lake Victoria, Nyassa/Malawi, Tanganyika, Nakuru. They come in all sizes; big and small which are either saline or freshwater lakes. The lakes receive their water mainly from precipitation, some are dominated by drainage runoff, and others are controlled by groundwater systems. They are of great social-economic value to the riparian communities and the respective countries, which benefit from them. The Great Lakes of East Africa are of special interest to culture because the region is believed to be the cradle of human evolution. Olduvai Gorge in Northeastern Tanzania is home to the remains of Zynjanthropus dating back 1.75 million years. African lakes are sensitive both climatic and land use changes in their neighboring landscape. The waters of the lakes are used for purposes like hydroelectric power generation e.g. Kariba dam, domestic, recreation, they offer great potential for tourism, fisheries, agriculture, industrial, and transportation. Consequently, there have been great changes in the water level of most African lakes, and greater changes in the level of smaller bodies of water such as Lake Naivasha, Lake Chilwa.The bigger lakes e.g. Victoria, and Tanganyika were seriously damaged by the wet years of the early 1960s. Lake Malawi rose to damage its principal fisheries laboratory at the same time. Lake level fluctuations vary with the water balance of the lake and its catchment. It is now generally appreciated that lakes cannot be understood in isolation of the drainage basin. The lake water ecosystems have been poorly understood and are under increasing pressure and intensified anthropogenic use.

Change in the use of Lakes and their drainage basin

The African continent is vast and has varying climatic conditions with the lakes interspersed within. Most of the lakes and the wetlands are openly accessible and not protected, thereby increasing the intensity of the forces that threaten their wholeness. The dominant form of land use in most of the drainage basins and the wetlands are agriculture and livestock keeping. In the lakes, both subsistence and commercial fishing depending on the locality of the lake dominate it. Comparatively the population has grown tremendously and inversely proportionally to the resources available. Consequently the acreage per household has been decreasing. Resultantly, areas of low intensity cultivation and stock farming are being used for intensive activities thereby creating a lot of pressure on the land. Local communities are reclaiming land for settlements, farms, fishing, cattle grazing, urban-industrial establishments and wastewater disposal sites. For example, in Lake Victoria wetland loss is a result of conversion by drainage for three main uses: Agriculture (associated with upstream drainage and agrochemical use), urban-industrial establishments, and wastewater influences.

Problems facing the lakes and their drainage basins

Problems accompanying the change in the land use include among others loss of biodiversity (invasion by exotic species and introduction of non-indigenous species), sedimentation, degradation of the water quality due to pollution, eutrophication, over-abstraction of water, exploitative fishing. In Africa the lake ecosystems are threatened as a result of policies that emphasize development rather than management and conservation. This could partly be due to the limited understanding of the complex dynamics of the lakes ecosystem against the increasing anthropogenic pressure.

Over-abstraction of water: The impoundment and abstraction of waters in the catchment, and hence reduction in the amount of fresh water entering the lakes poses the single greatest threat. The lakes situated in the arid and semi-arid regions (e.g. lake Chad) are critically important to the people who inhabit its shores. In recent years, such wetlands and lakes have come under increasing pressure from drought, plans for water resource projects, and intensified anthropogenic use. Gradually, if the amount of water leaving the lake does not balance with the recharge the lakes eventually decrease in size and at times dry up all together.

Sedimentation: Catchment activities particularly devegetation (biomass harvests), burning, overgrazing and others easily lead to changes in structural components in surface water. The removal of the top cover exposes the land to soil erosion when the rains fall. The soil is also carried away by the wind. Deposition of the soil and larger particles in the lakes reduces the depth of the lake with consequential effect on the biodiversity of the lake ecosystem.

Eutrophication: Soils on the African continent have been subjected to natural environmental degrading processes, such as leaching of nutrients and erosion over a period of time. Resultantly the soil lacks fertility and is therefore more vulnerable to human activity, especially deforestation and its attendant consequences of desertification and soil erosion. Nitrogen fertilizers are often applied at rates greater than plants' requirements and there is nearly always a significant excess of nitrogen beyond that removed from agricultural fields as harvested crops. The rest of the nitrogen is stored in the soil, leached in surface runoff or in groundwater or volatilized to the atmosphere. In the absence of substantial wetland buffer strips or hyporheic denitrification, levels of nitrate in lakes may rise to levels well above those required by freshwater biota, so aquatic communities may be greatly altered by intensive agriculture The water bodies become progressively enriched with nutrients, mainly nitrogen and phosphorous, with a resulting excess production of plant (usually algae) biomass with the ecological functioning becoming grossly disturbed.

Pollution: Increasing use of excessive pesticides, which are transported downstream to the wetland sites and the lakes, high inputs of human waste and effluent from both domestic and industrial activities disposed off in soak-away sewage disposal systems enrich rivers and lakes with organic and inorganic compounds (e.g. heavy metals). Installation of a reticulated sewage system would carry the sewage to the treatment works and avoid pollution of the water bodies. Salinization of the soil is increasing, partially due to irrigation, natural water seepage and evaporation (climate change), and by engineering works such as dams and channels. For example, as a result of diversion into irrigation projects, the volume and extent of the formerly stable lake Ol'Bolossat, has been dramatically reduced; the level has dropped, its volume decreased, and its salinity increased.

Exploitative and unsustainable fishing: Economically, the natural fisheries provide the principal protein source for human inhabitants of the riparian nations and as a source of the much-needed income. The effects of fishing have been compounded by introductions of exotic competitors and predators. An over-exploitative mechanized commercial fishery has gradually replaced sustainable subsistence fisheries. The increases in production brought about by the introduction of exotic species may appear impressive, as they fulfill immediate socioeconomic objectives, but a major question regarding sustainability of the resources remains. For example, the increase in production from Lake Victoria is a result of catches of stocked species; particularly *Lates niloticus* (Nile perch) while the production in both Lakes Tanganyika and Malawi has declined. The fish populations in the lakes have suffered declines recently from drought, over fishing, diversion or blockage of in stream flows, and increased juvenile catch due to use of smaller mesh.

Loss of Biodiversitty: Degradation of the water ecosystem adversely affects the floral and fauna in the lake. Indigenous species are seriously compromised and often driven to extinction with the proliferation of exotic species. Good examples are: the proliferation of the water hyacinth and a food web dominated by the exotic species, Nile Perch in lake Victoria, The bird life in lake Chad is threatened by decreasing water levels. Recent concerns include the availability of nesting sites for the endangered West African subspecies of black-crowned crane (*Balearica pavonina pavonina*) and adequate wintering grounds for intercontinental migrants such as the ruff (*Philomachus pugnax*) (World Bank 2000, Meine and Archibald, 1996).

Sedimentation: This is attributed to poor land use practices in the catchment/watershed area, including unsustainable farming practices, deforestation, over-grazing which lead to soil erosion and consequent siltation of wetlands and the lakes. It is advisable to retain the land cover and swamplands because they form buffers that provide water quality regulation in receiving waters.

Proposed intergrated management plan

Hydrologically, drainage basins cut across political boundaries that have varying regulations regarding their management. The African continent has the challenge to formulate a viable plan regarding the protection and management of its natural resources with future sustainability and development in the context. The following is applicable to almost all the African lakes.

International drainge basins: The drainage basins cut through international political boundaries, which happen to have differing priorities, policies, management initiatives and capacities regarding the lake basins. These differences in their outlook more often than not create conflicts over the use and subsequent management of the basin. At times the conflicts can become volatile, sparking conflicts between neighboring countries. To embrace sustainable management, would be necessary for them to develop a management approach compatible to all of them.

Participation: The participation of all stakeholders (community, policy makers, scientist, government technical officers etc) at all the levels of the management programme is crucial.. Principally the community would have to identify itself with the drainage basin and how the benefits they draw from the lake region are connected to the sustainability of the same. The level and quality of communal participation in the environmental management would depend largely on the specific context of the community led activities.

Policies: The policies guiding the various sectors (e.g. water, environment, forest, wildlife) in individual countries of the continent do not have an integrated approach in matters pertaining to the sustainability of the environment. Each department addresses its issues in isolation. Eventually this creates conflicts in the administration of the resources. For proper management of the lake basins, there is need to harmonize the policies.

Technical understanding: A deep and knowledgeable understanding of the delicate dynamics of the lake basin is paramount for effective and efficient sustainable management of the same. At the present the lake basin is poorly understood and hence the continued deterioration of the same. A lot of emphasis ought to be put on the study and research pertaining to the lakes and later used to draw the management plans.

Gender and cultural issues: In order to succeed in environmental conservation, it necessary to involve women, the youth and men. The women folk support their families through subsistence agriculture, making baskets, working as casual laborers, working on their shambas etc while their male counterparts immigrate to the towns looking for greener pastures. So it is the women and the youth who interact more frequently with the environment, hence their key role in effective participation. Dissemination and understanding of scientific concepts to the layman is a difficult task. They may not be willing to accommodate the scientific advice. Therefore the process should be approached with a lot of sensitivity with respect to the community's indigenous practices.

Ownership: Most of the lake basins are owned by the government and hence open to access and consequent degradation. The trustees of these areas should develop a basin protection policy, which should be strictly adhered to. This would forestall degradation from the habitual and the occasional environmental abuser.

Legal and institutional sustainability: Naturally, there are those unwilling to abide by the set-up of the appointed institutions. Laws should therefore be enacted to enforce the management strategies spelt out by the participating entities. The bodies entrusted with implementation of the management plan should be life long because the changes take along time to be felt. The healing process of such a basin also takes long.

Financial support: such a programme requires a resource base to operate and bear fruits. Resources to operate can be generated from the levies charged on those carrying out income generating projects, a vote from the central government, organized community groupings, non-governmental organization and any other willing supporter.

A brief on a case study ol lake ol'bolossat

It lies on the border of Central and the Rift Valley Province in Kenya. It is within the swamp in the Upper Ewaso Ngi'ro drainage basin. Its water catchment is within the Nyandarua range, Supuku escarpment and Dundori hills. The lake has an area of 4.2084km^2. Its inflow is from permanent streams (25%), direct runoff (40%), while the rest is assumed from groundwater inflow. It is rich in biodiversity resources of global concern. These include lake, swamps, mudflats, and riverine habitats all being home to a number of living species including water birds, hippos and other wildlife.

Social-economic aspect

The basin is intensively cultivated with both small and large scale farming being practiced. The acreage per family is 2.28ha while the cultivated is about 1.3 acres. Mostly these are families that own land elsewhere more so in the arid areas adjacent to the swamp. During the dry season commercial crops (e.g. tomatoes, spinach, kales, potatoes, cabbage, pigeon peas, French beans, etc) are grown through irrigation and at times the land is left bare. The water is abstracted from the lake and the springs that feed the lake. Ranching, forestry, dry season grazing, fishing, wildlife conservation and tourism are some of the other activities carried out around the lake. The wetland plays a crucial role as a dry season refugia. Both domestic and wildlife animals turn to the swamp to graze and drink water during the dry season. As the wildlife comes to graze, they also eat what the farmer has grown on the farm. This creates conflicts human-wildlife conflict and the farmer feels justified to kill the wild animals. These land uses are aimed at food sufficiency and generation of some income. In the face of the fast growing population, some of the land uses are unsustainable and contribute to environmental degradation and loss of biodiversity. There are urban centers within the close vicinity of the lake; one of them being the Nyahururu town.

Threats to the lake basin

As earlier stated, we cannot isolate a lake from the activities taking place in the drainage basin. Lake Ol'Bolossat is not an exception and currently it is faced with a number of problems. These are eutrophication, loss of biodiversity, pollution and sedimentation, unsustainable farming practices, over-abstraction of water and unsustainable fishing among others. These problems stem from the poor social-economical status of the community leaving in the area. It is further compounded by lake of a management plan for the region and the fact that the local council of Nyahururu holds it in trust. It is therefore open to access by all and open to degradation.

Integrated management plan

An inter-disciplinary team comprising the government ministries and departments, local universities, Non-Governmental organization (NGOS), community Based Organizations (CBOS), local communities, investors and development partners has been formed. The team aims at reducing the challenges and threats that the lake and its catchment are facing. It has formulated a sustainable management plan for the lake basin. Among the issues in the plan, it has identified income generating projects that are vital tools for sustaining conservation and development, poverty alleviation, rural development, contribute greatly to restoration of degraded sites, improve the environmental quality through soil, water, forest and wildlife conservation in the catchment basin. They will promote sustainable agriculture and land use practices and a cleaner environment by advancing effective waste management and reduction of pollution from domestic and agricultural practices. Quality of the natural resource base and the environment will improve once the plan is implemented by all agencies

Conclusions

From the studies done and our own observations, it is evident that the African lakes are fragile dynamic ecosystems that require to be understood more deeply. There existence will always be relative to the respective drainage basins and the world at large. Their natural variability will change in both predictable and unpredictable ways catalyzed by the activities carried out on our environment. The effectiveness and efficiency of our lake basin integrated management plans depends on how we shall regard the environment.

Poverty alleviation, fighting the AIDS scourge and generally raising the social-economic status of its enormous population is a big challenge many African continent. The individual countries and ultimately the whole continent have to come up with a plan, backed by an environment impact assessment study on how well to utilize their natural resource base without degrading it.

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References

Abila, R.O., and Jansen, E.G., 1997. *Socioeconomic of the Lake Victoria Fisheries: From Local to Global Markets.* IUNC East Africa Programme Report No. 2. Nairobi: Central Graphic and Services.

Davis, T.J. ed. 1994. The Ramsar Convention Manual: A guide for the Convention on Wetlands of International Importance especially as waterfowl habitat. Ramsar Convention Bureau, Gland, Switzerland. 207 p.

Frazier, S. 1996. *An overview of the world's Ramsar sites*. Wetlands International Publ.29. 58

Bond, G., and Lotti, R., 1995. Iceberg discharges into the North Atlantic on millennial times scales during the last glaciation. *Science* 267:1005-1010.

Acreman, M.C. and G.E. Hollis, editors. 1996. Water management and wetlands in Sub-Saharan Africa. IUCN, Gland.

Beadle, L.C. 1981. The inland waters of tropical Africa. Longman Group Limited, England.

Gasse, F., S.Juggins & L.B.Khelifa 1995. Diatom-based transfer functions for inferring past hydrochemical characteristics of African lakes. *Paleogeography, Paleolimnology, Paleoecology.* 117: 31-54.

Gierlowski-Kordesch, E & K.Kelts eds. 1994. *Global geological record of lake basins.* Volume 1. Cambridge: Cambridge University Press.
Street- Perrott, F.A. & S.P.Harrison 1985. Lake levels and climate reconstruction. In A.D.Hecht (ed.), *Paleoclimatic Analysis and Modelling:* 291-340. New York: John Wiley and Sons.

Impacts of land use change on the hydrology of the transboundary Mara River

Mati Bancy M.[1], Simon Mutie[1], Patrick Home[1], Felix Mtalo[2] and Hussein Gadain[3]

[1]Jomo Kenyatta University of Agriculture and Technology (JKUAT), Kenya
[2]University of Dar es Salaam (UDSM), Tanzania
[3]United States Geological Survey, FEWS NET, Nairobi, Kenya

Abstract

The transboundary Mara sub-basin across Kenya and Tanzania, drains into the Lake Victoria and ultimately forms the upper catchments of the Nile Basin, and is considered to be one of the more serene sub-catchments of the Lake Victoria basin. This perception is based on the fact that the basin traverses the internationally acclaimed Maasai Mara Game Reserve and Serengeti National Park respectively. Moreover, Mara basin also accommodates natural forests in its water tower on the Mau Escarpment, large-scale mechanized farming, smallholder subsistence farms, communal pastoral grazing lands and some wetlands. However, there is growing evidence of unprecedented land use change in the upper catchments, resulting from deforestation in the headwaters, while current privatization of pastoral lands is attracting immigrants to the watershed causing a population growth reaching 7% annually. This downward trend from the former "pristine" Mara is picking up momentum, causing adverse effects with far-reaching consequences for the long-term sustainability of the natural resource base and livelihoods.

To address these issues, a study is being undertaken jointly by JKUAT, UDSM and UCLAS to quantitatively estimate the interactions between the water cycle, biogeochemical cycles and wetlands in the Mara basin. The study utilizes remote sensing and GIS tools, as well as socio-economic ground truth studies to determine changes in land use/cover, hydrology and hydrochemistry and to construct GIS integrated models for environmental impact assessment (prediction of risks of flooding, erosion and eutrophication) and for planning. Preliminary results show that in just 14 years between 1986 and 2000, agricultural land has increased by 55% through the combined encroachment of forests and savannah grasslands, which have in turn decreased by 23% and 24% respectively. In addition, destruction of closed forests has seen land under tea/open forest increase by 82%. These changes, have affected the hydrology of the Mara, causing sharp increases in flood peaks and reduction in base flows, factors that could not be linked to changes in rainfall amounts and characteristics. The consequent erosion from the catchment has caused sedimentation of downstream wetlands, which have increased by a factor of 131%. This is forcing farmers and livestock to abandon formerly arable lands and grazing lands. Research on developing coping strategies is underway, but even these early results provide evidence for the need for urgent action to save the Mara.

Key words: Land use change, land cover, hydrology, wildlife, rangelands, croplands, forests, wetlands, remotely sensed data, GIS, Lake Victoria, Mara Basin, Kenya, Tanzania.

Introduction

The Mara River Basin, is considered one of the more pristine catchments that drain into Lake Victoria, and is therefore forms part of the upper catchments of the Nile basin. It is a transboundary basin, traversing Kenya and Tanzania as defined geographically by the hydrological catchments that drain into the Mara River. The Mara basin covers 13,750 km^2, lying roughly between longitudes 33^o 47' E and 35 ^{o} 47'E and latitudes 0^{o}38' S and 1^{o}52' S, with the upper 65% area (8941 km^2) in Kenya, while the remaining lower portion is in Tanzania (Figure 1). Altitudes range from 2932 m at its source on the Mau escarpment to 1134 m on Lake Victoria. The 395 km long Mara River originates from the Napuiyapui swamp in the Mau Escarpment in the highlands of Kenya. The main perennial tributaries are the Amala and the Nyangores, originating in the Mau escarpment. The Talek River starts from the Loita plains and joins the Mara in the Maasai Mara Game Reserve. The Engare Engito originates from the Ilmotyookoit Ap Soyet ridges while the Sand River, which is the last main tributary, joins the Mara at the Kenya-Tanzania border in the Serengeti plains, flowing through Mosirori Swamp, finally draining through the Mara bay into Lake Victoria at Musoma in Tanzania. The basin is bound d by the Soit Olooloo Escarpment on the west and the Loita and Sannia plains to the east. Rainfall varies with altitude in the Basin. Mean annual rainfall ranges from 1000-1,750 mm in the Mau Escarpment, 900-1000 mm in the middle rangelands to 700–850 mm in the lower Loita hills and around Musoma. Rainfall seasons are bi-modal, falling between April and September, and again between November-December.

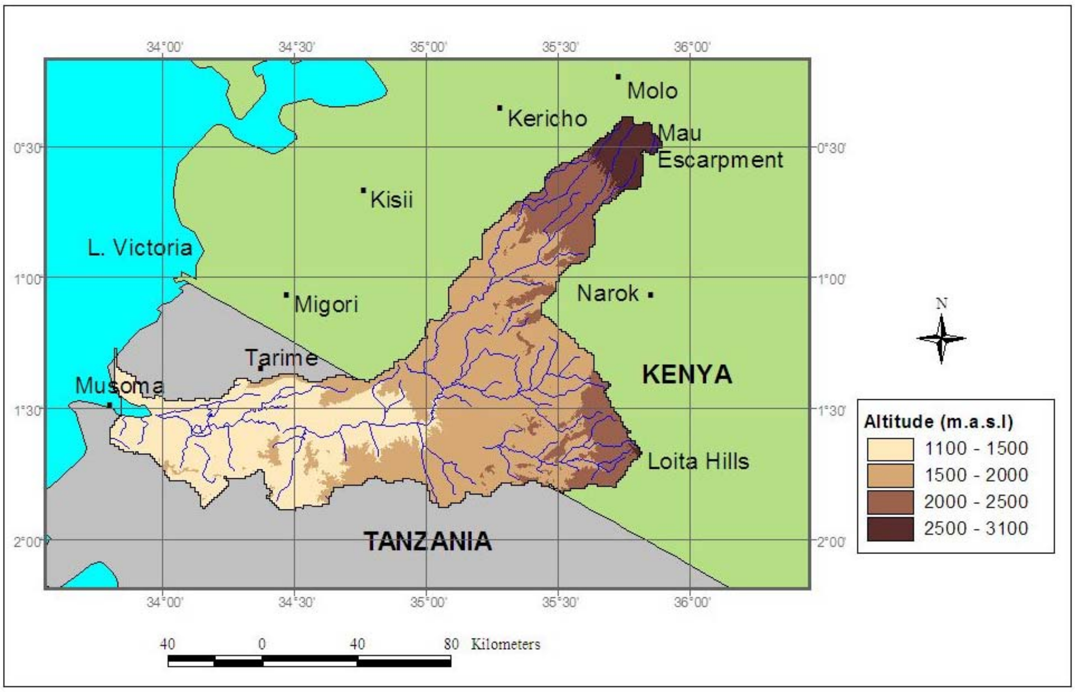

Figure 1. Location and Relief of the Mara River Basin in Kenya and Tanzania.

The Mara-Serengeti ecosystem contains the most diverse combination of grazing mammals in the world, holding 400,000 wildlife and livestock. The River Mara is critical in the unique annual wildbeest migration and for balancing the ecosystem. However, this may not be for long, based on the findings of recent research in the Mara, which points to the fact that there is accelerated loss of vegetation cover in the upper catchments and associated land degradation, which consequently pose a threat to the river flows and the ecosystem (Mati *et al.,* 2005; Machiwa, 2002; Dwasi, 2002; IUCN, 2000; Aboud *et al.,* 2002). Despite the diversity in spatial extent and land use, the dominant social-economic activity to the majority of the population remains crop farming. About 62% of the households are smallholder farmers (Aboud *et al.,* 2002), with livestock rearing being a second dominant activity, yet agriculture occupies about 28% of the available arable land. Tourism and wildlife are important economic activities. At the heart of the Mara basin lie the Maasai Mara Game Reserve on the Kenyan side and the Serengeti National park on the Tanzanian side. The Serengeti is a World Heritage site and a Biosphere Reserve and therefore of global conservation significance. The Mara-Serengeti ecosystem is a world-famous wildlife sanctuary of great economic international importance, supporting a thriving tourism industry. This area is surrounded by nomadic pastoralists, who sell traditional artifacts, while tourist related services provide important additional income for local communities (Thompson, 2002). In the dry season, the Mara River is vital to both tame and wild animals, including the migrating herds in the Maasai Mara and Serengeti national parks. The Mara basin also supports the livelihoods of pastoral people, farmers, fishers, some hunter-gatherers in the forested catchment areas, and other people who directly or indirectly rely on tourism. In addition, the use of forest resources remains an important source of livelihoods to the people in the highlands, while fishing is more important in the areas around the Lake Victoria.

For sustainable utilization of the resources of the Mara, there is a need to obtain reliable data for developing suitable policies and management principles integrated with socio-economic human activities affecting the ecosystems and livellihoods (e.g. agriculture, urbanisation, forests, wetlands, wildlife, national parks). Few historical hydrological and hydrochemical data exist. In this connection, this study was planned with the aim to answer questions such as determining the extent of land use/cover change in the basin, finding out what changes have taken place in the hydrology of the Mara, identifying the effects of these changes, including on the wetlands, and thereby proposing types of interventions that can result in the stabilization the hydrograph of the Mara, and ecosystem restoration.

Major ecosystem threats in the Mara basin

Even though the Mara is among the more pristine of the rivers draining into Lake Victoria, yet there are major threats to the formerly serene ecosystem. Over the last 50 years, the Mara basin has undergone large changes in land cover. Forests and savannah grasslands have been cleared and turned into land for agriculture, charcoal burning, overgrazing and expansion of agricultural activities (Machiwa, 2002; Dwasi, 2002; IUCN, 2000), while grazing resources have dwindled. For instance, the area under cultivation in the Amala sub-catchment increased from less than 20% in 1960 (Olenguruone Settlement Scheme) to more than 51% in 1991. This is partly due to the rapid population growth, as a result of high rates of immigration, as between 1999-2002, the number of households increased by 13% in the upper catchments. The middle catchments of the Mara basin around Mulot, Longisa, Norengore and Kaboson in Kenya used to be sparsely populated rangelands, but in recent years, the district has attracted immigrant farmers. As a consequence, the water cycle has become short-circuited leading to faster runoff which erodes the landscape and causes eutrophication of the receiving wetlands and waters, including Lake Victoria. But data on sediment yield is scarce, estimated at between 113 and 432 tonnes/day (Scheren et al., 2002; LVEMP/COWI, 2002; Machiwa, 2001). These changes have had impacts on riparian wetlands affecting their stabilizing filtering effects (Makota, 2002; Mwanuzi et al. 2002). As more land is opened for crop production, pastoralists are finding it increasingly difficult to support their families and are highly vulnerable to drought. For instance, in 2000, pastoralists lost 35% of their cattle due to drought, while over the last 20 years there has been a decline in wildlife of over 50% (Reid et al. 2003; Ottichilo et al., 2001). The livestock water demand in the Mara catchment has been estimated as 159 m^3yr^{-1} in 1990, 190 m^3yr^{-1} in 2000, and is predicted to rise to 228 m^3yr^{-1} in 2010 (JICA, 1992) and the rising pollution levels, from agricultural activities upstream is threatening wildlife habitats. For instance, local people reported that hippos used to swim certain sections of the river, but now the water can be only knee-high during the dry season, and they are starting to migrate. Field surveys revealed that at Keekorok where the Mara leaves Kenya into Tanzanian, the river crosses the border heavy laden with sediment. In addition, the locally driven degradation has increased the vulnerability of thousands of families who have no alternative incomes (Thompson, 2002).

Smallholder farming, growing subsistence crops like maize, sorghum and cassava, forms the predominant economic activity in the lower reaches of the Mara basin, including around the Mosirori wetland lying at the lowest end towards Lake Victoria. Here, the upper hills have been deforested mostly through charcoal burning resulting in massive land degradation. Major environmental changes have been observed in recent years including scouring of the riverbed and silt build-up near the Lake causing back-water flow from the lake up to 40 km inland. Moreover, the wetland has increased in area affecting the ecology and economic activities. Field studies revealed that local people at Bisarwi can recall walking across the wetland to the other side of the river Mara about 20 years ago, and also grazing livestock on the lowlands. By 2004, the entire valley was so waterlogged that livestock grazing and cultivation were impossible. The increased erosion from Kenya has brought rich alluvial soils, which are cultivated without the need for fertilizers on the edges of the wetland. In general, land degradation upstream has caused increases in the spatial extent of the wetland, as well as excessive flooding of the river during the rainy season. In addition, the water resources of the Mara are inefficiently utilized, resulting in lost development opportunities, low economic growth, rural poverty and upstream-downstream conflicts in the Basin (Sweco, 2003), calling for new approaches in addressing these constraints. There is a need to discuss hydrological impacts due to land use change since the introduction section focused only on the basic features of the basin and activities taking place in the basin. Jumping from that aspect to hydrology directly might mislead the reader.

Data and methods

Mapping of land cover change utilized available high spatial resolution snapshots of Landsat TM/ETM, for the years 1986 and the 2000 respectively covering a span of 15 years (We also analysed 1974 MSS data). Image preprocessing of satellite images was carried based on techniques like those of Coppin *et al.,* (1996); Lillesand and Kiefer, (1987).Ttherefore, the data used in this study was geo- referenced and corrected for sensor irregularities. In addition, topography was mapped from the recently released Shuttle Radar Topography Mission (SRTM) data. GPS-supported field surveys for collecting ground data on vegetation, land cover, and participatory socio-economic studies to verify scientific data and capture logical and local knowledge were also done. This information is considered as the ground truth in developing the land cover maps for the time frame considered.

The hydrometeorological data for the basin were acquired from Kenya and Tanzania. Due to gaps in the available records, only stream flow records from Mara Mines gaging station in Tanzania and the Nyangores River (1LA03) at Bomet in Kenya were used. The data span the period 1970 -1991 and 1963-2000 periods for the two stations respectively and were considered appropriate for this study (Figure 2). Daily rainfall data varied in time and space. The data is not evenly distributed throughout the basin. More rainfall stations are concentrated in the Kenyan side while fewer stations are available for the Tanzanian side. Most of the data are available for the recording period of 1960's to late 1990's.

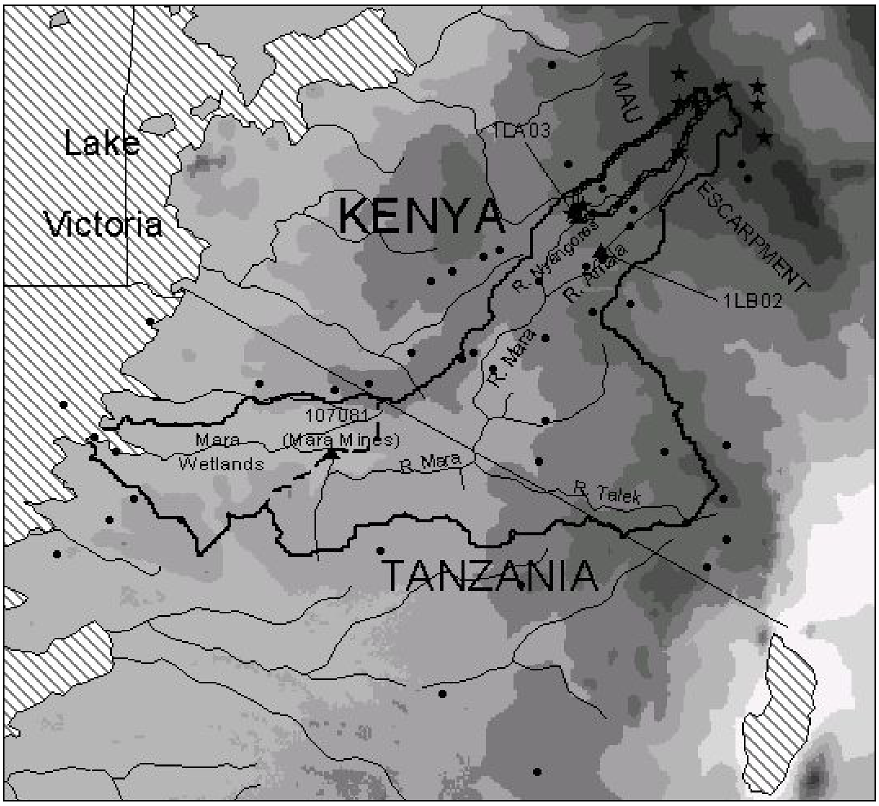

Figure 2: Location of the rainfall and river gauging stations in the Mara Basin.

Data reconstruction is carried out to fill missing gaps. Rainfall time series is filled using seasonal mean values for the specific missing dates. For monthly discharges and excess and deficit flow volumes, cross correlations were used between monthly values at the same gauging station. The replacement of missing excess and deficit flow frequencies uses both the behaviour of the time series and correlation analysis. Cross correlations were used whenever the filling by this procedure was not possible particularly for the cases of fluctuating series or when the flows before and after the gap are on the opposite sides of the threshold (Valimba *et al.,* forthcoming). The variable quality, length and period of rainfall records necessitated their selection. Rainfall records were further required to span the period encompassing that of flow record. Therefore only 10 records with 30 or more years of data and which started before or in 1961 and continuing through the early-1990s to 2002 were retained and the criterion of less than 10% of missing monthly values finally retained only 9 stations for the inter-annual variability analysis. However, monthly discharges were determined only for months with at least 90% of the daily observations available while monthly indices of excess and deficit flows as well as rainfall amounts were determined only for complete months. The frequency and severity of floods and drought in the Mara River basin were studied using excess and deficit flows as well as maximum and minimum flows. The excess (deficit) flow frequency was defined as the number of days above (below) the threshold defining the flood (drought) flow while excess (deficit) flow volume was the cumulative flow volume above (below) the respective thresholds (Valimba *et al*., forthcoming).

Results and discussion

Land use/cover change

The respective land use/cover (LULC) as derived from Landsat TM/ETM data for 1986 and 2000 are shown in Figures 3 and 4 respectively. The spatial coverage of each land use and land cover class may be visualized on both maps. In general, the major land use/covers in the Mara Basin include closed forest, open forest and tea in the upper mountain slopes, agricultural land, shrublands and grasslands used for grazing or as game reserves, savannah grasslands, which comprise shrub-grasslands, and wetlands. Figure 5 shows the land use/cover change in the Mara basin between 1986 and 2000 based on the analyses for the Landsat imagery, and the spatial extent of these changes have been shown in Table 1. From these maps, it emerges that, the Mara Basin is predominantly a rangeland, whereby in 1986, about 69% (9,594 km^2) of the land was under natural pasture, as savannah, grasslands or shrublands, mostly used for grazing livestock and/or wildlife reserves. However, by 2000, these rangelands had been reduced by 24% to only 7,245 km^2 due to encroachment by agriculture, whose area has increased by 55%. Similarly, except for the water body, all the other land use/covers have undergone change in the 15 years under review (Table 1). The natural vegetation has been declining as closed forests reduced by 23%. due to forest clearing for tea and/or as timber harvests, which have increased opened land by 82%. Meanwhile, downstream wetlands especially the Mosirori in Tanzania have increased in area by a factor of 131%. This increase has been associated with the build up of sediments downstream, as a result of erosion in the catchment due to high water velocities and high peak flows, which in turn has been caused by reduction in the vegetation cover due to deforestation, opening of natural rangelands to agriculture, and poor soil conservation efforts.

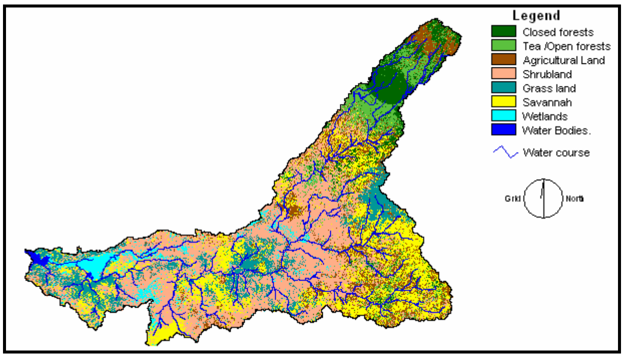

Figure 3. Land use/cover in the Mara basin in 1986 (from Landsat TM).

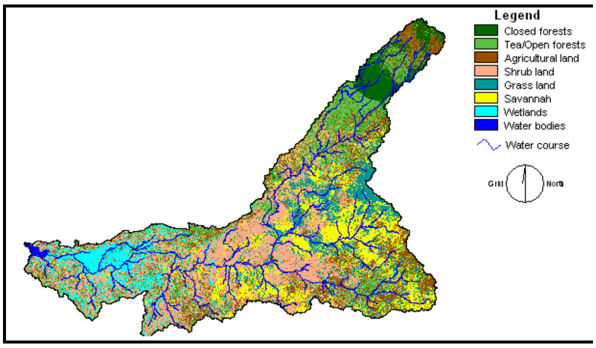

Figure 4. Land use/cover in the Mara basin in 2000 (from Landsat ETM).

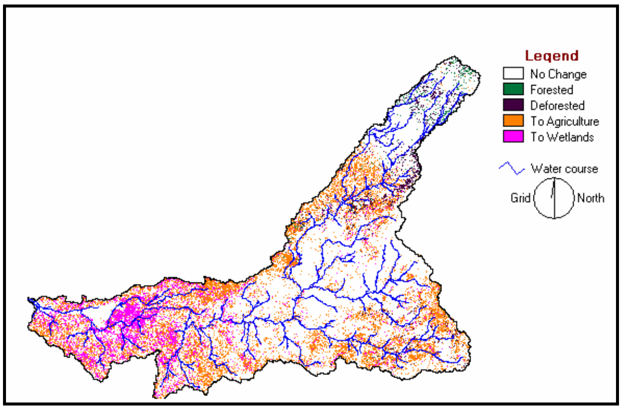

Figure 5. Land use/cover change in the Mara basin between 1986 and 2000.

Table 1: Extent of Land use\cover changes in the Mara Basin between 1986 and 2000.

Land use\cover types	1986(km^2)	2000(km^2)	LULC Km^2	LULC (%)
Closed forest	892.7	688.6	-204	-23
Tea/Open forest	1072.9	1948	875	82
Agricultural land	1617.4	2503.5	886	55
Shrub and grassland	9593.8	7244.5	-2349	-24
Wetland	603.6	1394.4	791	131
Water body	54.2	55.6	1	3

Changes to the hydrological regime

Statistical methods of change-point analysis indicated significant changes of the flow regimes in the Mara River basin which have mainly occurred in the 1973-1977 period. These changes correspond mainly to increasing flows in December, April and May (Table 2). However, the mean flow increases in these months at the most downstream gauge (Mara Mines) are mainly caused by increases in the high (excess) flows while low flows are not significantly affected. Furthermore, there are indications of decreasing flows during the low flow (February and March) period as indicated by increasing frequencies and volumes of deficit flows (Table 2). The changes were more evident in the upstream catchment of the Nyangores than at Mara Mines gauge. The results further indicated a reduction of the frequency of flows exceeding the flood flow index in June at the Nyangores gauge. The implication of increasing flood flows in April and May and decreasing in June could be that the flood flows have become voluminous and slowly attenuates than they used to be. Consequently, the flood characteristics are hypothesized to have undergone changes to peaked behaviour. Thus, less time is available for infiltration of flood water to recharge the ground water and this could be responsible for the decreased flows during the low flow period (e.g. February-March) indicated by increasing volumes and frequencies of deficit flows (Table 2).

The indices characterising the high and flood flows (flow maxima, excess flow frequencies – EFF and volumes – EFV) indicated a significant increase of high flows in December and April and slightly in May around 1977. The increases characterise both the averages (Table 2) and variances. At the threshold of average plus half standard deviation flow of the series of flow maxima in each month, the monthly maxima in April and December at Mara Mines, for example, have exceeded these thresholds by 6-94% and 46-100% respectively. The record maxima occurred in 1988/89. No exceedence was observed in December in the pre-1977 while all the four exceedences were observed after 1977 while five exceedences in April characterised the post-1977 compared to a single exceedence in the pre-1977. At the Nyangores, the increase characterised only April and May. Moreover, EFV and EFF have increased since 1977 and characterised mainly these indices in December, April and May (Nyangores) and December (Mara Mines). In the post-1977, flows in April and May in the catchment of the Nyangores frequently and abundantly exceed the flood flow threshold in these two months suggesting that floods are becoming more frequent and severe in the basin. The changes in the high flows suggest some modification of the characteristics of high flows. The disappearance of EFF in June since the late-1960s in the catchment of Nyangores (Table 2) while the increase characterise EFF and EFV in May indicate that high flows are become less persistent in this catchment in recent decades. This further suggests a shortening of the duration of floods resulting rather into flashy-like floods.

Isolated intense rainfalls (exceeding 40mm) and persistent heavy (30-39.9mm) rainfalls for several consecutive days were hypothesized to be instrumental to river flooding (Valimba et al., forthcoming) The short-duration heavy rainfall intensities normally result in voluminous direct runoff volumes due to little time available for infiltration. As a consequence, there will be insufficient recharge of groundwater leading to a decrease of dry season flow resulting sometimes into prolonged and severe droughts. Such flow regime changes are hypothesized to be driven either by rainfall changes or artificial influences that altered the surface of the catchment and consequently its runoff production mechanisms. Therefore, analyses of indices of rainfall are expected to highlight the potential influences of changes of rainfall on the identified flow regime changes. The changes of land cover due to changing socio-economic activities in the catchment could independently lead to such flow regime changes or intensify the changes induced by changing rainfall characteristics.

Table 2: Summary of changes in indices characterizing flow regimes of the Mara. EFV = excess (flood) flow volume, EFF = excess flow frequency, DFV = deficit (drought) flow volume, DFF = deficit flow frequency (empty cells indicate discontinuities in data).

1LA03 (Nyangores)	Dec			Feb			Mar			Apr			May			Jun		
Monthly flow index	**Date**	**Before**	**After**	**Date**	**Before**	**After**	**Date**	**Before**	**After**	**Date**	**Before**	**After**	**Date**	**Before**	**After**	**Date**	**Before**	**After**
Mean (m3/s)							**1970**	4.8	2.2	**1976**	5.6	10.1	**1976**	9.2	15.3			
EFV (m3/s/d)	**1981**	0.2	8.9							**1977**	10.3	28.8	**1977**	4.5	16.5			
EFF (days)	**1980**	0.1	2.3							**1976**	2.7	5.6	**1976**	1.3	6.6	**1969**	2.6	0.4
DFV (m3/s/d)	**1980**	25.8	16.6	**1979**	34.7	49.5	**1979**	37.6	58.7	**1976**	25.3	11.6	**1976**	10.2	0.0	**1974**	5.5	0.0
DFF (days)	**1975**	18.7	11.6				**1969**	11.5	24.3	**1975**	14.9	8.3	**1975**	7.3	0.0	**1973**	3.0	0.0
Mara Mines	Dec			Feb			Mar			Apr			May			Jun		
	Date	**Before**	**After**	**Date**	**Before**	**After**	**Date**	**Before**	**After**	**Date**	**Before**	**After**	**Date**	**Before**	**After**	**Date**	**Before**	**After**
Monthly flow index																		
Mean (m3/s)	**1977**	6.2	30.7							**1976**	48.3	89.1	**1976**	32.9	73.5			
EFV (m3/s/d)	**1977**	0.0	130.5							**1973**	178.5	772	**1976**	43.6	345.			
EFF (days)	**1977**	0.0	2.3							**1973**	3.3	5.9	**1976**	1.0	5.4			
DFV (m3/s/d)																		
DFF (days)																		

Conclusions

The Mara basin is a complex agro-ecosystem in terms of natural and demographic characteristics, and due to its trans-boundary nature, solutions to the problems facing the basin require a joint effort between the two countries, Kenya and Tanzania as was done in this study. The study has produced baseline and derived thematic data on land use/cover as well as rainfall and river flows to enable assessment of their effects on the hydrology of the basin. The results have shown that although the major land use/cover type in the Mara still remains rangelands, comprising savannah grasslands or shrublands, mostly used for grazing livestock and/or wildlife reserves. However, agricultural encroachment is taking its toll, reducing these lands from 9,594 km^2 in 1986 to 7,245 km^2 in 2000, a net decline of 24%. Similarly, forest cover has been reducing by 23% while the open forest and/or tea areas have increased by 82% and agricultural lands holding field crops have increased by 55%. One major observation has been the increase in wetlands and the lower reaches of the Mara, at, especially the Mosirori wetland in Tanzania, which has increased in area by a factor of 131%. The increase in wetlands has been associated with backwater flow from Lake Victoria due to sediment build up downstream, resulting from soil erosion in the upper catchments. This in turn has been caused by reduced vegetation cover due to deforestation, overgrazing of the dwindling rangelands, farming on hilly slopes and poor soil conservation efforts.

The flow regimes in the Mara River and its tributaries have also experienced significant changes since the mid-1970s, resulting in increasing high flows particularly during the long rains months of April and May, and corresponding low flows in January to March. The local people have reported that hippos used to swim certain sections of the river, but these days, the water can be only knee-high during the dry season, forcing the animals to migrate. In general, land degradation upstream has adversely affected the water resources of the Mara, which are also inefficiently utilized. This translates to lost development opportunities, low economic growth, rural poverty and upstream-downstream conflicts in the Basin. The water cycle has become short-circuited leading to higher runoff flows, which can reach over 89 m^{3}s^{-1}. As more land is opened for crop production, pastoralists are finding it increasingly difficult to support their families and are highly vulnerable to drought. In addition, the locally driven degradation has increased the vulnerability of thousands of the wildlife and posing a risk to the sustainability of their natural habitats. This calls for new approaches in addressing land use policies and how they affect land allocation, planning, utilization and management in the Mara at local, national and sub-regional levels.

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References

Aboud, A.A., Obweyere, G.O., Mutinda, M.M. and Raini, J.A. 2002. *A Rapid Participatory Socioeconomic Assessment of the River Mara Basin.* Mara River Catchment Basin Initiative WWW-Eastern Africa. Regional Programme Office (EARPO).WWF, Nairobi.

Coppin, P. & Bauer, M. 1996. Digital Change Detection. *In: Forest Ecosystems with Remote Sensing Imagery.* Remote Sensing Reviews. Vol. 13. p. 207-234.

Dwasi, J.A. 2002. Trans-boundary Environmental Issues in East Africa: An assessment of the Environemntal and Socio-economic Impacts of Kenya's Forestry Policy on Tanzania. Manuscript.

IUCN, 2000. *Forest cover and forest reserve in Kenya*. Policy and practice. Working paper No 5.

JICA (Japan International Cooperation Agency) 1992. *The Study of the National Water Master Plan.* Japan International Cooperation Agency. Ministry of Water Resources and management, Nairobi.

Lillesand, T.M. and Kiefer, R.W. 1987. *Remote Sensing and Image Interpretation*. John Wiley & Sons. New York.

LVEMP/COWI, 2002. *Integrated Water Quality/ Limnology Study for Lake Victoria.* Consultant Technical Report. Dar-es-salaam

Machiwa, P. 2001. *Assessment of Pollution Loading into Lake Victoria from Terrestrial Sources: The Case of Phosphorus*. A paper presented at The National Technical Workshop in Mwanza on 6 th-10 th, August, 2001

Machiwa, P. 2002. Water Quality Management and Sustainability: In: *The experience of the Lake Victoria Environmental Management Project (LVEMP).* 3rd WaterNet/Warfsa Symposium 'Water Demand Management for Sustainable Development', Dar es Salaam, 30-31 October 2002.

Makota, V. 2002. *Application of remote sensing techniques for Wetland resource monitoring part of Magu in Mwanza region*, LVEMP technical report.

Mati, B. M., Mtalo, F., and Mtalo, G.E. 2005. Use of Geo-spatial Tools for Planning and Sustainable Management of Hydrological Determinants in the Mara Basin. *Proceedings of the 3rd EARSeL Workshop on Remote Sensing for Developing Countries.* National Research Centre, Cairo, EGYPT, 26-29 September 2004. (forthcoming

McCarthy, J., and Gumbricht, T., 2003. Multisource rule-based contextual classification of ecoregions of the Okavango Delta, Botswana. *Remote Sensing Reviews*, In press.

Mwanuzi, F., H. Aalderink and A. Mdamo, 2002. *Simulation of pollution buffering capacity of wetlands fringing the Lake Victoria using a wetland model,* LVEMP technical report.

Ottichilo, W.K., de Leeuw, J., Skidmore, A.K., Prins, H.H.T., and Said, M.Y., 2001 Population trends of large non-migratory wild herbivores and livestock in the Masai Mara ecosystem, Kenya, between 1977 and 1997. *African Journal of Ecology*, 38, 202-216.

Reid, R.S., Ogutu, J., Rainy, M., Kruska, R.L., Nyabenge, M., McCartney, M., Worden, J., Wilson, C.J., Kshatriya, M., Kimani, K., and N'gan'ga, L. 2003. *Mara Count 2002: People Wildlife and Livestock in the Mara Ecosystem.* Report, Mara Count 2002, International Livestock Research Institute, Nairobi, Kenya.

Scheren, P.A.G.M., H.A. Zanting and A.M. Lemmens, 2000. Estimation of water pollution sources in Lake Victoria, East Africa: Application and elaboration of the rapid assessment methodology, *Journal of Environmental Management*, 58, 235-248

Sweco, 2003. *Preparation Phase of the Mara River Basin Integrated Water Resources Management Project,* Draft Inception Report, Stockholm, 73 p.

Thompson, D.M. 2002 *Livestock, cultivation and tourism: Livelihood choices and conservation in Maasai Mara buffer zones*. PhD Thesis, University college of London.

Valimba, P., Mtalo, F.W. Servat E. and Mati, B.M. forthcoming. *Evidence of changing flow regime in the Mara River basin in Kenya/Tanzania*. Project Report. Civil Engineering, University of Dar es Salaam.

Water quality management of River Nzoia in Lake Victoria basin

Nyongesa, Fred

Central Water Testing Laboratories, Ministry of Water and Irrigation, P. O. Box 55477 - 00200 Nairobi, Kenya
E-mail: fnyongesa05@yahoo.com

Abstract

Pollution and depletion of water resources in a drainage basin results from unpleasant human activities in the basin. The water quality impacts of rivers and lakes include: effects of eutrophication, siltation, oxygen depletion, salinization, acidification, toxic contamination, aquatic ecosystem and turbidity. Majority of people in river Nzoia watershed of lake Victoria drainage basin typically have little or absolute understanding of their roles, individually or collectively, in causing water problems. Further, they may have equally little understanding of what they could do to assist in alleviation of problems. It is noted, poverty of many people in developing countries has restricted or eliminated their options for survival; they will require assistance to change their habits. In other cases, however, it is simply a matter of instructing or educating individuals as to how their possible roles can alleviate environmental pollution problems. The media can be especially useful in this regard, in highlighting the public role in the process of identifying the causes of and the solutions to human-induced water problems, as well as in changing the basic attitude of people towards protection and desired wise use of water resources. As such, proper utilization and management of river Nzoia watershed is of paramount importance for the river is a major tributary to lake Victoria and it transverses through large and small-scale agricultural farmlands, industrial towns equipped with partially operating wastewater treatment plants and urban centers characterized with open-waste-dumping. Behavioral change at the individual, household and community levels is essential and that public participation is mandatory, no longer optional part of effective watershed management. Monitoring, sensitization and sustainable management of river Nzoia watershed is strongly recommended to provide solution to lake Victoria pollution and Budalangi floods.

Key words: Pollution, sensitization, watershed.

Introduction

The damage that is being done on environment by man's activities is irreparable. Moreover, any ill effects from pollution are not confined within borders of polluters. Monitoring and assessing water quality can no longer be confined to local or national boundaries. Large rivers may pass through several nations, major aquifers may transverse national boundaries, and two or more countries may share shores of a large lake. Water, the earth's life support system is a fragile natural resource that has been abused and misused by man.

Rivers and streams are major avenues of transfer of phosphorus and nitrogen to many lakes and reservoirs, and they do integrate the various point and no-point sources of phosphorus and nitrogen within their watershed. Mining of phosphate and the industrial fixation of nitrogen; agricultural, industrial and domestic uses of phosphorus and nitrogen have tremendously increased during the last decades. Other activities of modern societies, such as clearing of forests, extensive cultivation and urban waste disposal, have enhanced transport of phosphorus and nitrogen from terrestrial to delicate aquatic environments. While point and non-point sources of phosphorus and nitrogen contribute to eutrophication, non-point sources often are dominant and present complex management challenges especially in developing countries.

Eutrophication of lakes is a serious environmental problem caused by enrichment with plant nutrients, primarily nitrogen and phosphorus that leads to reductions in water quality. Increases in human populations and in agricultural, industrial and urban developments contribute greatly to eutrophication. The scientific basis for evaluating causes and impacts of eutrophication is encompassed by limnology, and training in limnology should be an integral of the education of everyone responsible for managing lakes.

Lake Victoria and its tributaries

Lake Victoria is situated in East Africa and lies at longitudes 31^o 39'E - 34^o 53'E and latitudes 00^o 3'N - 3^o 00'S. It is the second largest freshwater lake in the world. Lake Victoria basin lies between longitudes 34^{o}E and 36^{o}E and between latitudes 1^{o}15' N and 1^{o}55' S. The total area of surrounding catchments is estimated to be 184,000 km^2 out of which 44,000 km^2 (23%) lie in Kenya, 84,000 km^2 (43%) lie in Tanzania; 32,000 km^2 (16%) lie in Uganda while 33,600 km^2 (17%) lie in Rwanda and Burundi. The entire drainage basin covers an area of 258,000 km^2. The shoreline is irregular and covers 3,450 km out of which 17% is in Kenya, 33% is in Uganda while Tanzania commands 50% of the total shoreline. Rivers are the main lifeblood of lake Victoria. River Nile leaving near Jinja in Uganda to Egypt through Sudan covering about 6,000 km is the only river out-flowing lake Victoria.

The Kenyan side of lake Victoria basin has an area of about 47,164 km^2 and is divided into two catchments areas: Lake Victoria South and Lake Victoria North. While major rivers Nyando, Sondu/Miriu, Gucha/Migori and South Awach drain Lake Victoria South, Lake Victoria North catchment is drained by major rivers Nzoia, Yala, Sio and the North Awach. The total area of the Lake Victoria North catchment is 19,628 Km^2. This is distributed amongst the river catchments as follows: (1). River Nzoia has a catchment area of 12,842 km^2, a length of 355 km and a mean discharge of 118 m^3/s; (2). River Yala has a catchment area of 3,351 km^2, a length of 261 km and a mean discharge of 27.4 m^3/s; (3). North Awach River has a catchment area of 1,985 km^2 and a mean discharge of 3.8 m^3/s; (4). River Sio has a catchment area of 1,450 km^2, a length of 84 km with a mean discharge of 12.1 m^3/s.

Table 1. Morphoedaphic characteristics of lake Victoria.

Characteristic	Measure
Altitude (m.a.s.l)	1,134
Catchment Area (km^2)	184,000
Lake area as percentage of catchment	37
Lake Surface Area (km^2)	68,800
Maximum Width, East-West (km)	240
Maximum Length, North-South (km)	400
Maximum Depth (m)	97
Mean Width (km)	172
Mean Depth (m)	40
Volume (km^3)	2,760
Inflow ($km^3 yr^{-1}$)	29
Outflow ($km^3 yr^{-1}$)	39
Annual Fluctuations in level (m)	0.4 - 1.5
Precipitation ($km^3 yr^{-1}$)	114
Flushing time (yrs)	138
Residence time (yrs)	21
Evaporation ($km^3 yr^{-1}$)	109

Source: S. J. Bulirwa (1998). Lake Victoria wetlands and the Ecology of the Nile Tilapia *Oreochromis niloticus.*

River Nzoia

River Nzoia originates from two high-ground areas of mt. Elgon (4,320 m. a. s. l) and Cherengany Hills. The area receives enough rainfall, which is well spread throughout the year except for three months, from December to February, during which time little or no rainfall occurs. Numerous springs collect to form streams that feed main rivers namely: Kamukuywa, Sosio, Kimilili, Kibisi, Kuywa, Malakisi, Tisi, Lwakhakha, Suam, Kisawai and Kimothon. Generally, rivers are fast flowing with radial to parallel drainage pattern on the upper and mid-slopes respectively. Having been fed by the above rivers from mt. Elgon area, river Nzoia roars and it then drains into lake Victoria after passing through large/small - scale cultivated farms, many coffee factories, Pan Paper factory at Webuye, Nzoia and Mumias sugar factories, many towns with no sewerage systems and if there is one then it is poorly run by Town Councils. This river has been the cause of two big problems: Flooding in Budalangi and pollution of lake Victoria by eutrophication, siltation, acidification, loss of biodiversity, salinization, deoxygenation and toxic contamination.

Flooding

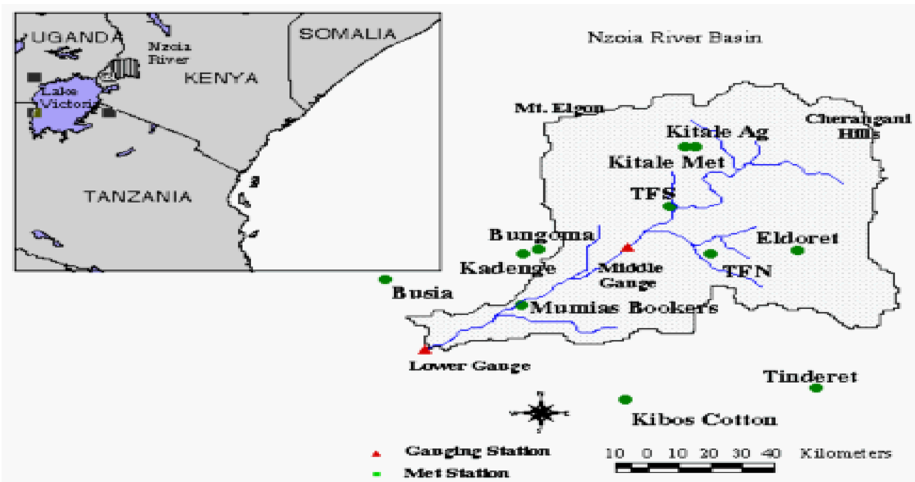
Budalangi division is in Western Province of Kenya. It lies to the north of Lake Victoria near the Kenya-Uganda border. Rainfall pattern in Budalangi is mainly bi-modal (two rainfall seasons in a year). The major season occurs in March to May (the long-rains season) while the other season (short-rains) occurs in October to December. However some areas receive significant rainfall in August and September. The period June-July is generally dry unlike other areas in western Kenya, which observe a major rainfall peak during the period. The months of January and February are also generally dry though occasional wet conditions may occur especially in January.

Figure 1. Rainfall distributions in areas upstream and downstream of river Nzoia. Source: Ranet (Radio Internet Communication Systems) - Kenya Project.

Cause of flooding

Budalangi area has been identified with floods for decades. Hundreds of people die and others get displaced due to floods. As a matter of fact, flooding does not occur due to heavy rainfall in the area and river Nzoia catchment areas. Annual rainfall analyses indicate that the amounts of rainfall in the areas alone may not be enough to cause such floods. Massive water in-flows emanating from the bursting of river Nzoia banks happens to be the main cause of floods. It bursts because its riverbed is raised due to heavy siltation. Mt. Elgon, Cherangany Hills and

Budalangi areas are known to have high rainfall amounts almost throughout the year. They receive average annual rainfall amounts of over 1250 mm while Budalangi area receives an average of about 1100 mm. River Nzoia gathers strength as it flows downstream to an extent of bursting as it reaches the Budalangi areas.



Ways to eliminate the floods

The flooding in Budalangi area can only be arrested by constructing strong and high dikes, which are capable of resisting the strong currents of river Nzoia. The construction of dikes ought to be based on records of highest amounts of rainfall ever recorded especially in the catchment areas of the river. This would help in determining the height of the dikes to be constructed. The other solution would be construction of dams to regulate water level downstream. Aside from run-off-the-river installations, a major function of dam construction is to regularize a river's annual regime by augmenting low-flow periods and greatly reducing periods of flooding in order to make available a more constant water supply for hydropower generation, navigation, source of food e.g. fish farming and commercial irrigation, water sports and related recreational activities.

Pollution of lake Victoria by river Nzoia

In recent past, the catchment areas of river Nzoia has experienced a lot of deforestation due to fertility of the area and good rainfall patterns that has attracted human settlement. This human activity in search for food has resulted in the earth being left bare to be easily carried away by rain. And it is because of this soil erosion that siltation becomes a problem to innocent river Nzoia and resultantly lake Victoria. As river Nzoia gathers strength through agricultural fields, it enriches itself with run-offs of phosphorus and nitrogen as the commonly used chemical fertilizers in these farms are Diammoniumphosphate (DAP) and Calcium ammonium nitrate (CAN). The run-offs also do contain herbicides, pesticides, acaricides and chemicals used in the Greenhouses (Persistent organic pollutants-POPs, Polychlorobenzene-PCB, Polychlorophenol-PCP). As if not satisfied, the river gets polluted with wastewater from moderate urban centers with poor sewerage systems along its course. The factories situated along the river empty their partially treated effluents into the river thereby complicating its chemistry.

Table 2. Annual loads of nitrogen and phosphorus to Lake Victoria from River Basins.

River Basin	Area (km^2)	Discharge (m^3/s)	Total Nitrogen (t/y)	Total Phosphorus (t/y)
Nzoia	12,842	118	3,340	946
Gucha-Migori	6,600	62.7	2,849	283
Nyando	3,652	14.7	520	175
Sondu-Miriu	3,508	40.3	1,374	318
Yala	3,351	27.4	999	102
South Awach	3,156	6.0	322	39
North Awach	1,985	3.8	112	15
Sio	1,450	12.1	248	47

Source: Lake Victoria Environment Management Project, LVEMP 2003.

Human activities that affect water in Mt. Elgon

Human activities have interfered with the natural hydrological cycle. In mt. Elgon, there is a very high environmental degradation as far as water resources are concerned. By 1963, Cheptais and Kopsiro divisions had 45 permanent rivers. In 1975 there were 31 and by 1985, there were 24. There are only 8 that flow throughout the year today. This lack of permanent flow is attributed to many causes some of which are discussed below.

Deforestation

Deforestation of watersheds or catchments by man through logging and for cultivation has led to open bare land being exposed to agents of erosion. This problem has been made worse by the fact that many mt. Elgon residents do not see the forest as part of their heritage that has to be passed down the generations. Thus, they destroy the forest both knowingly and unknowingly. Also, some residents of mt. Elgon consider the forest area to be very fertile, virgin land that they resort to when their legitimate farms depreciate in fertility and require fallow periods to regain their fertility.

Population pressure

Population pressure has led to encroachment into the forest as witnessed in the once expansive indigenous forests in Chepyuk, Chepkitale and Cheptoro. These have been cut down and the land put under crops.

Firewood/timber

Mt. Elgon residents adjacent to the forest see this resource as a store of fuel-wood and source of charcoal. As their parents used to fetch firewood from the forest, they wish to continue this practice, but have also resorted to exploiting the forest to sell fuel-wood on a commercial basis. One of the worst activities in mt. Elgon is the rush for scarce indigenous timber such as Elgon teak. This has depleted large tracts of vegetative cover in the catchment areas. This is likened to Mt. Tanakami Sabo Association, in Otsu City case whereby trees were cut to construct temples in Nara leaving the ground bare and susceptible to erosion (Mother Lake 21 plan).

Pasture

Population pressure has caused reduction of farm sizes thus leading to adoption of intensive farming. As pasture land decreases, communities take their domestic animals into the forest. Many overstock the forest pasture, hence aggravating overgrazing. This promotes loss of soil cover thereby encouraging soil erosion in the watersheds and water points.

Lack of land registration

One problem that has caused the environmental degradation is that Chepyuk is regarded as no man's land. Squatters predominantly use the land unsustainably because they are not sure they will be there tomorrow. There is a need for official excision of land, demarcation of the forest boundary, adjudication, and distribution of title deeds to the residents. This would make people feel responsible for their own lands. Erosion, apart from its direct effect on top soil has siltation effects. This is evident by the brown water seen in most of the mt. Elgon rivers. On assessing the destruction of the catchment area, one former Western Province Commissioner likened the land to an abused woman and remarked thus, "Here is a beautiful lady, Mother of Humanity, yet she has been stripped naked."

Continued deterioration of mt. Elgon water catchments can also be attributed to laxity on the part of law reinforcement agents. Policies such as not cultivating close to the river banks, or not planting blue gum trees along the river have been neglected. Just like in the northern part of lake Biwa, generally, water pollution is not a major problem affecting the water resources in mt. Elgon except on the lower slopes where coffee factories discharge their wastes into the rivers. Water pollution can also be traced at watering point from livestock's discharge of urine and dung into the river. Some residents, due to their cultural beliefs, only trust rivers to wash their clothes and beddings, as a safeguard from witchcraft.

Sustainable management of Mt. Elgon forest and River Nzoia basin

Utilization of forest goods and services by people is indispensable. However, forest resources can be sustainably managed by putting in place measures that will ensure continuity in availability of various goods and services. Problems of river Nzoia and eventually lake Victoria can be solved through collective responsibility as it is of global concern. The objectives will be environmental public awareness, knowledge, attitude/behavior, skills, values/participation.

Sustainable utilization of a resource refers to exploitation without risk of depletion. Inventories need to be carried out to determine the quantity of a particular resource that is available. It is on this basis that the amount of resource to be exploited from the forest can be determined and planned for. Furthermore, the mode of extraction of the resource should be chosen based on a properly analysed Environmental Impact Assessment (EIA) report. Any chosen form of exploitation is subject to abuse unless closely supervised through adherence to well developed and defined guidelines.

Sustainable management of mt. Elgon forest seeks to address problems and issues arising from uncontrolled loss of forest cover. This can be achieved by changing from traditional protectionist approach to that of collaboration with stakeholders of mt. Elgon forest. The following steps should be taken as means towards sustainable management of mt. Elgon forest:

1. Reafforestation: Exotic soft wood plantations of cypress, eucalyptus and pines species formerly occupied most areas on the lower fringes of the reserves. These former plantations have turned into bushland as no reafforestation has been carried out. Forest establishment through a well-supervised non-residential cultivation programme should be encouraged. Other reafforestation activities include enrichment planting to replenish the over exploited natural forest. In this way, patches of open areas within the forest can be covered with forests. Reafforestation can also be initiated by closing off open areas to regenerate on their own with species whose seeds have fallen on that specific site.

2. Afforestation: Mt. Elgon region has open areas that have the potential to support closed canopy forests. Such areas should be cultivated and trees planted on them. These areas should be allowed to naturally regenerate and develop to form a close canopy.

3. Development of alternative land uses: This is by far the best way of easing pressure on mt. Elgon forest. The forest adjacent community should be made to adopt land uses that will enable them to depend less on the forest as a source of wood and other forest products through agroforestry (farm forestry) and zero grazing. Agroforestry is a land use system where trees, crops and livestock are deliberately integrated on the same unit of land. The tree component of agroforestry can also enrich the soils, especially if nitrogen-fixing species like *Sesbania sesban* are used. This practice reduces, in part, the farmers' requirement for fertilizer and can be planted along boundaries, terraces, within paddocks and even within cropland. Agro-forestry trees and shrubs include *Grevillea robusta, Sesbania spp., Leucaena* and *Calliandra spp.* Communities should also be encouraged to start herbal farms through planting of medicinal plants. Cheap alternative energy sources like biogas (animal dung), coffee husks, sugar stocks, saw dust, solar, improved charcoal saving *jikos* and traditional cooking stones. Women groups are particularly effective institutions of disseminating information about conservation of energy and as such these groups should be brought on board; listen to their opinions and criticisms.

Recommendations and conclusions

It is emphasized that raising environmental public awareness is the strongest tool to build up the public support for implementation of environmental action plans. Development programmes on environmental protection risk fail if they do not take into account the experience of the people to whom they are designed to benefit. It is necessary to listen to the people to understand their behavior towards natural resources and to involve them in the decision-making process.

Ultimately, comprehensive implementation of the watershed concept is fundamental to develop sound management strategies. The concept will facilitate integration of stakeholders, partners, users and will stimulate partnerships between the community and Public-private-institutions. The promotion of links to manage water quality and quantity is essential, as well as the interaction between social, biogeophysical, and economical issues at the watershed level. Organization of environmental audit committees and of the implementation of decision support systems is also strategic as a policy for water management.

Water regulations at national (federal) state, municipal and village levels are to be integrated in coordinated actions that ensure functional and sustainable implementation. The improvement of regulations that includes the principle of the charging system for bulk water is fundamental to give sustainability to the water resource management. The enforcement of the legislation at the village level is paramount for sustainability of regulatory mechanisms. Integration of national or federal agencies for water management with state, municipal and local agencies is essential. Coordination at these three institutional systems is fundamental. However, institutional strengthening requires genuine cooperation of various agencies, and delegation of water management at the lowest level (watershed).

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References

Saburo Matsui, 1993. *Environmental Pollution Control. The Japanese Experience, a Historical Perspective*.

Saburo Matsui, 1997. Guidelines of Lake Management, Toxic substances management in lakes and reservoirs; *The International Lake Environment Committee Foundation and United Nations Environment Program,* volume 4.

Masahisa Nakamura, 2000. *[Mother Lake 21 Plan] Lake Biwa Comprehensive Conservation Plan.*

Miki Sudo & Masahisa Nakamura, 2002. *Inflow and outflow of agricultural chemicals in Lake Biwa.*

Victor Muhandiki, 2005. *International cooperation for water resources management: personal reflections, JICA group training course in Lake Water Quality Management.*

Shinji Ide (2005). Fundamental knowledge of water quality Data Processing and Modeling, JICA group training course in Lake Water Quality Management.

Analysis of surface runoff and soil loss under different land use types in River Njoro watershed using a mini rainfall simulator

M.O. Okelo[1], J.O. Onyando[2], W.A. Shivoga[3] and S. N. Miller[4]

[1, 2 &3]Egerton University P.O Box 536, Njoro
[4]University of Wyoming,Rangeland Ecology and Watershed Management, Department 3354,1000 E. University Avenue, Laramie, WY 82071
Email: malafred@yahoo.com

Abstract

River Njoro is the main perennial river draining into Lake Nakuru. There has been an increase in sediment load in River Njoro. This has been occasioned by erosion in the River Njoro watershed caused by the changing watershed management conditions. The objectives of this study were as follows; (i) to assess soil erosion in the various land use types (Agricultural, Grazing, Plantation forest, Indigenous forest, and Deforested area) in the River Njoro watershed (ii) to determine the effects of different land use types on surface runoff. A mini-rainfall simulator was used to study the effects of different land use treatments on surface runoff and soil loss in the River Njoro watershed. The study was done on runoff plots. Soil characteristics; pH, organic matter content, texture and bulk density were established for each land use type. Soil loss decreased in the order agricultural land, deforested land, grazing land, plantation forest and Indigenous forest. The values of mean soil loss were 86, 31, 18, 2, and zero g/0.1 m^2 respectively. The mean values for surface runoff in ml. for the grazing, agriculture, deforested land, plantation forest and indigenous forest were, 1200, 920, 860, 380 and 20 respectively. High runoff values were found to be consistent with high soil losses, especially when there was a generous supply of detached soil particles. Human activities like overgrazing, inappropriate tillage methods and removal of forest / vegetation / litter cover are the likely causes of the relatively high runoff and soil loss in grazing, agriculture and deforested land uses. The major source of soil erosion problem in the watershed was found to be the agricultural land use. Intervention measures to curb the soil erosion problem and reduce surface runoff in the watershed are discussed. These include reforestation and soil conservation measures.

Key words: erosion, rainfall simulator, runoff. Introduction

Introduction

In River Njoro watershed, once forested areas have been replaced by fragmented landscape composed of remnant forest patches, advanced and emerging secondary vegetation, grassland, tree plantations and highly compacted surfaces including roads, paths and housing units. The fragmentation resulted from decades of forest clearing for fuelwood and timber for construction, increased agricultural activities and overgrazing. This was often done in disregard of conservation measures.

These have led to hydrological modifications resulting in increased runoff and soil erosion in the upper parts of River Njoro watershed. The main indicator of erosion in the watershed is silt deposit in flat areas and along the river channel where the gradient is less steep. Sedimentation causes the river to decrease in depth thereby making the water to overflow the banks even in less intense storms and the river to dry up in dry seasons. As a result, water scarcity has been experienced in the lower parts of the watershed in the dry seasons. River Njoro is currently a major contributor to the sediment load in Lake Nakuru, thus threatening the lake's biodiversity. Consequently, the need to investigate areas of the watershed with high potential of runoff and sediment generation became imperative.

The use of a rainfall simulator and runoff plots provides a valuable research tool and are often used in soil erosion and runoff studies. Cheruiyot (1984) used this approach to study infiltration rates and sediment yield in Kiboko, Kenya. The present study used the same method but with a mini simulator (Kamphorst, 1987) to study the effects of different land use treatments on soil loss and surface runoff.

Materials and methods

Study area

The study area is river Njoro watershed ($0^0$15'S, $0^0$25'S, $35^0$50'E, $36^0$05'E) in Nakuru district, Kenya. It is drained by river Njoro, which originates from eastern mau hills (2700 m.a.s.l.) and drains a total area of 250km^2. The river, 50km long, discharges into Lake Nakuru (1700 m.a.s.l.)

The soils and geology of the area are influenced by the volcanic nature of the Rift valley. The River Njoro watershed is covered by loamy soil in the upper forested parts having developed from ashes and other pyroclastic rocks of recent volcanoes (Ralph and Helmidt, 1984) and deep to deep well drained to moderately deep loamy sandy clays (vitric andosols). The lower reaches are covered by erosive lacustrine soils (Chemilil, 1995). Average annual rainfall is 1200 mm, distributed trimodally with peaks in April, August and November.

Land cover detection using Landsat Satellite Imagery by Baldyga (2004) shows a rapid loss of forest cover in the upper reaches of river Njoro watershed. Plantation and Indigenous forest cover have been lost to other land uses by 9% and 10% respectively in the period between 1986 and 2003. A 6 % reduction in the large scale farms in the area has been noted. In the same period, there has been an increase of 26% in land area used for small scale agriculture and pasture.

Rainfall Simulator

The potable field rainfall simulator used in the study had the dimensions of 0.4m x 0.25m to give a plot size of $0.1m^2$ (Kamphorst, 1987).

Procedure

The study was carried out on runoff plots on 20% slopes, which were used to assess soil erosion and surface runoff in river Njoro watershed in Kenya. The study was done on a randomized block design with five land use treatments and three replicates (sites) per treatment. The sites were mapped using GPS and plotted in a GIS environment. On every study site, rainfall was applied at an average rate of 10 mm/h on the three plots using the rainfall simulator. Soil loss and surface runoff generated from the three plots of every site of different land uses were measured. The soil characteristics including, bulk density, organic matter content, texture and pH were measured at each experimental site.

Statististical methods

Data on soil loss, runoff, bulk density, organic matter content and pH were subjected to statistical analysis in the computer using STATISTICA for windows (Statsoft, 2001). The source of variation for each component was estimated using a Randomised Complete Block Design (RCBD) (Steel and Torrie, 1980). The value for an individual variable (Y_{ij}) can be explained by the ideal statistical model,

$$Y_{ij} = \mu + \tau_i + \beta_j + \varepsilon_{ij}$$

Where

μ = represents the overall mean

τ_i = effect of land use

βj = effects of sites

ε_{ij} = random error component

Analysis of Variance (F statistics) was used to test whether component means were significantly different at the 0.05 level within the land uses and within the sites. The significance of the difference among means was evaluated using Duncan Multiple Range Test (DMRT) after the analysis of variance testing.

Results

Table 1 presents the results obtained from following the procedure described above. The data on soil loss, runoff and other soil properties were recorded. The analysis of variance tests showed no significant differences ($p>0.05$) between the three sites in every land use. The data on soil loss and runoff which are the averages of 18 replications per land use show that soil loss ranged between 86 g/ 0.1 m^2 to zero g/ 0.1 m^2. Soil loss decreased in the order agricultural land, deforested land, grazing land, plantation forest and nil in Indigenous forest. There were significant differences ($p< 0.05$) between agriculture land use and all other land uses (Table 2). However, there were no significant differences between the following land uses: deforested, grazing, plantation and indigenous forest. These registered mean soil loss values (in g/ $0.1m^2$) in the order of 32, 18, 2 and Zero respectively. The mean values for surface runoff in ml. for the grazing, agriculture, deforested land, plantation forest and indigenous forest were, 1200, 920, 860, 380 and 20 respectively. Grazing land recorded the highest surface runoff while indigenous forest recorded the least. There were statistically significant differences ($p<0.05$) between all the land uses except between agriculture and deforested land uses and agriculture and grazing land.

Table 1. Mean soil loss and soil properties for the five land use types in Upper River Njoro watershed.

Land use	Bulk density (g/cm^3)	Organic matter (%)	Soil pH	Soil Texture	Mean surface runoff (ml)	Mean soil loss ($g/ 0.1m^2$)
Agriculture	0.85	5.6	6.2	Clay loam	920	86
Grazing	1.05	5.0	5.9	Clay loam	1200	18
Plantation	0.95	6.2	6.4	Clay loam	380	2
Deforested	0.78	10.1	5.8	Sandy clay loam	860	32
Indigenous forest	0.74	9.5	6.2	Sandy clay loam	20	0

Table 2. Results of Duncan's Multiple Range Test* ($p < 0.05$).

Land use	Runoff
Grazing	a
Agriculture	a
Deforested	a b
Plantation	c
Indigenous	d
Land use	**Soil loss**
Agriculture	a
Deforested	b
Grazing	b
Plantation	b
Indigenous	b
Land use	**Organic matter**
Deforested	a
Indigenous	a
Plantation	b
Agriculture	b
Grazing	b
Land use	**Bulk density**
Grazing	a
Plantation	b
Agriculture	b
Deforested	b c
Indigenous	b

* Land use types that have a letter in common in a particular column do not differ significantly at the probability level shown above for each soil characteristic analyzed.

Discussion

It is evident from this study that surface runoff and soil erosion were largely influenced by the watershed management conditions. The forested land use types, namely, the indigenous forest and plantation forest, recorded the lowest surface runoff and soil loss. The land use types characterized by intensive interference of land cover and soil surface conditions like agriculture, grazing and deforested areas recorded high runoff and soil loss values.

The analysis of bulk densities showed grazing land use as having the highest compaction values among the land uses studied. This can be attributed to compaction of the grazing area by livestock (Navar and Synnott, 2000: Castillo *et al.,* 1997). Compaction often leads to the total sealing of soil pores, leading to reduced infiltration and increased runoff as was the case in this land use. Overgrazing by livestock in the grazing land also led to high runoff values due to removal of vegetation cover (Castillo, 1997: Sutherland, 1990). Lack of vegetation reduces interception, evapotranspiration, resulting in high runoff conditions on soils without a protective covering (Michel, 1990). The grazing land use was exposed to surface runoff due to lack of interception as a result of less vegetation occasioned by overgrazing (Sutherland, 1990: Ziegler et al., 1997). Removal of vegetation cover also led to low organic matter in the grazing land which led to low infiltration and high surface runoff (Hai and Ong, 2000: Kironchi *et al.,* 1993).

Runoff in agriculture and deforested areas was statistically indistinguishable in River Njoro watershed. However, there were significant differences between the two land use types and the forested areas in the watershed. The differences in runoff between agricultural and the forested areas may be due to low infiltration in the agricultural land. Stycen and Morgan (1992) reported that through an increase in infiltration and soil moisture capacity, vegetation may decrease the amount of runoff generated during a storm. It is evident that the high runoff in agricultural areas relative to forested areas in the River Njoro watershed may be due to high infiltration in the latter occasioned by good vegetation cover.

According to the results of this study, soil loss was highest in agricultural land use type in River Njoro watershed. Many studies (Ziegler et al, 2000; Castillo, 1997) have shown that high run off is consistent with high soil losses. In this study, agricultural land use had notably high mean runoff second only to grazing land. The mean runoff difference between the two land use types was, however, not statistically significant ($p>0.05$). The results for agricultural land use area confirm the results of other studies that high runoff rates are consistent with high soil losses. High runoff in agricultural land use led to high soil losses. The major contributor of high soil loss in agricultural land was tillage, which detached a generous supply of loose aggregates and soil particles. The relatively low organic matter content of agricultural land use led to the detachability of soil particles and consequently high soil losses.

Contrary to expectation, grazing area lost less soil inspite of having highest mean runoff. This was probably because of limited loose aggregates and soil particles occasioned by compaction and subsequent crusting, which increased the shear strength of the soils. The relatively high organic matter and litter cover reduced soil loss in the deforested sites. Forested land use types (indigenous and plantation) are undoubtedly the best in terms of soil erosion control. These areas recorded the lowest soil losses. This was due to surface cover conditions, which led to high infiltration, less runoff, relatively high organic matter and low bulk density especially for the indigenous forest land use type.

Previous studies (Aboud, 1992; Anyango 2000; Karanja *et al.,* 1985) have shown soil erosion as a major problem in the watershed. This study has identified agriculture, grazing and deforested land use types as the main cause of the problem. From the results of this study, it can be argued that by maintaining and/or increasing the forest cover, surface runoff and soil erosion will reduce. Increasing the forest cover through afforestation and reforestation programmes should be a priority for the stakeholders in the watershed. Research has shown that tree cover does not only improve the hydrological conditions of the land, but also improves soil fertility. (Verinumbe, 1987; Belsky, 1994).

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References

Aboud A. A., 1992. *Population Pressures, Environmental Degradation and Farmers Adaptive Strategies in Nakuru District of Kenya.* Urbana, Illinois, U.S.A.

Anyango C., 2000. *Awareness and Response by Smallholder Women Farmers to Soil Erosion. A Case Study of Njoro Kenya*. Egerton, Njoro.

Baldyga, T.J, Miller S.N, Shivoga W. and Gichaba M., 2004. *Assessing the Impact of Land Cover Change in Kenya using Remote Sensing and Hydrologic Modelling.* Colorado, USA.

Belsky, A. J., 1994. Influences of trees on savanna productivity: Tests of shade, nutrients, and tree-grass competition. *Ecology,* 75 (4): 922-932.

Castillo V M, Martinez-Mena M, and Albaladejo J., 1997. *Runoff and soil loss response to vegetation removal in semi arid environment.* Department of soil and water conservation, Murcia, Spain.

Cheruiyot S.K., 1984. *Infiltration rates and sediment production of a Kenya bushed grassland as influenced by vegetation and prescribed burning.* MSc thesis, Range Science Department, Texas A & M university College, Station Texas.

Hai M.T., Ong C.K and Mungai D.N. 2000. *The impact of land use and rainfall on runoff and sedimentation in the lake Victoria basin*. ICRAF. & U.O.N, Nairobi, Kenya.

Kamphorst A, 1987. A Small Rainfall Simulator for the Determination of soil Erodibility.*Netherlands Journal of Agricultural Science*, Netherlands.

Karanja *et al.,* 1985. *The influence of land use on Njoro river catchment between 1975 and 1985*. Egerton University, Njoro.

Kironchi G, H.P. Liniger, Mbuvi J.P. (1993) Degradation of soil physical properties in overgrazed rangelands in Laikipia district. *In.* Gichuki, F.N., Mungai D.N Gachene C.K.K., Thomas, D. B. eds. *Land and water management in Kenya; Towards a sustainable land use.* Proceedings of the fourth national workshop. Department of Agricultural Engineering /Soil of Water Conservation Branch, U.O.N., Nairobi.

Navar J and Synnott T. J., 2000. *Surface runoff, soil erosion and land use in North Eastern Mexico*. University of Nuevo Leon, Mexico.

Ralph and Helmdt, 1984. A*gricultural Management Handbook*. Ministry of Agriculture and Livestock Development, Nairobi, Kenya.

Steel R.G and Torrie J.H. (1980). *Principles & Procedures of Statistics. A Biometrical Approach2nd edn*. Mc.Graw Hill International Editions, Singapore.

Styczen M.E., Morgan R.P.C., 1995. Engineering properties of vegetation. In R.P.C Morgan, R.J. Rickson eds. *Slope stabilization and erosion control; A bioengineery approach*. Chapman & Hall.

Sutherland R.A. & Bryan R.B., 1990. *Runoff and erosion from semi-arid catchment. Baringo district, Kenya*. Butterworth & Co, Canada.

Verinumbe, I (1987) Crop Production on soil under some plantations in the Sahel. Agroforestry Systems, 5; 185-188.

Vliet L.J.P. *et al.,* 1986. *Soil Erosion Loss Monitoring And Prediction Under Semi-Arid Agriculture in the Peace River Prairie Region Of NW.* Canada.Land Resource Research Institute, Vancouver.

Ziegler A.D and Giambelluca T.W., 1997. Importance of rural roads as source areas for runoff in mountainous areas of Northern Thailand. *Elsevier Science*, USA.

Watershed degradation analysis of Nzoia River drainage basin from a policy change perspective and using remote sensing data

Onywere Simon[1], Zachary Getenga[2], Shadrack Mwakalila[3], Charles Twesigye[4], Josephine Nakiranda[5]

[1]Kenyatta University, P.O. Box 43844 00100 Nairobi, Kenya
[2]Western University Collage of Science and Technology, Kakamega, Kenya
[3]University of Dar Es Salaam, Dar es Salaam, Tanzania
[4]Kyambogo University, Uganda
[5]Makerere University, Kampala, Uganda
Email: Email:onyweres@yahoo.com

Abstract

Lake Victoria is a vital resource for Kenya, Uganda and Tanzania. Growing populations in the watershed of the lake and multiple activities have increasingly come into conflict with recent strategies on socio-economic development in the basin reflecting a shift towards a focus on poverty eradication. This is reflected in policy documents such as National Poverty Eradication Plans; Poverty Reduction Strategy Papers; Economic Recovery Strategy for Wealth and Employment Creation (RoK, 2002d) and State of Environment Reports (NEMA, 2004). Formation of institutions and research programmes such as Lake Victoria Environmental Management Programme (LVEMP), Lake Victoria Research Initiatives (VICRES) and Nile Basin Initiative (NBI) have lead to development-oriented research. This study was carried out in two selected sites of Nzoia River drainage basin at Nzoia sugarcane growing area and Budalangi floodplain in the Western region of Kenya under VICRES. It examines land cover changes based on analysis of time series Landsat satellite imagery of 1973, 1985, 1995, 2001, 2003 and 2005. The interpreted land cover changes and impacts on watershed resources show a strong relation and influence of policy on socio-economic development trends. The characteristics of poverty and other inequalities in River Nzoia watershed indicate that the poor are identified by their categories (culture) and habitat (land) despite natural resources potential; broad based rural development; and expansive sugarcane growing. This is mainly as a result of mono-cropping patterns of sugarcane and land ownership policy. Attempts at using Budalangi floodplain for rice growing and other food-crop production strategies face biophysical and socio-economic challenges and have failed to address the people's vulnerability to famine, water-stress and land degradation. The challenges presented are those considered to impede access to resources, employment opportunities, improved agricultural productivity and that lead to environmental degradation from the use of the rivers as sinks for agricultural, domestic and industrial wastes and from persistent siltation. The watershed degradation is negatively impacting on Lake Victoria.

Introduction

The watersheds of Lake Victoria drainage basin suffer functional degradation and are threatened by land degradation, hydrologic alteration, and water pollution. The exponential growth in human population coupled with pressure on land and increasing poverty levels have pushed riparian communities to ecologically marginal and fragile wetland ecosystems in the basin for their fundamental socio-economic survival. This prevailing situation threatens the local physical environment and the sustainability of its biodiversity.

Recognising the challenges in the basin, the Government of Kenya has put in place a number of policies and programmes such as development plans, sessional papers and strategy papers that emphasize rapid economic growth. After four decades of trying to implement the development plans, the socio-economic inequalities still persist. Poverty in the country has increased from 3.7 million people affected in 1973 to 15 million people at the turn of this century. The elaborate 1997 welfare and monitoring survey in Kenya indicated that 53% of people in rural areas were categorized as overall poor and 51% as food poor (RoK, 2001a). The poor were clustered into certain socio-economic categories that included small-scale farmers, pastoralists, agricultural and casual labourers, unskilled and semiskilled workers, female-headed rural households, the physically handicapped, HIV/AIDS orphans and street children (RoK, 2001a). Most people in these clusters are landless and lack formal education. The poverty situation has spawned a wide gap between the rich and the poor. Statistics from Central Bureau of Statistics (CBS) show that income and social services are skewed in favour of the rich (CBS, 2000). Inequalities are manifested in many forms with the country's 10% of the population controlling 42% of the total wealth while the bottom 10% control less than 1% (CBS 2000). Further, regional inequalities within the country are extreme with for example the difference in life expectancy between Central and Nyanza Provinces being a staggering 19 years (SID, 2004). The social and economic disparities area great UNDP, (2002).

Nzoia River drainage basin the focus of this study traverses a vast area from its upper catchment in Cherangani Hills; Mt Elgon; and Nandi Escarpment with its adjoining Trans Nzoia Plateau through its mid-watershed at Nzoia and Mumias sugarcane growing areas of Bungoma and Butere-Mumias Districts to the Budalangi floodplain within Busia and

Siaya Districts. It is the largest drainage basin in western Kenya. At port Victoria where the river discharges into Lake Victoria, the river has built a sediment fan into the lake. Two test sites at Nzoia Sugarcane growing area (Bungoma District) and at the Budalangi floodplain area (Busia District) are considered. Busia District extends from latitude 0º to 0º 45" North and longitude 33º55' to 34º25' East ($1262km^2$) with $137km^2$ of its land under permanent wetland conditions. The census report of 1999 (CBS, 2000) puts its population at 370,608 with a growth rate of 2.95% per annum. Bungoma District on the other hand lies between latitude 0^0 25.3' and 0º 53.2' North and longitude 34º 21.4' and 35º 04' East (2068.5 km^2), which is about 25% of the total area of Western Province. Despite the two districts being placed in agro-climatic Zone II and III and should be productive from rain-fed agriculture, they face the challenge of food scarcity and environmental impacts associated with land degradation due to poor land use practices and impacts from floods due to unplanned settlement in the floodplain.

Research methodology

A survey design to collect primary data for the study was motivated by the land cover variations and change interpreted from Landsat satellite imagery composite for six dates (the image reference is path/row_Landsat system_date of acquisition year-month-day):

- p183r060_1mss_19730202 (Budalangi)
- p170r060_5tm_19860308
- p170r060_5tm_19950402
- 182r060_1mss_19730201 (Nzoia)
- p170r060_7emt_20010205
- p170r060_7etm_20030518
- p170r060_7etm_20050421

The images were used to assess the land cover changes and land use patterns in the area at approximately 10-year interval since 1973. Image interpretation was made based on land cover types, supervised land cover classification and the Normalised Difference Vegetation Index (NDVI) computed from the various image dates. Tonal, hue values and image texture and structure were considered in making decisions on the land cover changes.

A review of the area's Agricultural Reports, District Poverty Strategy Papers, District Development Plans as well as institutional reports was made before conducting a field survey in April 2005 to 55 households in the Budalangi floodplain area and 43 households in the Nzoia Sugarcane growing area. Interview schedules were also administered to institutions and the civil society such as NGOs and CBOs operating in the area and involved in community development projects.

To gauge the level of water degradation, physico-chemical parameters of water samples were determined using field equipment (water sampler). Geographical positions and altitudes of the sampling sites were measured using a GPS instrument and the following sites were considered: at River Kuywa (at the intake for the Bungoma Town water supply); at Webuye (Pan Paper factory water intake); at Webuye (10km down stream of discharge point from Pan Paper factory; at River Kuywa (3km down stream of Nzoia Sugar Factory) discharge point; Nzoia River downstream of the discharge from Mumias Sugar Factory; Nzoia River at port Victoria 2km from the Lake; Nzoia River at Rwambwa Bridge within the Budalangi floodplain; and Nzoia River at Rwambwa market.

Kenya Government policy on poverty and inequality reduction

Kenya has pursued the Service Center Strategy Plan and the Rural Trade and Production Centers to facilitate rural centers get services and to stimulate rural industrialization by promoting cottage industries that utilize local raw materials. To enhance on-farm income, the government established agricultural based institutions such as the Agricultural Development Corporation (ADC), National Irrigation Board (NIB), Kenya Agricultural Research Institute (KARI), National Cereals and Produce Board (NCBB), Lake Basin Development Authority (LBDA) and sugarcane factories. Within the framework of these institutions, agricultural programmes were initiated by the 1970s by establishing agricultural schemes in the River Nzoia basin (Bunyala Rice Irrigation Scheme, Nzoia and Mumias Sugarcane growing Schemes, Busia Cotton Ginnery, and Yala Swamp Rice Project). From image interpretation of the time series Landsat satellite imagery of the area, the Bunyala Rice project was for example vibrant until 1986. It thereafter collapsed though dismal revival is seen in the 2005 satellite imagery (see figure 2). Sugarcane cultivation on the other had has been expanding taking over all wetlands and the land meant for subsistence agriculture. This mono-crop cultivation and reliance on the sugarcane industry has brought in a dilemma due to inadequate food-crop cultivation by the communities. The sugarcane factories have been under receivership at various times due to poor management. In addition, the cotton ginnery has not been in operation since mid-1980s. The government policies and strategies have therefore failed to ensure that the agricultural schemes meet their mandate of rural development. These failures are attributed to bad governance and lack of accountability.

In 1999, the National Poverty Eradication Plan (NPEP) was framed (RoK, 1999a). The Plan outlines the scope of poverty and sets targets for poverty reduction by increasing productivity, reducing unemployment, improving infrastructure and creating enabling environment for increased investments by 2015. It also aims to strengthen the capabilities of the poor and vulnerable groups to earn income, reduce gender and geographical disparities, and have a healthier, better-educated and more productive population, challenges that the Millennium Development Goals (MDGs) address (RoK, 2005). Kenya Participatory Impact Monitoring (KePIM) RoK (2002e) and the Interim Poverty Reduction Strategy Paper (I-PRSP) incorporated the adoption of the Medium-Term Expenditure Framework (MTEF) as the organizing budgetary framework for the use of resources and formulation of Poverty Reduction Strategy Paper (PRSP) was an important initiative to tackle the problem of increasing poverty levels in the country (RoK, 2001a). The planning approach adopted by the PRSP is that of bottom up with involvement and participation of the stakeholders at the grassroots level. The PRSP, however, faces obstacles such as lack of adequate funding, lack of political will and inadequacies in linking environmental opportunities and risks to the development agenda of the country (see Figure 1). Most of the interventions measures put in place by the Government are however donor driven and are therefore not integrated into the country's long-term development goals, and consequently they suffer from discontinuity and inconsistency.

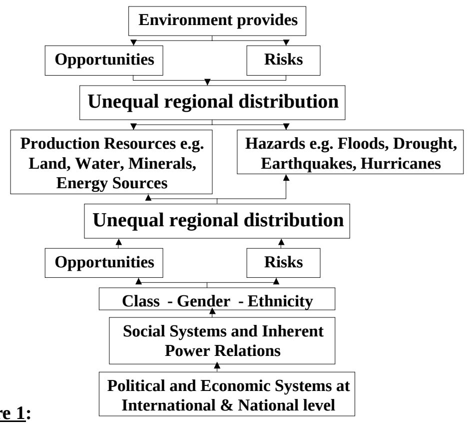

Figure 1:
Relationship between the environment and socioeconomic and political systems.

The National Food Policy (1994) promoted enhanced food production and targeted relief through food for work programmes designed to reduce the poor households' vulnerability to food shortages. The HIV/AIDS Policy (RoK, 1997) provides the framework for strategies aimed at preventing and controlling HIV/AIDS, which is a threat to the country's socio-economic development. This was done through the National Aids Control Council (NACC) with which the National Disaster Management Authority (NDMA) works closely to deal with the pandemic. Other linkages to existing National Policies and Legislations in addressing the issues of poverty are the National Development Plan (NDP) 2002-2008 (RoK, 2002a) which recognizes that 56% of the Kenyan population is afflicted by poverty and that environmental disasters such as draught and floods can push more people below the poverty line. The draft on Disaster Management Policy (RoK, 2002b) and the Environmental Management and Coordination Act (EMCA) (RoK, 1999b), have provisions for hazards prevention. The Water Act, 2002 (Cap 372) is biased to clean water provision without consideration to disasters arising from excess water in inhabited areas including floodplains (RoK, 2002c) although the Kenya Meteorological Department's Information Policy has a provision for availing forecasts in general terms.

These national policies, some of which are driven by the international discourse have therefore not had a profound effect in tackling rural poverty. This is mainly because of the government's failure to translate them into action. To reduce the vulnerability of the people to disaster and other challenges, social and technology systems must adapt to changing physical and social environment. The formulation of institutions and research programmes such as Lake Victoria Environmental Management Programme (LVEMP), Lake Victoria Research Initiative (VICRES) and the Nile Basin Initiative (NBI) provides a forum for addressing challenges facing communities such as those living in Nzoia River Drainage Basin.

Understanding flood impacts in Budalangi floodplain

The failure of the Busia District Development Plan (RoK, 2001b; RoK, 1996) to propose any programmes for flood management signifies major weak linkages between National Planning and District Planning that is reflected in the Budalangi flooding problem. Despite the government's extensive grassroots based development and administrative network, which would effectively manage emergencies, and disasters such as famine and floods in the area, there is no attempt to mainstream disaster management in the National Development Plans or the District Development Plans. Consequently, sectoral budgets on disaster management are lacking. This also reflects weaknesses in the socio-economic system and class structure that allocates income and access to resources leading to impacts in terms of the people's ability to cope with flood hazards.

Weather related vagaries (alternating draught and flood periods) and other environmental challenges: plague/locusts of 1931 and 1933; army worms of 1931 and 1977; famine of 1942, 1968, 1979, 1984 and 1995; floods of 1975, 1976, 1985, and 1998 have impacted the area for a long time (RoK, 2002b). The prevalence of floods in Budalangi has been unique in frequency and character. The Southern half of Budalangi Division is flood-prone lowland (see Figure 2). Flooding in Budalangi has been noted even when there are no rains in the region raising concerns about the need to manage the entire Nzoia River drainage basin. Although in Budalangi, floods are a perennial problem, the present arrangements for disseminating information are inadequate. Flood forecasts are not disseminated immediately they are received from the forecast formulation team at the Meteorological Department in Nairobi. For this to be effective, response teams need to be set up on the ground. The local communities have therefore no means of getting the forecast information and if any come it is often too late and it does not reach a wide audience. Sometimes unwarranted flood warnings are issued based on inadequate data leading to tension among the affected communities. Further, there is no proper co-ordination and control of activities based on information from hydrological monitoring stations along the river.

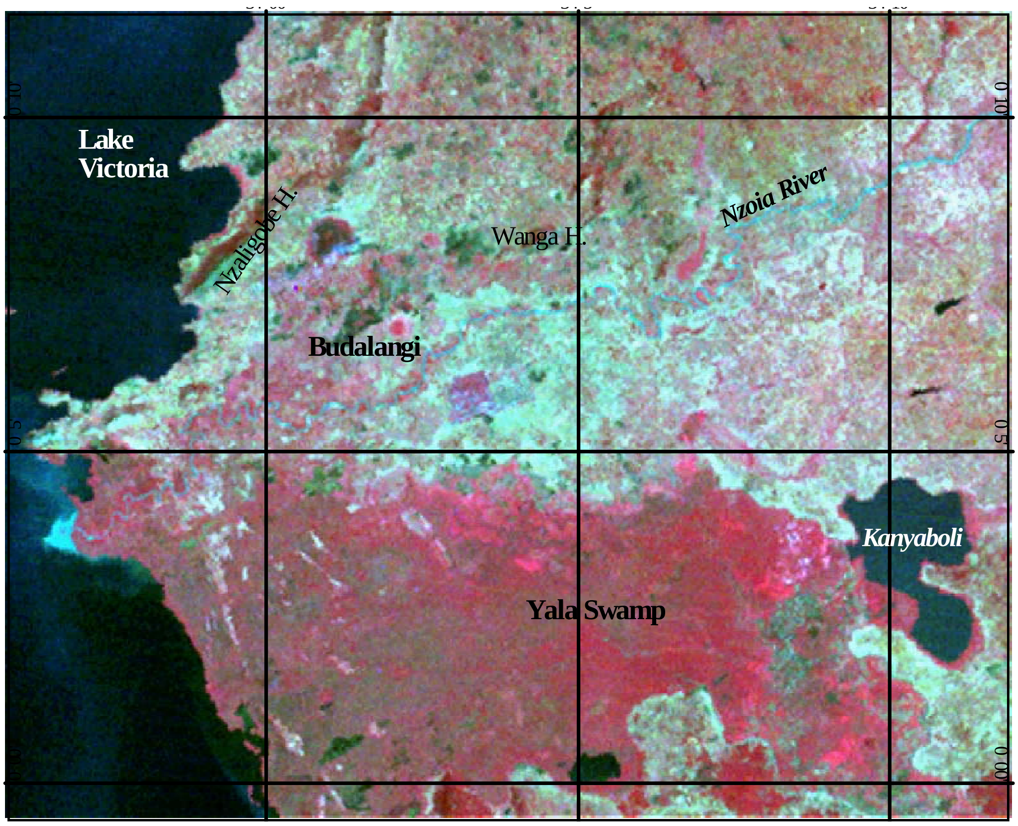

Figure 2. A setelite photo of the Southern half of Budalangi Division is flood-prone lowland.

Nzoia River seen from the false colour composites (FCC) images of the 6 years considered exhibits general blue waters that are an indication of high levels of turbidity and siltation (Figure 2 and Figure 3a and c), a reflection of the poor land use practices at the watershed area. There has been a progressive increase in the sediment load as reflected in the size of the sediment fan at the mouth of Nzoia River. Progressive depletion of riparian vegetation has also increasingly impacted the riverine and reduced its biodiversity.

Figure 3: Challenges faced by the communities settled in the Budalangi flood plain. Photo 1 is courtesy of UNEP/RoK survey July 2003; other photos are from field.

Although scientific and engineering techniques have developed remarkably over the decades, success has not been achieved in insulating humans from the effects of floods. Households in Budalangi Division can avoid residing in areas that experience periodical flooding. However, population growth and pressure for land continue to push them to flood prone areas thus putting themselves at risk. Their plight is clearly evident from exacerbated poverty level in the area. Figure 3b shows the status of the settlement with some donated tent structures to those whose houses were washed away. Long-term solution to the flood problem has been the Government's effort to construct dykes, although this has failed to contain the problem. Despite the presence of the dykes, and according to to emergency evaluation report (Macharia, 2003), floods displaced nearly 25,000 people out of Budalangi's population of 53,000 in 2003. Some 10,000 people were accommodated in the District Officer's camp, necessitating health emergency measures to control possible outbreaks of malaria, bilharzias, cholera and other water borne diseases. When the floods come, roads are made impassable, children are cut off from schools, food-crops in farms are swept away and latrines and buildings collapse, destroying the very livelihoods of the people. The extent of the area impacted and that under threat from future occurrence is seen from interpretation made on Landsat satellite imagery, bands 453 RGB captured on 18th May 2003. A comparison is made with the status of the floodplain in 1973, 1986, 1995, 2001 and 2005 (see Figure 2 for the dates of reference).

Analysis of the 1997-2001 and 2002-2008, Busia Development Plans (RoK, 1996 and RoK, 2001b) and Poverty Reduction Strategy Papers (RoK, 2001a) indicate that there was dismal implementation of the plans with high rate of failure of poverty reduction programmes. The plans showed that financial constraints led to only 30% of the plan being implemented. Other plans have failed before. For example, the revitalisation of the agricultural sector through the Ministry of Agriculture's National Irrigation Board established a rice irrigation scheme in the Budalangi area in the late 1970s and early 1980s. By 1986 the scheme showed a vibrant economy and a record of the status of the scheme is reflected in the Landsat image interpretation of 1986 image (figure 2b). Subsequent Landsat satellite image records show a depressed agricultural activity with complete abandonment of the scheme by 2001. The scheme has only been revived in the second year of the current government (Figure 2f). Other challenges due to the failure of the government development plans are for example lack of implementation of the Sirisia and Mufula Youth Polytechnics project that was meant to utilise local raw materials to bring income. The project failed due to the collapse of the dairy, cotton and sunflower schemes and lack of financial support from institutions offering credit facilities. An integrated sustainable flood management action plan for Budalangi Division that takes into account land reform, policy review, formulation and implementation if put in place would enhance the socio-economic activities in the area. This will contribute significantly towards disaster reduction and poverty alleviation.

Challenge of sugarcane growing and other impacts in Webuye-Nzoia area

The main economic activity in Webuye-Nzoia area is sugarcane growing that is processed at Nzoia Sugar factory. The sugar factory and its nucleus plantation occupy land taken away from local communities and a wetland at River Kuywa. The promises by the government of industrialization of the area and income from out-growing sugarcane to service the sugar factory with raw materials lead the local communities to abandon subsistence farming in favour of sugarcane cultivation. Initially sugarcane cultivation provided a good income. Over the years more land has been dedicated to sugarcane growing leading to over dependence on the crop and its oversupply and therefore low prices. There is also interference from middlemen who act as brokers between the sugar factory and tractor owners on the one hand and the out-growers on the other, leading to loss of income as most of the returns purportedly go to servicing the facilities (land preparation, seedlings, fertilizer, harvesting and transport) provided by the factory. In addition the sugarcane bushes do not adequately cover the ground. They also impact on the soils by compacting it. This has lead to increased runoff with little if any percolation. This is reflected in poor discharge from springs, increased turbidity and reduced discharge in the major tributaries of Nzoia (Kuywa, Chwele, Nambirima, and Chebaiywa). The total suspended solids (TSS) range from 21.4mg/l at Rwambwa market in Budalangi area to 128.3 mg/l at Bungoma Water Works on River Kuywa. Some 3Km down stream of Kuywa river from Nzoia Sugar Factory the TSS is 64mg/l and indicate clearer water downstream of Nzoia River.

Effluent from Pan Paper factory has overwhelmed Nzoia River itself and the surrounding area (Figure 4). There is impact from air pollution for several kilometres around the factory from fumes, dust, smoke and dry foam blown by the wind from the treatment ponds (Figure 4c). The effects from the air pollutants are seen in the corroded iron roofing tile on building, poor health of the people, poor quality of soils hence poor crop yield and poor quality of water in the rivers. The discharge from the factory also contains humus materials and sludge, which give the water a dark colouration and also renders it useless. The river water is not used for any form of activity (domestic use, washing, livestock, irrigation) for at least 20km downstream during the dry months of September to March of each year when the water level in the river is low and with a foul smell emanating from the water. People resort to the use of shallow wells. There is also no fish in the river at this time of the year. There have been records of dead fish at the mouth of the river. The biodiversity associated with a normal riverine area is absent. The Pan Paper effluent thus negatively impacting on the waters. During this period the water shows the following physico-chemical charactersistics measured from the field in February 2005: Dissolved Oxygen (7-17mg/l); Total Suspended Solids (21.4 – 128.3 mg/l); Chlorides (9-22 mg/l); Conductivity (123-198 μS/cm), NO_3 (0.034 – 0.528mg/l).

Figure 4. Biophysical challenges to poverty reduction in Nzoia River Basin occasioned by pollution from the Pan Paper Milling Factory.

Along river Kuywa 3km downstream of Nzoia Sugar Factory and within the out-grower zone cultivation practices allow for a buffer zone along the river, which protects against erosion. This is seen in the lower levels of total suspended solids in the water; 64 mg/l compared to 128.3 mg/l at the Bungoma Water Works.

Socio-economic factors impeding poverty reduction in Nzoia river basin

The majority of the households in the Busia areas traversed by Nzoia River have between 5 to 10 members, most of whom (72.65%) live on less that 4 acres of land. 80.9% of the families are male headed households which reflect on and imply a male dominated land ownership. The basin has a rich paternalistic cultural tradition leading to cultural-based land inheritance by the male children. The women, apart from those windowed, have no right to any land and although very enterprising, cannot get access to credit where land can be used as a collateral.

The first generation land title deed issued shortly after independence is the only legal evidence of land ownership. The title deed holders have been reluctant to subdivide their land among the older sons most of whom are married with their own families. This has slowed down the agrarian efforts of poverty reduction that could be spearheaded by the more educated and informed sons. In general 20.7% of the population has not received any formal education. Those with formal tertiary education, which is responsible for imparting life and economic transformation skills, accounted for only 7.5 % of the population. This has serious implications on poverty reduction campaigns, especially absorption of new innovations. It limits socio-economic ideas that are necessary for change. Management skills of any enterprise ventures aimed at economic utilization of the wetland resources are grossly lacking in the basin. Numerous interventionist efforts by a myriad of governmental and non-governmental organizations over the years have also failed to turn around the development fortunes of the area.

Within the Budalangi area 46.9 % of the population are engaged in farming of crops such as maize, sugarcane, cassava and beans and keeping of livestock. In the Nzoia area on the other hand, 78.7 % of the people are engaged in sugarcane growing. Population growth in the two areas far outstrips economic growth leading to deprivation of many households especially during the time of distress such as during flooding and draught. Problems bedevilling the agricultural sector in the area include farm inputs credit, donor dependence, high cost of farm inputs, lack of access to production assets and inhibitive cultural practices. The commercialisation of farming activities in the basin has also hampered local development especially where land has been leased to farmers residing outside the basin such as in Nairobi, Kakamega, and Bungoma among other towns.

Lake Victoria has had a big influence on the population in Budalangi area with men engaging in fishing activities. Fishing account for 34.4% of the livelihoods of the households. Most young women in the area engage in vending the fish leaving the aging population and the children to take care of the farm. Middlemen from outside the district exploit the fishermen by dictating the prices of the highly perishable fish. Fishing is therefore a high-risk business and is done as cultural activity rather than a serous economic activity. There are also constant incidences of conflicts in the basin caused by boundary disputes with neighbouring Uganda.

Extension services are inadequately dispensed across the basin. 62.5% of the respondents indicated that extension service, which is essential channel of propagating new technologies, does not serve them. This is attributed to change of government policy from supply driven to demand driven extension initiated in 1988. Agricultural extension officers on the ground are thinly spread due to employment freeze by the government since 1988. Common avenues of disseminating the scanty extension services information included seminars, workshops and public gatherings (*barazas*). Veterinary services that had largely been offered by the Government were privatised in 1998 therefore throwing into jeopardy livestock improvements in the basin. There is thus lack of emphasis on extension policy by the government making its role in the basin inconsequential to farmers. In addition, the readily disposable income from fishing is fuelling HIV/AIDS pandemic and heightening school dropout rate as children are pulled out of school to engage in economic production. This implies that child labour is rampant in the basin making the eradication of poverty through education and awareness creation in the long term, a mirage.

Biophysical challenges to poverty reduction in Nzoia basin

In Busia District, about 168,000 ha (90%) of the total land area is suitable for agricultural development. However, large tracts of the land about 83,000 ha (51%) is composed of swamp, bushes, fallow or grazing land out of which some 13,000 ha (8%) can be made agriculturally productive through irrigation and drainage. Most of the land areas in the lower part of the basin are either under extensive traditional grazing, bush or marsh. Crop production is therefore mainly subsistence in nature. The basin has a potential to grow a wide variety of crops and can increase livestock production but has been hampered by a number of biophysical factors including flooding, tsetse fly infestation and lack of market and veterinary services. This is well elaborated in the flood management strategy for Lake Victoria basin, (2004).

The good soils coupled with gently sloping terrain and high water table in the floodplain has attracted high population density with ownership of small land units. Although the floodplain is most suitable for arable farming, it is more vulnerable to flooding annually with intensive flooding occurring in a 5-year and a 10-year cycle. The floods destroy the accumulated assets. The government efforts to built dykes has not been effective due to high sediment load which often blocks the meandering river consequently leading to change of the river course that break the dykes. Poor farming methods in the upper catchment and throughout the basin cause serious soil erosion leading to soil fertility loss and declining productivity per unit area. In some cases, direct farming on the dyke weaken the structure. The high water table often lead to water-log rendering some part of the floodplain unsuitable for agriculture and settlement.

Conclusion

Despite a shift in the government's policies, strategies and priorities on socio-economic development towards a focus on poverty eradication, it has had dismal impact in the Nzoia River Basin. Rural development is essential for the reduction of poverty and inequalities especially vulnerability to famine, water-stress (floods and draughts) and land degradation. In the Lake Victoria drainage basin, the various challenges have led to the formation of institutions and research programmes such as the Lake Victoria Environmental Management Programme, the Lake Victoria Research Initiatives and the Nile Basin Initiative in order to come up with participatory, target specific, and home grown solutions to conservation and sustainable development for rural poverty reduction and inequalities in the basin.

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References

Central Bureau of Statistics – CBS, 2000. *Economic Survey of Kenya, 2000*.

Macharia, Robert, 2003. Floods, 2003 *Emergency Response Evaluation Report*. Unpublished.

National Environment Management Authority – NEMA, 2004. *State of Environment Report, 2004, Busia District*.

Republic of Kenya – RoK, 1994. *Food Security Policy Paper No.2 of 1994 on National Food Policy*. Government Printer, Nairobi, Kenya.

Republic of Kenya – RoK, 1996. *Busia District Development Plan 1997-2001.* Government Printer, Nairobi, Kenya.

Republic of Kenya – RoK, 1998. *Bungoma District Development Plan 1997-2001.* Government Printer, Nairobi, Kenya.

Republic of Kenya – RoK, 1999a. *National Poverty Eradication Plan 1999-2015.* Department of Development Co-ordination, Ministry of planning and national development, Nairobi, Kenya.

Republic of Kenya – RoK, 1999b. *Environmental Management and Co-ordination Act No.9 of 1999*. Government Printer, Nairobi, Kenya.

Republic of Kenya – RoK, 2001a. *Poverty Reduction Strategy Paper 2001-2004. Ministry of Planning and National Development.* Government Printer, Nairobi, Kenya.

Republic of Kenya – RoK, 2001b. *Busia District Development Plan 2002-2008.* Government Printer, Nairobi, Kenya.

Republic of Kenya – RoK, 2002a. *National Development Plan. Government, 2002-2008.* Printer, Nairobi, Kenya.

Republic of Kenya – RoK, 2002b. *National draft Policy on Disaster Management, June 2002*. Government Printer, Nairobi, Kenya.
Republic of Kenya – RoK, 2002c. *The Water Act 2002 (Cap 372)*. Government Printer, Nairobi, Kenya.
Republic of Kenya – RoK, 2002d. *Economic Recovery Strategy for Wealth and Employment Creation 2003-2007*. Government Printer, Nairobi, Kenya.
Republic of Kenya – RoK, 2002e. *Kenya Participatory Impact Monitoring (KePIM); Perspectives of the Poor on the Anti-poverty Policies in Selected Districts.* Ministry of Finance and Planning, Government Printer, Nairobi, Kenya.
Republic of Kenya - RoK, 2004. *Flood Management Strategy for Lake Victoria Basin, Kenya*. Draft Report.
Republic of Kenya - RoK, 2005. *Special Issue on the Millennium Development Goals; The Official Newsletter of the Ministry of Planning and National Development.* Ministry of Planning and National Development, Nairobi, Kenya,
Society for International Development – SID, 2004. *Pulling Apart, Facts and Figures on Inequality in Kenya*. SID, Nairobi.
UNDP, 2002. *Addressing Social and Economic Disparities. Human Development Report United Nations Development Programme*, Nairobi, Kenya.

Chronological change of nitrogen and phosphorus loading and concentrations in rivers and lakes in Japan

Sato Kazuaki [1], Toru Abe[1], Yoshifumi Mitani[1] and Makoto Watanabe[2]

[1]: Foundation of River & Watershed Environment Management, 11-9 Nihonbashi-kodenmacho, Chuo-ku, Tokyo 104-0042, Japan

[2]: Foundation of River & Watershed Environment Management, Hokkaido Branch Office, 16-2 Nishi 7chome, Minami 1jyo, Chuo-ku, Sapporo-shi 060-0061 Japan

Email: sato-k@kasen.or.jp

Abstract

Eutrophication of lakes has occurred through altered land use and waste disposal practices of mankind. In Japan, the eutrophication has been accelerated by imported N and P loads from foreign countries as foods and fertilizer. In this study statistical data on public sewerage-serviced population, fertilizer production, livestock population and detergent production throughout Japan were arranged to examine the N and P loads, and it was revealed that the amount of N and P loads from these sources has gradually decreased in recent years. We found the similar decrease of the estimated N and P loads in the watershed of Lake Inba-numa, where a case study was carried out. This study also presents histograms of T-N and T-P concentration data from recent years and approximately 1980 obtained at approximately 500 water quality-measuring points in Japanese rivers and lakes. A comparison of the data has confirmed that during the 20-year period, T-P concentrations tended to decrease. But, in some lakes in Japan the water quality has not been improved yet, though the nutrient loads to the lakes has decreased like in Lake Inba-numa. More study is needed to recover the sound water environment for the lakes that once experienced severe eutrophication.

Key words: Eutrophication; nutrients; rivers and lakes in Japan

Introduction

In Japanese rivers and lakes, new problems related to water quality are appearing to replace the former problem: organic pollution indicated by high BOD values. Although present water-quality problems include micro chemical pollutants and so on, the most important concern is now the eutrophication of water bodies by rising concentrations of nitrogen and phosphorus.

The Ministry of Land, Infrastructure and Transport (MLIT) has measured water quality and accumulated data since 1960 at class A rivers of 109 national water systems. This study presents trends in water quality on nitrogen and phosphorus in rivers and lakes throughout Japan arranged by statistical data. Statistics on public sewerage-serviced population, fertilizer production, livestock population and detergent production, which may affect water quality, were arranged to examine their relationship on nitrogen and phosphorus concentrations in rivers and lakes. The study also presents the results of examination on chronological nutrients loading data in the watershed of Lake Inba-numa, where a case study was carried out.

Present status of T-N and T-P concentrations and trends of the past 20 years - data arrangement

The data was arranged based on the water-quality data in the water bodies under the direct control of the MLIT, using the “Water Quality Annual” (MOCRB 1999) and the “Yearbook of River Water Quality in Japan” (JRA 2000). The “Water Quality Annual” compiles annual water-quality data that the MLIT measures at class A rivers. Water quality measurement is carried out monthly, in principle, at 1,094 points in 109 national river systems. The “Yearbook of River Water Quality in Japan” published by the Japan River Association includes data on water quality, flow rate and watershed characteristics of each river system.

Distribution profiles of T-N and T-P concentrations in rivers and lakes in recent years were drawn, and then compared to data of approximately 20 years ago. Collection of data on T-N and T-P of rivers in Japan began with the establishment of T-N, T-P Environment Standards in Lakes in 1982. Accordingly, the three years from 1995 to 1997 were selected as recent years for data that has already been arranged, and the three years from 1979 to 1981 were selected for comparison as past years when the data first began to be collected. It was confirmed that these six years were not special flooding or drought years.

Table 1. Numbers of data comparison points between recent years and 20 years ago Numbers in parentheses are data collecting points for T-N or T-P.

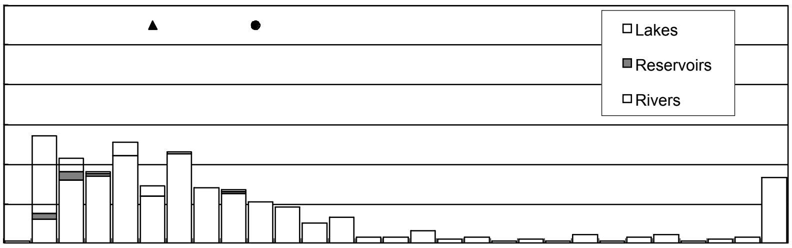
	T–N	T–P
Rivers	379 (591)	371 (585)
Reservoirs	9 (21)	8 (22)
Lakes	61 (82)	68 (82)
Total	449 (694)	447 (689)

As to the data arrangement, first the yearly average value was calculated using the data at each measuring point, and then the representative value at each point was calculated to obtain the average of three yearly average values. The total number of data arrangement points was about 700, and the number of points where the data from recent years as well as from 20 years ago were complete, was 449 for T-N and 447 for T-P, as shown in Table 1.

Results of the data arrangement

The histograms of T-N concentrations of rivers and lakes in Japan in recent years and 20 years ago are illustrated in Figure 1. The median values in recent years and 20 years ago are 1.24 mg/L and 1.10 mg/L, respectively, which roughly coincide. As to the other feature of T-N distribution, the points where the concentration exceeded 1 mg/L accounted for as many as 59% of the total, a trend that has continued for the past 20 years. The value of 1 mg/L of T-N is the value that the Japanese Government established as Class V Environmental Standard in Lakes, which is the minimum requirement for lake water and which was set from the Agriculture Water Quality Standard based on paddy field irrigation and for offensive odor problems in lakes. This means that more than half the water bodies in Japan have a high level of nitrogen concentration.

Figure 2 shows the histogram of T-P concentrations in recent years and 20 years ago. The number of points where the concentration was less than 0.05 mg/L has increased in recent years, and the median value of 449 points decreased from 0.071 mg/L 20 years ago to 0.052 mg/L in recent years. The value of 0.1 mg/L T-P was established as Class V Environmental Standard for lakes. Therefore, if the T-P concentration exceeding 0.1 mg/L is considered to be a high level, as it is for T-N, the ratio of high-level points would be 28% in recent years and 32% for 20 years ago. The T-P condition appears to be improved.



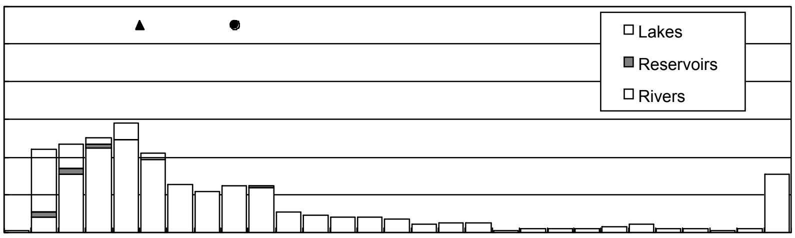


Figure 1. Histogram of T-N concentrations in rivers and lakes in Japan.

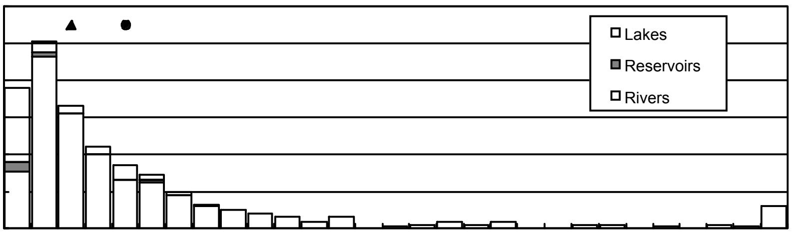


Figure 2. Histogram of T-P concentrations in rivers and lakes in Japan.

Estimation of factors changing T-N and T-P concentrations

Table 2 shows the numbers of points where the nutrient concentrations decreased or increased compared to those from 20 years ago. Here, the point where the difference in the concentration is less than 10% is classified as "unchanging". Paying attention to nutrient concentrations at each point, according to T-N the number of points where the concentration increased is greater than that where the concentration decreased, whereas the T-P concentration had decreased at many points. The cause of the T-P concentration decrease differs among rivers and points, and social factors such as the spread of public sewerage systems, the introduction of nutrient effluent standards and the diffusion of phosphorus-free detergent.

Table 2. Changes of T-N and T-P concentrations in last 20 years at water quality measuring points.

T-N	Rivers	131 (35%)	135 (36%)	113 (30%)	379 (100%)
	Reservoirs	2 (22%)	4 (44%)	3 (33%)	9 (100%)
	Lakes	11 (18%)	26 (43%)	24 (47%)	61 (100%)
	Total	144 (32%)	165 (37%)	140 (31%)	449 (100%)
T-P	Rivers	229 (62%)	78 (26%)	64 (17%)	371 (100%)
	Reservoirs	7 (88%)	1 (13%)	0 (0%)	8 (100%)
	Lakes	46 (64%)	11 (16%)	11 (16%)	68 (100%)
	Total	282 (59%)	90 (20%)	75 (17%)	447 (100%)

Figure 3 shows the trend of Japanese population and the spread of sewerage service during the past 40 years. The sewerage-serviced population rate was 30% in 1980, but it increased to 62% by 2000. In accordance with this rapid spread, the BOD values in Japanese rivers decreased significantly; however, the effect on nutrient concentration is not clear, for the removal rate of T-N and T-P is usually not very high, about 30% in conventional sewage treatment (PWRI 1988). Almost all the large cities in Japan face the ocean and bays, and the effluent from big sewage treatment plants is discharged directly into the sea. In this case, the bypass effect by the extended sewerage system may decrease the nutrient concentration in the rivers.

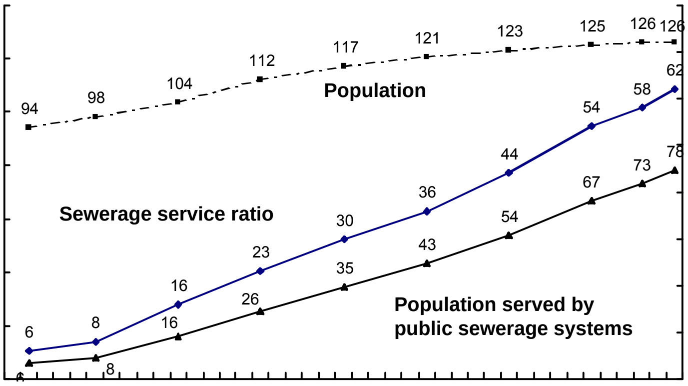


Figure 3. Changes in coverage by public sewerage systems in Japan.

Nitrogen and phosphorus in the fertilizer that is applied to paddy and farmland may reach the rivers through groundwater or rain runoff. The trend of N and P amounts from fertilizer in Japan was arranged using statistical data of production and shipping amounts shown in the "Yearbook of Fertilizer" and standard N and P contents of each product. Figure 4 shows the result of this arrangement. The amount of N and P from fertilizer reached its peak in the 1970s and has gradually decreased in recent years.

Feeding numbers of cattle (beef cattle and milk cattle) and pigs were arranged using "Livestock Statistics" and the N and P loads were calculated by multiplying the N and P generation unit load. Figure 5 shows the result of this arrangement. The amount of N and P loads from livestock matches that from fertilizer and reached its peak in the 1990s.

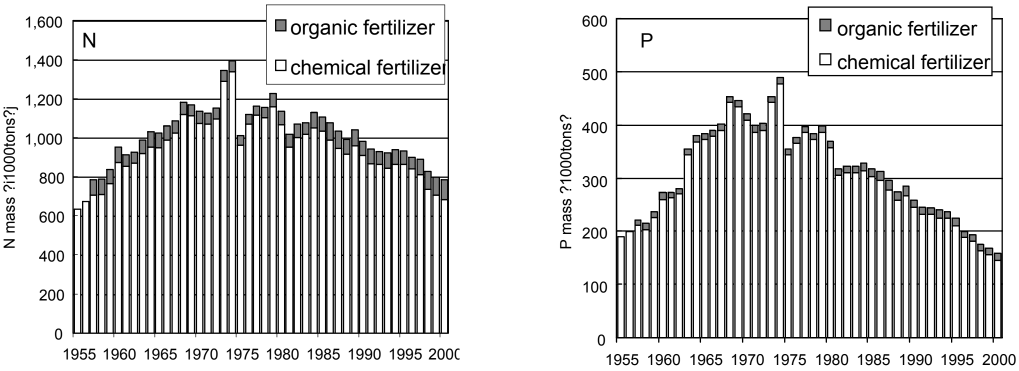


Figure 4 Yearly change of N and P mass in fertilizer (in Japan).

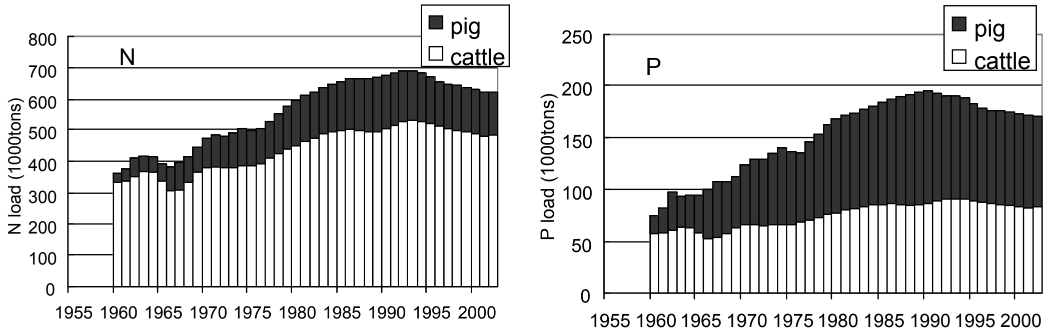


Figure 5 Yearly change of N and P loads from livestock.

The introduction of phosphorus-free detergent was one of the major factors that affected phosphorus concentration in public water bodies. Responding to the increase in eutrophication of Lake Biwa, a typical Japanese lake, the movement to use soap powder instead of synthetic detergent containing phosphorus spread among the residents of Shiga Prefecture where Lake Biwa is situated. In response, Shiga Prefecture enacted the "Lake Biwa Eutrophication Control Ordinance" in 1979 and started effluent control of nitrogen and phosphorus for the first time in Japan, and banned the use of detergent containing phosphorus.

The soap and detergent industry therefore changed detergents with phosphorus builder to phosphorus-free detergents; subsequently, almost all detergent in Japan became phosphorus-free within five years. Figure 6 illustrates the data of detergent production originating from the Japan Soap and Detergent Association. Rapid production of phosphorus-free detergent began in 1980 and resulted in the significant reduction of phosphorus load from domestic wastewater. The unit P load per capita in domestic wastewater was changed from 1.8 g/capita/day to 1.2 g/capita/day in the planning of water quality control (JSWA 1993). Calculation of P load based on the population at that time indicates the reduction of 25,600 tons of P load. By dividing by approximately 250 billion m^3 of yearly discharge of Japanese rivers, this amount is equivalent to a concentration of 0.1 mg/L. As this calculation ignores the pollutant runoff coefficient for rivers, the result may not be so simple; however, the introduction of phosphorus-free detergent must be a major factor for the decrease of P concentration in river waters during the last 20 years.

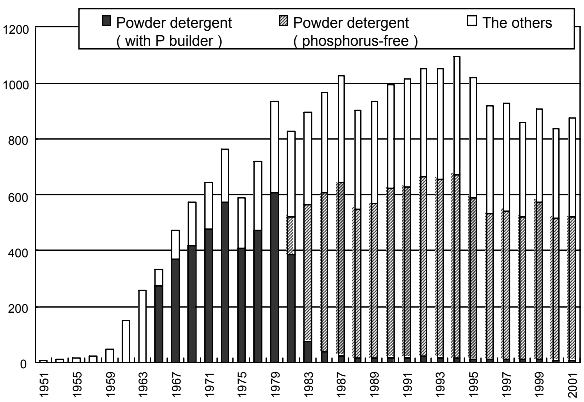


Figure 6 : Yearly change of detergent production.

Case study of Nitrogen and Phosphorus loads in the watershed of lake Inba-numa

Lake Inba-numa is located in the northwestern part of Chiba prefecture, where rapid urbanization occurred when Tokyo expanded. The watershed of this lake is 488.6 km^2, and the outflow flows through the Nagato river into the Tone river. Lake Inba-numa is composed of two parts, North and West. The surface area is 11.55 km^2 in total, 5.29 km^2 in the North part and 6.26 km^2 in the West part. The lake is shallow, with a maximum depth of 2.5 m and mean depth of 1.7 m. The water is used for drinking-water supply, industry, and agriculture, and fishery. Lake Inba-numa was selected as a designated lake based on the Lake Law in December 1985, since when efforts have been made to improve water quality based on the Plan for the Preservation of Lake Water Quality.

A case study on the chronological changes of nitrogen and phosphorus loads in the watershed of Lake Inba-numa was carried out as follows. Since 1985, when the Plan for the Preservation of Lake Water Quality was drawn up to improve the water quality of Lake Inba-numa, frames of the watershed have been surveyed every year, and the pollutant loads from the watershed have been calculated. The concentration of nitrogen in the lake water has been measured since 1972, and phosphorus has been measured since 1978.

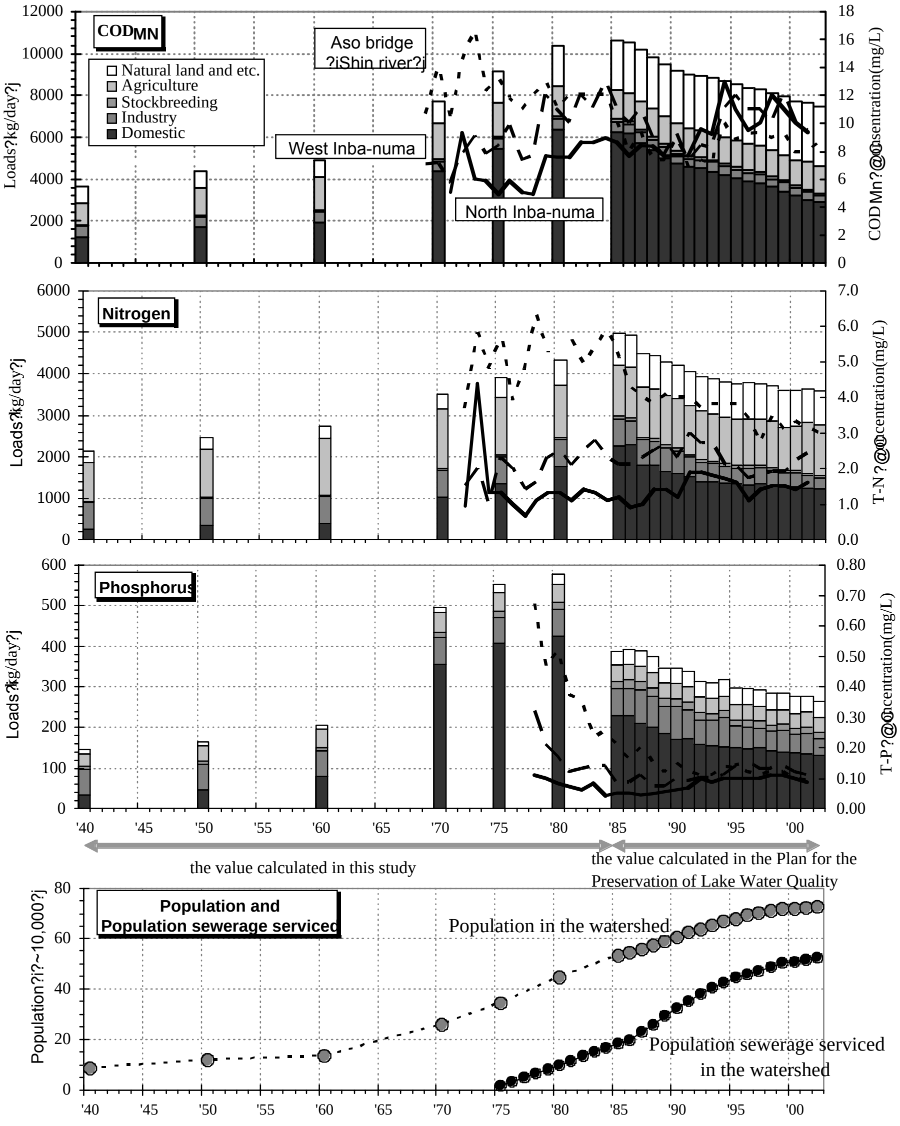


Figure 7. Chronological trend of pollutant loads, and population in the watershed of Lake Inba-numa.

For the years before 1985, the pollutant loads from the watershed were calculated based on the following assumptions. These calculations were carried out beginning in the 1940s when data could be gathered, and the load on the lake was minimal.

- Domestic load: The unit pollutant load originating from gray water was set at the same value in recent years. But for phosphorus the unit load was changed, as in Japan phosphorus was once used in detergent production and turned to phosphorus-free detergent, as referred to in the previous chapter. The unit load of the septic tank was fixed at a recent value. The load from the three nightsoil treatment plants was fixed at the value in 1985.

- Industry load: The unit load of the industry in 1985 was applied.

- Stockbreeding loads: The livestock population was calculated from the recent frames and the national trend, and the unit load in recent years was applied.

- Agriculture load: The unit load of agriculture was set at the 1985 value which was calculated by the national trend of fertilizer amount used and gross agricultural area.

Load from natural land, etc.: The loads from forests, urban areas, and rainfall were set at the values of 1985.

Figure 7 shows the result of the load calculations for COD_{MN}, T-N and T-P in the watershed. Since 1985 the loads from the watershed have decreased every year. From 1940 to 1980 the loads calculated in this study increased as the population in the watershed increased, and the peak loads are estimated to be between 1980 and 1985. Since then, the loads have decreased as sewerage systems have spread. In 2002 the value for nitrogen is estimated to have decreased to near the level in 1970, and the value for phosphorus is estimated to have decreased to below the level in 1970 maybe perhaps because of the introduction of phosphorus-free detergent.

Figure 7 shows the yearly change of water quality at West Inba-numa and North Inba-numa as well as at Shin river which is a typical river flowing into the lake. Regarding the water quality of the lake, the concentration of phosphorus decreased rapidly in the early 1980s, but has tended to increase since then. The concentration of nitrogen hardly changed during this period.

The loads of the watershed are estimated to have decreased in recent years, but the nutrient concentrations of the lake have remained stable or tended to increase slightly. In Japan there is a similar situation in other lakes where water quality has not been improved yet, even though the nutrient load on the lakes has decreased. Further study is needed to restore a good water environment in lakes that once experienced severe eutrophication.

Conclusions

The following conclusions were reached after examining recent data on T-N and T-P concentrations in rivers and lakes in Japan.

1. More than half the water bodies in Japan have a high nitrogen concentration (higher than 1 mg/L), a trend that has continued for the past 20 years.

2. The concentrations of T-P have decreased compared to those of 20 years ago.

3. The introduction of phosphorus-free detergent must be a major factor for the decrease of P concentration in river waters during the last 20 years.

4. In the watershed of Lake Inba-numa, where a case study was carried out, the nutrients loading has been reducing certainly in these ten years, but the water quality improvement has not been observed yet in the lake. This situation seems to be common for the other lakes in Japan where severe eutrophication once progressed.

Acknowledgements

The authors would like to thank the staff of Chiba Prefecture for their generous support for arranging data on the case study of Lake Inba-numa.

References

Japan River Association, 2000. *Yearbook of River Water Quality in Japan, 1997.* (in Japanese)

Japan Sewage Works Association, 1993. *Manual of Comprehensive Basin-wide Planning of Sewerage Systems.* (in Japanese)

Ministry of Construction, River Bureau, 1999. *The 38th Water Quality Annual, 1997.* (in Japanese)

Public Works Research Institute, 1988. *Report of planning of advanced wastewater treatment and its introduction program, PWRI Technical Memorandum No.2633.* (in Japanese)

GIS spatial characterization of water quality in three bays of Lake Victoria, Uganda with different historical catchment land use

Stephen Brian Kintu Sekiranda [1], James Okot-Okumu [2], Fredric William Bourdon Bugenyi [3] and Janet Sebuyira Naluwairo[1]

[1]Fisheries Resources Research Institute (FIRRI), P.O. Box 343 Jinja, Uganda
[2]Makerere Institute of Environment and Natural Resources, MUIENR, P.O. Box 7062, Kampala, Uganda
[3]Department of Zoology, Makerere University, P.O. Box 7062, Kampala, Uganda
Email: skintu2000@yahoo.com

Abstract

Geographic Information System (GIS) was used to delineate conductivity, water transparency, dissolved oxygen, pH, total phosphorus, and chlorophyll- a within Murchison, Fielding and Hannington bays. The aim was to show the degree to which water quality in the three shallow bays varies relative to the adjoining catchment land use pattern. Murchison is drained by an urban catchment land use while Fielding and Hannington bays are drained by suburban and rural catchment land uses respectively.

Water quality in Hannington and Fielding bays did not vary much among the sampling sites as it did in Murchison Bay. Conductivity values ranged from 110 to 305 µS cm^{-2} in Murchison Bay while in Fielding and Hannington respectively, it ranged from 98 to 110 µS cm^{-2}. Dissolved oxygen (DO) averaged 12.7 mg L^{-1} in Fielding, 6.8 mg L^{-1} in Hannington and 6.3 mg L^{-1} in Murchison Bay. However, at Nakivubo channel mouth site in Murchison bay, DO was below 2 mg L^{-1}. Mean total phosphorus in inner Murchison Bay ranged from 557 to 1046 $\mu g\ l^{-1}$ while in the other two bays the range was between 56 and 72 $\mu g\ l^{-1}$. Comparative spatial delineations of water quality parameters in the three bays identified Murchison Bay to have the highest degree of water quality impairment.

These sort of spatial delineations of water quality relative to adjoining catchment land uses illustrate the importance of integrating water-quality management and land-use planning in developing countries like Uganda. This implies that based on the understanding of land-water relationship in a watershed context, planners and policy-makers at different levels can be brought together for Best Management Practices (BMP) to prevent pollution and to plan for a sustainable future of the lake and its catchment's development.

Key words: bays, water quality, degradation, catchment

Introduction

Lake Victoria, Africa's largest and second largest lake in the world after Lake Superior is suffering from eutrophication. The lake was oligotrophic before 1960 (Talling, 1965). Eutrophication arose from increased inflow of nutrients into the lake from the catchment (Ochumba and Kibara, 1989; Hecky 1993; Hecky *et al*., 1994, Mugidde, 1993), which has led to gradual deterioration of water quality in the past few decades.

Gradual declining water quality of the lake is of concern to scientists and managers as far as fisheries and fish habitat are concerned. For instance, hypolimnetic waters (depth > 35 m) have become anoxic or hypoxic due to eutrophication and such conditions have deprived fish of vital habitat space. Hypoxicity is even exacerbated by marked stratification that lasts from September to April (Mugidde, 1993; Hecky *et al.,* 1994). Persistent hypoxia under such conditions promotes increase in phosphorus concentration in the hypolimentic zone to such levels that induce silicon (Si) deficiency (Schelske, 1988). This scenario is already apparent in Lake Victoria where a ten- fold decrease in Si concentrations from 22 - 60 *u*M over the annual cycle in the 1960s to 0.8-8.7 *u*M in the 1990s has occurred (Hecky, 1993; Lehman and Branstrator, 1993; Hecky and Bugenyi ,1992).

The deoxygenation of the lake's bottom waters now poses a threat as far as fish habitable space is concerned and when oxygen-deficient waters are brought to surface lake layers especially in shallower portions of the lake by periodic upwelling, massive fish kills result (Hecky, 1993). Eutrophication is not only a great challenge to water managers and researchers in identifying its symptoms, but also is daunting task to deal with those problems for such a vast lake. No single simplified model of this complex process has been able to attained this objective. The problem can be traced to land-based human activity, primarily in the forms of untreated sewage and effluent, and nutrient run-off from agricultural lands.

In this paper, we compare the water quality state among three shallow bays in northern Lake Victoria, Uganda surrounded by catchments of urban, semi-urban and rural land uses respectively using trophic state indicators of seicch transparency, total phosphorus and chlorophyll-a as delineated by GIS.

Materials and methods

Total phosphorus (TP), ortho- phosphorus, Chlorophyll-a, pH, conductivity, and dissolved oxygen and secchi transparency were examined at 6 geo-referenced sampling points respectively in Murchison, Fielding and Hannington bays along two transects spaced 60 m apart from the lake edge to assess water quality (Figure 1).

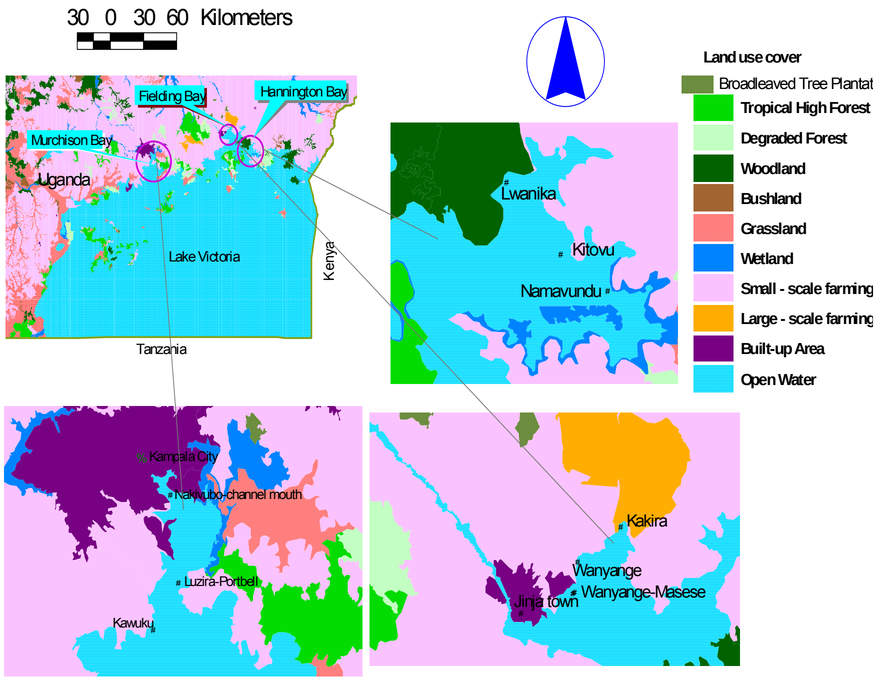


Figure 1 The study bays, sites and associated catchment land use cover.

Three sites were sampled 8 times between June 2001 and April 2002 respectively in each bay as shown in Figure 1. Murchison Bay is surrounded by a catchment mainly comprised by agricultural, commercial, industrial and residential land uses. Fielding Bay catchment is dominated by large-scale sugar cane farming and semi-urban land uses. Rural small-scale agriculture and sparse human settlements mainly comprise Hannington Bay catchment (Figure 1).

The ascorbic acid modification of the molybdenum blue reaction was used to determine ortho-phosphorus and total Phosphorus (TP) after acid persulphate oxidation. Chlorophyll-a (Chl-a) was determined spectrophtometrically after extraction with ethanol. Temperature, pH, and conductivity were measured in-situ with WTW LF 330 Conductivity meter. Dissolved oxygen was measured in-situ using WTW OXI 320 oxygen probe. Transparency was estimated using a 25-cm black-white secchi disc.

Carlson Trophic State Indices (TSIs) were calculated from Secchi depth (SD), Chl-a, and TP concentration (Carlson, 1977; Carlson and Simpson, 1996) thus:

i) TSI (Chl-a) = *30.6 + 9.8 In (Chl-a)*

ii) TSI (TP) = 4.15 + 14.42l In (TP)

iii) TSI (SD) = 60-14.4l In (SD)

The indices along with surface dissolved oxygen (DO), pH and conductivity were spatially delineated using GIS- ArcView 3.2 in the three bays relative to surrounding catchment land use pattern. Analysis of Variance (ANOVA) at p= 0.05 level of significance was used to examine the variability in the underlying patterns. Results

Estimates of non- point total-phosphorus (TP) loads into the bays

An estimated 4159 Kg day^{-1} TP from a population equivalent (PE) of 702,180 people is loaded to Murchison Bay from its catchment. To Fielding Bay, 1 Kg day^{-1} TP load from PE of 495 in its catchment was estimated and to Hannington Bay 2 Kg day^{-1} TP was estimated from PE of 2119 in its catchment (Table.1). More than 95 % of the load in Murchison Bay arose from Kampala City but contributions from elsewhere included Luzira prison, Portbell Trading Center and Gaba Fishing Village (Table.1). In Fielding Bay, Wairaka and Wanyange fishing villages were the major sources that were identified while in Hannington Bay, Kiterera and Ikulwe Trading Centers, and Musoli and Lwanika Fishing Villages were the majors sources were recognized (Table 1).

Table 1. Non point TP loads into Hannington, Fielding and Murchison bays of Lake Victoria.

Bay	Urban Centre	Type of urban centre	Contributing population	TP Estimated loads to Lake Victoria, Kg/day
Hannington	Lwanika	Fishing village	269	0
	Musoli	-Do-	237	0
	Ikulwe	Trading centre	941	1
	Kiterera	-Do-	672	1
Fielding	Wairaka	Fishing Village	158	0
	Wanyange	-Do-	337	1
Murchison	Kampala	City	688,830	4,012
	Luzira Prison, K'la	Prison	9,277	95
	Portbell	Trading centre	3,375	34
	Gaba	Fishing village	698	18

(Adapted from Mott and MacDonald consultancy report (2001)

Water quality within Bays

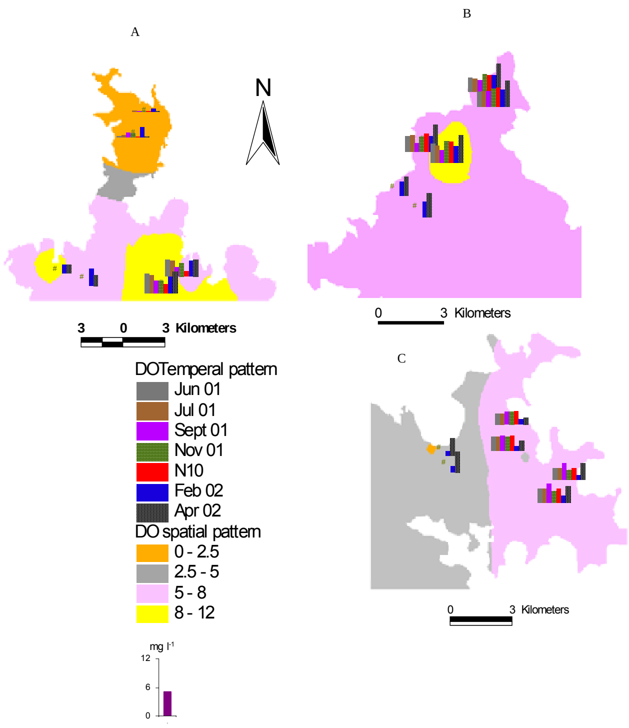


Figure 2. Spatial and temporal variation of dissolved oxygen in Fielding, Hannington and Murchison bays of L. Victoria (Uganda).

Dissolved oxygen in Murchison Bay ranged from nil to 9.4 mg l^{-1}, averaged 6.8 ± 2.4 mg l^{-1} but barely exceeded 2.5 mg l^{-1} at the Nakivubo channel mouth (Figure 2). In Fielding Bay, DO was more than 5 mg l^{-1} up to 12 mg l^{-1}. In Hannington Bay, the range was 2.3 - 7.7 mg l^{-1}. Single factor ANOVA revealed significant differences ($F_{5,26}$ = 16.323 > F_{crit}= 2.587) in mean dissolved oxygen among sites within Murchison Bay whereas in both Fielding ($F_{5,26}$ = 0.450 < F_{crit}= 2.587) and Hannington bays ($F_{5,26}$ = 0.488 < F_{crit}= 2.587), differences were not significant. Unlike DO, Secchi transparency was comparable in the three bays i.e. 0.7 m in Murchison Bay, 1.0 m in Fielding Bay and 0.8 m in Hannington Bay.

Conductivity range in Murchison Bay was 101.0-303.8 µS cm^{-2} and averaged 131.1 ± 44.0 µS cm^{-2} whereas in Hannington Bay, the range was 97.0-106.1 and averaged 102.0 ± 1.9 µS cm^{-2} and in Fielding Bay, it was 98.2-107.4 µS cm^{-2} with a mean of 102.9 ± 1.5 µS cm^{-2} .

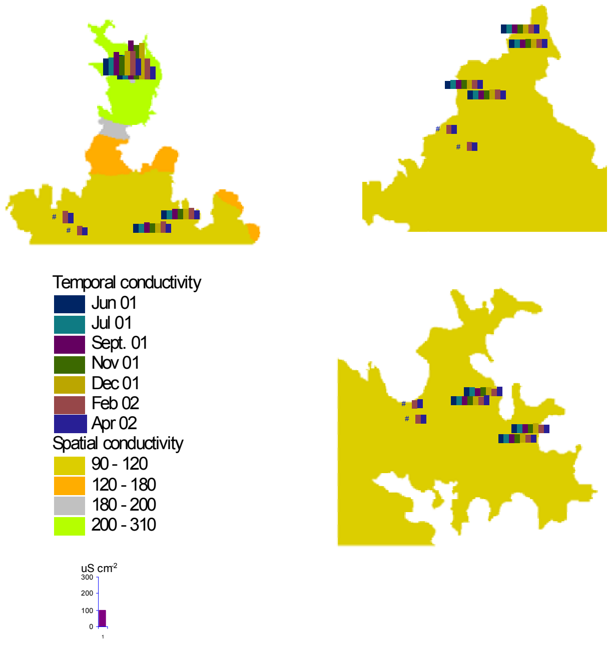


Figure 3. Temporal and spatial patterns of conductivity, uS cm^{-2} in Murchison, Fielding and Hannington bays of Lake Victoria located in catchment of varying land use history.

Bay (Figure 3). The spatial pattern is illustrated in Figure 3. ANOVA test on mean conductivity indicated insignificant differences in Fielding Bay ($F_{5,26}$ = 1.233 < F_{crit}= 2.587), and in Hannington Bay ($F_{5,26}$ = 0.122 < F_{crit}= 2.587).

For pH, ANOVA F-test showed no significant differences in Fielding Bay ($F_{5,26}$ = 0.429 < F_{crit}= 2.587) as well as in Hannington Bay ($F_{5,26}$ = 0.165 < F_{crit}= 2.587) . Conversely, significant differences were highlighted for Murchison Bay sites ($F_{5,26}$ = 8.474 > F_{crit}= 2.587) as spatial patterns on the maps clearly illustrate (Figure 4).

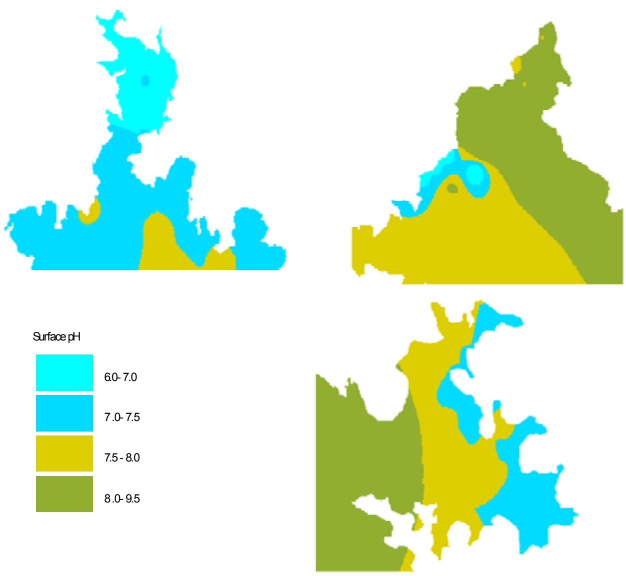


Figure 4. Spatial variation of pH in Fielding, Hannington and Murchison bays of L. Victoria (Uganda).

Trophic State Index (TSI): bounds of eutrophication in the bays

The lower boundary of chlorophyll-a for eutrophic conditions is 7 µg l^{-1} (Table 2). However, in all bays, mean chlorophyll-a was > 60 µg l^{-1} (Table 2). TSI values varied between 55.5 and 86.4 in Murchison Bay (Table 2.3), indicating eutrophic conditions in most of the bay to hypereutrophic conditions at Nakivubo channel mouth (Fig 5).

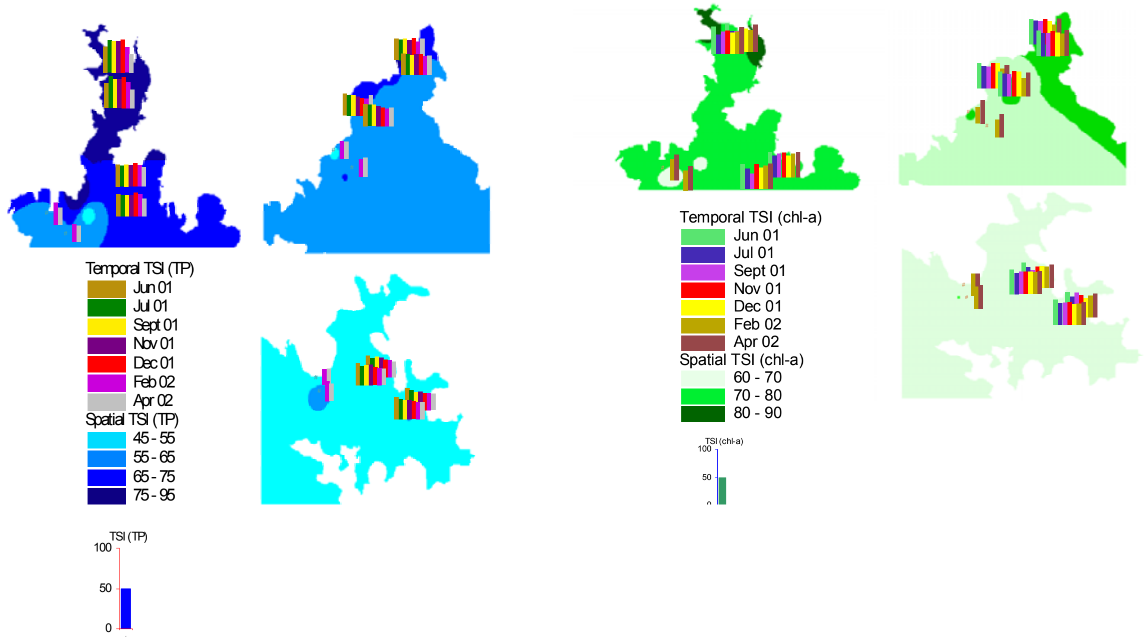


Figure 5. Temporal and spatial patterns of TSIs on TP, Chl-a and SD in Murchison, Fielding and Hannington Bays of Lake Victoria located in catchment of varying land use history.

Table 2. Total phosphorus, Chlorophyll-a and seichi depth and Trophic State Indices, and extent of eutrophication in three nearshore bays of Lake Victoria, Uganda: 2001-2002.

	TP (µg l^{-1})			Chl-a (µg l^{-1})			Secchi (m)			Carlson TSI***		
TROPHIC STATUS	O	M	E	O	M	E	O	M	E	O	M	E
STANDARD CRITERIA **	<11	11-24	>24	<3.0	3.0-7.0	>7.0	>4.0	2.2-4.0	<2.2	<35	40-50	>55
Murchison Bay	**21.6-1060.0** (107.2 ± 58.0)			**12.7-211.5** (71.9 ± 43.4)			0.4-1.0 (0.7 ± 0.1)			55.5-86.4		
Fielding Bay	**40.5-115.2** (83.1 ± 21.4)			**2.4-82.5** (60.8 ± 27.5)			0.7-1.3 (1.0 ± 0.1)			38.8-73.8		
Hanninton Bay	**29.9-164.21** (94.3 ± 17.8)			**30.2-80.5** (65.3 ± 14.6)			0.6-1.0 (0.8 ± 0.1)			53.0-77.6		

The standard criteria represent median values from published scientific literature O= Oligotrophic, M= mesotrophic, E = eutrophic * From Carlson (1977).

The TSI values varied between 38.8 and 73.3 in Fielding and Hannington bays, a situation that puts both bays within bounds of eutrophic conditions (Figure 5); and from 53.0 to 73.5 in Murchison Bay, a range that denotes hypereutrophic conditions. ANOVA test showed no significant differences in TSI-TP in Fielding Bay ($F_{4,20}$ = 0.917 < F_{crit}= 2.866) and in Hannington Bay ($F_{3,19}$ =0.049 < F_{crit}=3.239). Similarly, no significant differences in TSI-chlorophyll-a were shown in Fielding Bay ($F_{4,20}$ =0.142 < F_{crit}=2.866) and in Hannington Bay ($F_{3,19}$ =0.516 < F_{crit}=3.239) . No significant differences as well were shown for TSI-SD in Fielding ($F_{4,20}$ =0.433 < F_{crit}=2.866) and in Hannington ($F_{3,19}$ =0.073 < F_{crit}=3.239). On the contrary, the test showed significant differences for TSI-TP ($F_{3,19}$ = 23.843 > F_{crit}=3.239) and TSI-SD ($F_{3,19}$ =20.686 > F_{crit}=3.239) in Murchison Bay. With respect to TSI -chl-a though, no significant differences ($F_{3,19}$ =0.504 < F_{crit}=3.239) were detected by ANOVA test.

Discussion

A high flux of phosphorus was observed in Murchison Bay compared to either Fielding or Hannington bays probably due to high external phosphorus loading from the catchment, largely dominated by agricultural, commercial, industrial and residential land uses. Consequently, Trophic State Indices (TSIs) were substantially higher in Murchison Bay (> 65) than either in Fielding or Hannington bays (45-65), indicating hypereutrophic conditions in the former and eutrophic ones in the latter. For catchments of Fielding and Hannington bays that are dominated by large-scale sugar cane farming and semi-urban land uses, and rural small-scale agriculture and sparse human settlements respectively, it is judicious to expect lower external TP loading to these bays than to Murchison Bay. Water quality was poorest in terms of specific conductance in Murchison Bay (up to 320 µs cm^{-2}) when compared to either Fielding or Hannington Bay whose range was (90-110 µs cm^{-2}). The high level conductivity in Murchison Bay relative to either Fielding or Hannington Bay (Fig 3) possibly indicated a higher level of pollutants in the water in the former than the latter.

Apart from external loading, internal phosphorus release from manganese and iron complexes in lake sediments could be another source of phosphorus in the lake. This happens mostly from water bodies that suffer from hypolimnetic deoxgyenation (Dillon & Rigler 1974, Schindler 1977, Lipiatou *et. al.* 1996, Hecky *et. al.* 1996. Today, Lake Victoria is a victim of hypolimnetic deoxgyenation (Ochumba and Kibaara 1989, Hecky *et. al.* 1994). Phosphorus levels in bottom waters in the lake have been noticed to be gradually increasing from baseline levels before the 1960s due to a reducing environment generally associated with deoxygenation of waters overlying the sediments. Oxygen concentrations near or below 1 mg l^{-1} in deep waters prior to overturn have been reported (Hecky *et. al.,* 1994). Vollenweider (1970), Schindler (1977), Dillon and Rigler (1974), Lehman & Branstrator (1994) have observed that significant phosphorus release rates from sediments occur once oxygen levels drop below 2-3 mg L^{-1}. In this study, such low dissolved oxygen values (Figure 2) and elevated total phosphorus concentrations (Figure 5) were observed at Nakivubo channel mouth in Murchison Bay.

The oligotrophic conditions of the lake prior to the 1960s circumvented the internal hypolimnetic enrichment of P because such conditions bolster retention rather than release of P (Hecky *et a.,l* 1994). The eutrophic conditions the lake are diminishing phosphorus retention capacity of the sediments and instead augmenting internal hypolimnetic enrichment of P from the sediment (Hecky, 1993). Therefore, hypolimnetic internal phosphorus loading could be one of the formidable sources of phosphorus in the lake. Apart from hypolimnetic internal phosphorus loading, in inshore areas especially in unsheltered bays like Fielding Bay, sediment-bound phosphorus can be constantly suspended, deposited, and resuspended as a result of wave perturbations (Hecky, 1993). Besides dissolved oxygen, factors such as pH, temperature, ebullition, and bioturbation, nitrate concentration, microbial activity and macrophyte exudation affect rate of p flux from the sediments (Dillon and Rigler, 1974).

Conclusions and recommendations

Though phosphorus loading is a problem in the entire lake, it is evidently an acute one in Murchison Bay. Therefore, big cities or municipalities around the lake should make efforts to minimize the loading of phosphorus whenever possible from the catchment, e.g., using properly designed sedimentation basins to treat urban stormwater. Additionally, phosphorus levels in detergents and soap should be reduced.

In order to have a complete nutrient budget for the lake or parts of the lake like embayments and be able to diagnose the source of eutrophication, loading (flow and concentration) in relation to precipitation/runoff events and river inflows should be monitored to provide useful information on relationship of in-lake water quality over time. At this point, it would be premature to recommend water quality goals for the lake without a better understanding of the relationship of in-lake water quality mass balances.

Lakeshore residents or other persons who routinely use the lake are encouraged to participate in the Best Management Practices of land and the lake.

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References

Hecky, R.E., 1993. The eutrophication of Lake Victoria. *Verh. Internat. Verein. Limnol*. 25: *39-48.*

Hecky, R.E. & Bugenyi. F.W., 1992. Hydrology and chemistry of the African Great Lakes and water quality issues: Problems and solutions. *Mitt. Verh. Limnol. 23: 45-54.*

Hecky, R.E., Bugenyi F.W., Ochumba P., Talling J.F., Mugidde R., Gophen M. & Kaufman L., 1994. Deoxygenation of the deep water of Lake Victoria, East Africa. *Limnol. Oceanogr*. 39: 23-36.

Hecky, R.E., Bootsma H.A., Mugidde R. & Bugenyi F.W., 1996. Phosphorus pumps, nitrogen sinks and silicon drains: plumbing nutrients in the African Great Lakes. *In.* Johnson, T.C. and Odada, E., eds. *The limnology, climatology and paleoclimatology of the East African lakes.* Gordon and Breach, Toronto, pp.205-224.

Lehman, J.T., & Branstrator D.K., 1993. Effects of nutrient grazing on the phytoplankton of Lake Victoria. *Verh. Internat. Verein. Limnol. 25: 850-855.*

Lehman, J.T., & Branstrator D. K, 1994. Nutrient dynamics and Turnover rates of Phosphates and Sulphates in Lake Victoria, East Africa. *Limnol. Oceanogr*.

Lipiatou, E., Hecky R.E., Eisenreich S.J., Lockhart.D., & Wilknson P., 1996. Recent Ecosytem changes in Lake Victoria reflected in sedimentary natural rocks and anthropogenic organic compounds. *In.* Johnson, T.C. and Odada, E., eds. *The limnology, climatology and paleoclimatology of the East African lakes.* Gordon and Breach, Toronto, pp.523-541.

Mott and MacDonald consultancy report, 2001 "Management of Industrial and Municipal Effluents and urban Runoff in Lake Victoria Basin"Final report vol. 1 2001.

Mugidde, R., 1993. The increase in phytoplanton primary productivity and biomass in Lake Victoria (Uganda). *Verh. Internat. Verein. Limnol.* 25, 846-849.

Ochumba, P.B.O. & Kibaara D.I., 1989. Observations on blue-green algal blooms in the open waters of Lake Victoria, Kenya. *African Journal of Ecology.* 27, 23-24.

Schelske, C.L. 1988. Historic trends I Lake Michigan silica concentrations. *Int. Revue. ges. Hydrobiol.* 73, 559-591.

Schindler, D.W., 1977. The evolution of phosphorus limitation in Lake: natural mechanisms compensate for deficiencies of nitrogen and carbon in eutrophied lakes. *Science (W.D).* 195, 260-262.

Talling, J.F., 1965. The photosynthetic activity of phytoplankton in East African lakes. *Int. Rev. ges. Hydr.* 51, 1-32.

Impacts of upland land use on downstream water quality in River Njoro Watershed, Kenya

Shivoga William A.[1], Mucai Muchiri[2], Samuel Kibichi[2], Jethro Odanga[3], Scott N. Miller[4], Tracy J. Baldyga[4], Maina C. Gichaba[5]

[1]Department of Environmental Science, Egerton University, P.O. Box 536, Njoro, Kenya
Email: shivogawa@yahoo.co.uk
[2]Department of Fisheries, Moi University, P. O. Box 3900, Eldoret, Kenya
[3]Department of Wildlife Management, Moi University, P. O. Box 3900, Eldoret, Kenya
[4]Department of Renewable Resources University of Wyoming, Department 3354, 1000 University, WY 82071

Abstract

Data recorded from 10 sampling sites along River Njoro are used to examine the contribution of nutrients from subwatersheds land upstream draining each of the sampling sites. The data is also used to assess whether both the proportion of land uses and the size of subwatersheds account for the variability in water quality in River Njoro Watershed. Geographical Information System (GIS) analysis was used to determine the spatial distribution of land cover types and subwatershed contributing runoff to the sampling sites in the river. Standard DEM-based routines were used to establish the watershed contributing runoff to each site. Water and sediment samples were collected for chemical analysis and nutrient levels related to upstream land use types and size of subwatersheds. The mid-stream portion of the river near Egerton University, accounts for the highest contribution of the nutrients. The percentage contribution is magnified by additions from industrial, human settlements and agricultural land uses around the University. There is, however, significant decrease in nutrient levels downstream indicating natural purification as the river flows through an area of large scale farming with intense well preserved riparian and in-stream vegetation. Steep slopes of the land upstream before Egerton University enhances erosion and nutrient losses from the subwatersheds. Small scale agricultural and bare lands contribute over 55% of phosphorus load to the upper and mid reaches of River Njoro. The size of subwatershed account for about 53% variability in soluble phosphorus in the river. Land-use subwatershed proportions are important for characterising and modelling water quality in the watershed. Upland land uses are as important as near-stream land uses. We suggest that conservation of intact riparian corridor along the river and its tributaries contributes significantly to natural purification and recovery of the ecological integrity of the River Njoro ecosystem.

Key words: Riparian zone, upland land use, water quality, nutrient levels

Introduction

Human-induced disturbances due to land use activities have the greatest potential for introducing enduring changes to the ecological structure and functions of watersheds. Over the years, degradation of water resources within the River Njoro watershed has occurred due to a number of factors. High population growth rate and the associated change in land use have placed high demands on the watershed resources, thereby upsetting environmental stability (Karanja et al. 1989). This has impacted negatively on the ecological integrity and hydrologic processes in the River Njoro watershed (Chemilil, 1985; Bretschko, 1995; Mathooko, 2001; Shivoga, 2001).

The sprawl patterns of human settlements and land use change have been implicated as the primary causes of deteriorating health of the River Njoro watershed. However, little is known about the quantitative influence of specific land use patterns on the water quality of River Njoro. Land cover classification using Landsat images (Baldyga et al., 2004) show loss of about 20% in both indigenous and plantation forest between 1986 and 2003 in the upper River Njoro Watershed. There was also reduction of 6% in large scale farms areas. The conversion of forested and large scale farm areas has been accompanied by increase in small-scale mixed agriculture and human settlements.

This paper discusses the relationship between upstream proportion of land use types and riparian buffer zone to nutrient concentrations and supply to downstream reaches of the river. Further, it discusses the relationship between the proportion of land uses and the size of delineated subwatersheds (an area of land draining to a specific sampling site in the river) and the water quality in the upper and mid-streams of River Njoro Watershed. The findings indicate that near stream riparian zone reduces nutrients and sediment influx into the River Njoro channel and provides a natural purification and recovery of the ecological integrity of the river ecosystem downstream. Consequently, we conclude that land use near the stream account for the variability in water quality than land use for the entire River Njoro Watershed.

Materials and methods

Study area

River Njoro Watershed covers an area of 250 km^2 (SUMAWA 2004). The watershed is located in the Rift Valley Province of Kenya at 0°30' South, 35°, 20' East. The river originates in the Eastern Mau Escarpment, part of the Mau Forest Complex, one of the five major water towers for Kenya. River Njoro is about 50 km in length with its source at approximately 3000 m above sea level and mouth in Lake Nakuru at 1,759 m a.s.l. It drains forested and highly populated agricultural lands before serving the towns of Njoro and Nakuru. The lake is enclosed within the Lake Nakuru National Park famous for its large populations of flamingos. River Njoro is structured in a typical riffle-pool sequence, with soft substratum and bedrock in the pool and riffle sections, respectively. The soils in the watershed are predominantly Ultisols and Entisols. Soils textures range from clay loams in the lower portion of the study area to sandy clay loams in the plantation and indigenous forest areas at higher elevations. Mean annual rainfall of the area measured at the Kenya Agricultural Research Institute-Njoro from 1949-2001 is 939.3 mm The rainfall is tri-modally distributed with peaks in April, August and November. Average annual minimum and maximum temperatures in the area are 9° and 24° C, respectively.

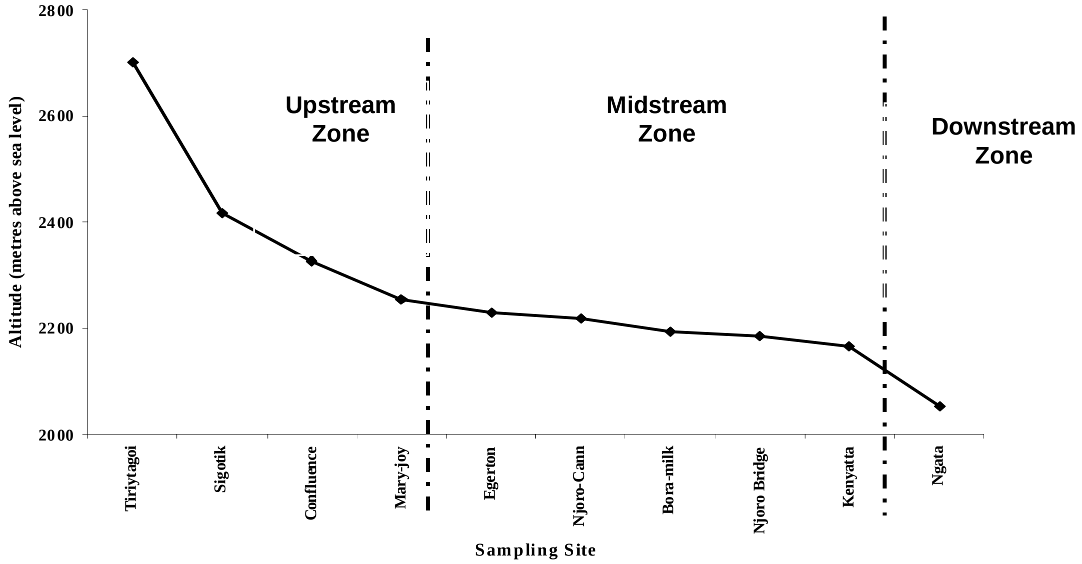

Figure 1. Landscape profile of the location of the sampling sites in upstream and midstream zones relation to the altitude above sea level.

The profile of the landscape elevation gradient of the sampling sites is shown in Figure 1. Based on their altitude sampling sites are categorized into two: upstream zone (Tiriytagoi, Sigotik and Confulence sites) and midstream zone (Mary Joy, Egerton University, Njoro Canning, Bora Milk Plant, Njoro Bridge, Kenyatta and Ngata). The upstream sites are characterised with a steep slope typical of a stream headwater reach and the midstream which has a gentle gradient.

Methods

Ten sampling sites were selected to represent combinations of natural and human factors which influence collectively the physical, chemical and biological characteristics of water quality in the upper and mid-reaches of the River Njoro Watershed. The sampling sites for water quality assessment were established along the river based on the land-uses in the upper and mid stretches of the watershed. Routine monthly sampling was carried out at each of the sampling site to collect water samples for chemical analysis of nutrients (phosphorus, nitrogen and carbon).

Three Landsat TM and one Landsat TM+ scenes were selected for this study. Table 1 lists sensors and acquisition dates for the four images used. Images were selected that correspond by year with census dates for eventual correlation between land cover change and human migration and population increases. Topographic contour maps were acquired and digitized to create a 50m Digital Elevation Model (DEM) using ArcINFO. This resolution allows Automated Geospatial Watershed Assessment (AGWA) to produce a suitable delineation of the watershed and subwatersheds for each sampling site. The subwatersheds represent the areas of land draining to each of the sampling sites in the river.

Table 1. Satellite data used

Image Date	Sensor	Resolution
January 1986	Landsat TM	30m
March 1989	Landsat TM	30m
January 2000	Landsat TM	30m
February 2003	Landsat TM+	30m

Results and discussion

Land cover

Table 2 shows the size and proportions of different land cover types in the upper and mid reaches of River Njoro Watershed contributing runoff to sampling sites. The study area has six land cover classes: indigenous forest, plantation forest, grassland, large scale agriculture, mixed small scale agriculture and bare land. There is a significant increase in the percentage area covered by small scale mixed agriculture and bare land downstream from the uppermost sampled site, Tiriytagoi, to the lower sites around Egerton University. In contrast, there is overall decrease downstream in the land cover area of grasslands, large scale agriculture and indigenous forest from the uppermost site of the watershed.

The greater part of forests has been converted to small-scale agriculture and there is large net loss in large scale agricultural systems. Since there appears to be little change in the proportion of land under plantation forests and grass (Table 2), the increase in the area of bare land and small scale agriculture are, apparently, at the expense of indigenous forests. The trend in forest loss is expected given the historical knowledge of the area. The larger Mau Forest Complex has been undergoing significant losses throughout its extent and (Kenya Forest Working Group 2001. Baldga et al. 2004 reported rapid plantation forest losses between 2000 and 2003 in the uppermost regions of the watershed. By 2003 the majority of plantation forests were converted to small-scale agriculture accompanied with large net loss in large agricultural systems.

Table 2: The size and proportions of land cover area in each watershed contributing runoff to the sampling sites in Upper and Mid Reaches of River Njoro Watershed.

Sampling Site	Subwater-shed Area (km^2)	Agriculture		Mixed agriculture		Bare land		Plantation forest		Indigenous forest		Grass-lands	
		km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%
Tiriytagoi	35.61	1.73	5	5.77	16	1.30	0	1.11	3	19.70	56	7.19	20
Sigotik	74.54	1.74	2	10.61	14	1.73	0	1.80	2	49.63	68	10.60	14
Confluence	79.06	1.75	2	14.20	18	1.90	0	2.31	3	50.02	64	10.60	13
Mary-Joy	118.95	2.00	2	35.08	29	4.18	0	3.32	3	67.17	57	10.97	9
Egerton University	119.07	2.00	2	35.17	30	0.43	0	3.32	3	67.17	56	10.97	9
Njoro Canning	121.62	2.02	2	37.62	29	0.48	0	3.32	3	67.21	55	10.97	9
Total	122.50												

The increase in area covered by mixed small agriculture may be due to increase in human population in the area and the extension of tillable land towards the forest reserves and the stream edge by small-holder farmers. This is leading to degradation of quantity and quality of water resources in the River Njoro watershed.

Spatial change in nutrient levels

Figure 1 shows percentage contribution of nutrients from upstream land uses to downstream sampling sites during the study. The pattern of percentage contribution of nitrates by stretches of land upstream each site follows that of phosphates. Spatially, the subwatersheds around Egerton University (i.e. between Mary Joy and Bora Milk) accounts for the highest percentage contribution of nutrients to the river. This is expected because the major land use activities around the University include human settlement, small scale agriculture, education institutions, vegetable canning factory and dairy plant which are probably the major point and non-point sources of phosphates and nitrates in the watershed. The main source of phosphate in surface waters comes from wastewater from sewage treatment works. The runoff draining the grazed grasslands, diffuse fertilized and manured crop fields and wastewaters from the institutions and factories add significantly to the nutrient load to the river. Detergents used for washing and cleaning in the river, homes, institutions and industries are a major source of phosphorus load to the river. Phosphorus enters the river through sewage inlets and non-point sources, for example washing sites along the river.

The steep slope of the land upstream and around Egerton University enhances erosion and nutrient losses from the subwatershed. Similar observations have been reported by Tiessen, (1995) who observed that greater slopes and a greater density of stream networks in a watershed increases erosion and phosphorus loss from the river drainage.

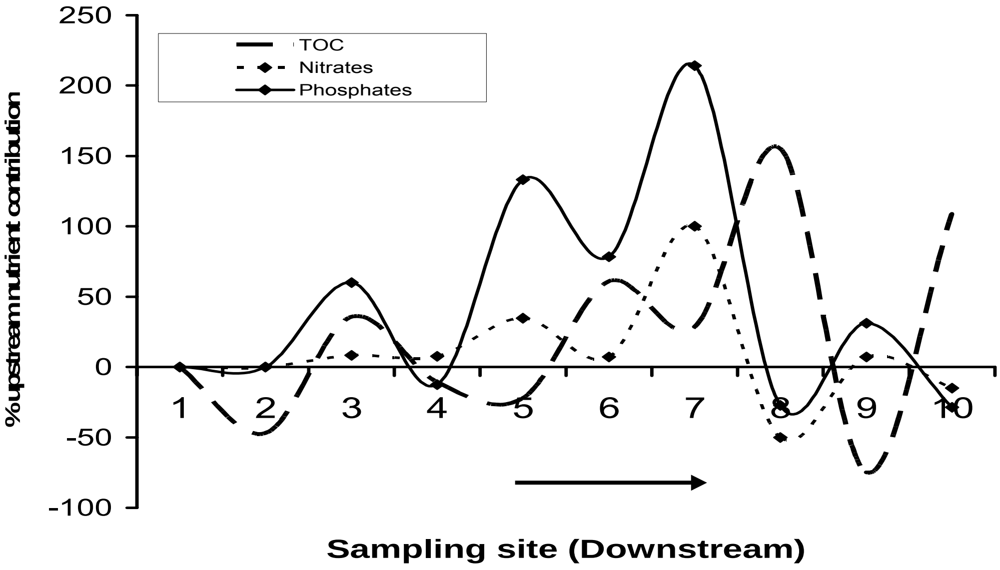

Figure 2. Downstream variations in percentage contributions of phosphate, nitrate and organic matter (TOC) by the stretch of land upstream of each site from the upper most forested site to Ngata the down most site. **1**= Tiriytagoi, **2**= Sigotik, **3**= Confluence, **4**= Mary Joy, 5= Egerton University, **6**= Njoro Canning Factory, **7**= Bora Dairy Plant, **8**= Njoro Bridge, **9** = Kenyatta, **10** = Ngata. (Negative percentage implies that the upstream site has concentration greater than the downstream site so that the actual trend is one of decrease.).

Organic matter contribution by land upstream of Sigotik sampling site is low as compared to the concentrations recorded at the forested Tririytagoi site (Figure 2). This is followed by an increase from Sigotik to the Confluence. There is some relatively intact forest along the stream valley at this stretch of the river.

It is noteworthy that low levels of phosphates and nitrates were recoded at sampling sites in the stretch of the stream between Bora Milk and Njoro Bridge (Figure 1). This is likely a consequence of the undisturbed dense riparian forest and instream vegetation in this part of River Njoro. The slope of the landscape at this stretch is gentle unlike the upstream fast flowing reaches. Consequently, the slow flow of water allows for a longer interaction time between phosphorus loads reaching the stream, stream bed sediments and biota. Nutrient loss can occur due to adsorption to soil particles and their subsequent sedimentation. In addition, the stretch is surrounded by large scale farms that pump water from the river and boreholes thereby reducing the need for frequent visits to the river by livestock and people. This allows for development of vegetation along the riparian corridor consisting of terrestrial and aquatic macrophyte plants (e.g. *Potamageton sp.*, *Papyrus sp. etc*). These plants are known to have a cleaning effect on nutrient rich waters. The plants are useful in constructed wetlands for wastewater treatment. In addition, this stretch of the river has a gentle gradient, which allows for uptake of dissolved nutrients by roots of the aquatic plants suspended in water.

An intact riparian zone can also serve to retain nutrients from the large scale farms surrounding the river therefore significantly reducing the contribution of nutrients from the arable land. Riparian forests adjacent to surface waters have been shown to reduce nutrients from agricultural runoff (Tiessen, 1995).

There are significant positive relationship between phosphorus loss and proportion of land cover area of small scale mixed agriculture (R^2 = 0.5602) and bare land (R^2 = 0.5464) in the River Njoro watershed (Figure 2). This indicates that the major source of phosphorus contamination of stream water is from non-point sources of small scale agricultural and bare lands. On average the two sources contribute over 55% of phosphorus load to the upper and mid reaches of River Njoro. Grassland cover had a negative relationship (R^2 = 0.4171) with phosphorus loss (R^2 = 0.4171). This shows that the more the grass cover, the lower the phosphorus loss from the subwatershed.

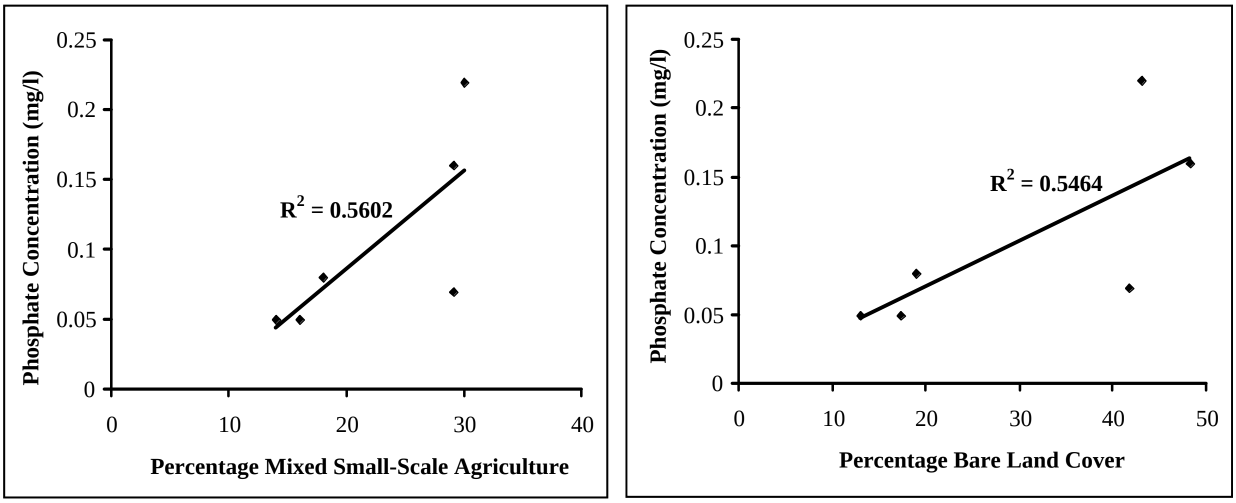

Figure 3: Relationship between phosphate concentration and percentage proportion of land cover areas of small-scale mixed agriculture and bare land.

Results from the study show that the sizes of subwatersheds account for about 53% variability in soluble phosphorus in the River Njoro (Figure 3). This implies that the bigger the size of the subwatershed, the more the contribution of phosphorus to the river water.

Figure 3: The relationship between the levels of phosphates and the size of subwatershed in River Njoro Watershed

Conclusions

The findings in this study generally support research on nutrient and sediment transport within small watersheds with forest or grass buffer areas between disturbed uplands. Rivers serve as integrators of landscape characteristics and as recipients of pollutants from both atmosphere and the landscape. Proportions of different land uses within a watershed can account for some of the variability in river water quality. For example, as the proportions disturbed land covers like mixed small agriculture and bare land cover increases, so does the amount of phosphorus in River Njoro Watershed. Furthermore, the bigger the size of the subwatershed the more the quantity of phosphorus generated from it. This indicates that land-use proportions are important for characterising and modelling water quality in the watershed. Upland land uses are as important as near-stream land uses. The study demonstrates that grassland cover reduces the phosphorus loss from the watershed. Riparian vegetation (forests and macrophytes) influence channel form and stream function by contributing particulate organic matter and large woody debris, by providing shade, bank stability, sinks for organic matter, sediment and by regulating movement and transformation of nutrients.

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References

Baldyga, T.J., S.N. Miller, W.A. Shivoga, and C. Maina-Gichaba, 2004. *Assessing the impact of land cover change in Kenya using remote sensing and hydrologic modeling.* Proceedings of the 2003 American Society for Photogrammetry & Remote Sensing Annual Conference, Denver, CO, May 23-28, 2004.

Bretschko, G., 1995. *Report on the Tropical River Ecology Initiative, First Workshop (31st January – 18th February 1994)*. Egerton University Njoro, Kenya.

Chemilil, 1995. *The effect of huma-induced watershed changes on streamflows*. Doctoral thesis, Loughborough University of Technology. 283pp.

Gregory, S., Swannson, F. J., Mc Kee, W.A. and Cummins, K.W. (1991): An ecosystem perspective of riparian zones. *BioScience* 41:540-5551.

Mathooko, J.M., 2001. Disturbance of a Kenyan Rift valley stream by the daily activities of local people and their livestock. *Hydrobiologia* 458: 131-139.

Karanja, A.K., S.S. China and P.Kundu, 1986. *The influence of land use on Njoro River Catchment between 1975 and 1985.* In proceedings of the Third National Workshop on Soil and Water conservation in Kenya, Nairobi.

Omernik, J.M., A.R. Abernathy, and L.M. Male. 1981. Stream nutrient levels and proximity of agricultural and forest land to streams: Some relationships. *J. Soil Water Conserv*. 36: 227-231

Shivoga, W.A., 2001. Influence of hydrology on the structure of invertebrate communities in two streams flowing into Lake Nakuru, Kenya. *Hydrobiologia* 458:121-130.

Tiessen, H. 1995. P*hosphorus in Global Environment: Transfers, Cycles and Management*. John Wiley &Sons. 462pp.

Siltation studies from valley incision and sedimentation in Roman and present day water reservoirs (Central Ebro basin, Spain)

Silva Crisanto Aguilera

Urbanizacion Villa Aliaza, sector 3 Carrera Comalapa N°4, Puerto Ordaz, Estado Bolivar, Venezuela
Email:crisantosilva@yahoo.com

Abstract

Focusing on catchment-scale erosion, temporal variability within the last centuries, valley infill, gully erosion, sediment accumulation and storage loss in reservoirs is studied. Roman and present-day water reservoir within the Huerva and Aguasvivas River, South of Zaragoza were selected in base of previous erosion rates investigations and the presence of several reservoirs since Romans times until the half of the XX century. The gully site near the Maria de Huerva village was extensively used during Roman times and produced after land abandoned in the beginning of the last century. Identification of sediment storage was estimated using Ground Penetrating Radar (GPR) mapping and photos taken directly form the exposed gully different values of sediment yield from Mid-Holocene times until the present, were calculated. A methodology was designed based in dating with ^{14}C AMS, GPR sections combined with GIS to quickly and efficiently assess sediment yield in ancient reservoirs and try to relate temporal sediment yield variability to natural background and human impact during Roman times. Results show that sediment yield values calculated at the Muel and Almonacid de La Cuba Roman dams are lower (10 times) than those provided by present day reservoirs (Las Torcas y Moneva). Differences in sediment yield will be related to important change in climate or land use change. Ancient water reservoirs would provide interesting information about climate and land use changes in the selected catchments.

Key words: Siltation; Valley insicion; Roman reservoir; Present day reservoir; Ebro basin; Spain

Introduction

Water reservoirs represent sedimentologically closed to semi-closed denudation-accumulation systems (Einsele & Hinderer, 1997). The procedures that are applied to determine the sediment yield in a catchment area considered the sediment that had been deposited in a reservoir in a period of time. Part of the total reservoir storage capacity in Spain (56 km^3) is lost due to siltation processes taking place. Surveys carried out for the Hydrographic Studies Centre of Spain (Avendaño-Salas et al, 1998) indicate that 6% of them have undergone a capacity loss below 20%. In Spain more than 72 dams Romans dams still preserved.

Important Roman dams like Cornalvo and Prosperina still in operation nowadays and most dam have been used like diversion dams. Only the Prosperina located at Extremadura dated I century a.C. (Arenilla Parra, 2003) and Almonacid de La Cuba in Aragon dated as well I century A.D. had been dated with absolute methods. The rest had been dated take in consideration architectonical and archeological criteria. The dam of Vinarragell is considered the oldest dam of Spain which is assigned to pre-Roman times. Nevertheless in Spain dams start to appear intensively during the Roman period from the beginning of the II century b.C. (Diéz-Cascon and Bueno, 2003). The Esparraplejo dam is considered like the oldest dam of the Roman Empire in Spain (Diéz-Cascon and Bueno, 2003). At least five of those dams are in the South of Zaragoza within the Huerva and Aguasvivas Rivers (Figure 1 and 2).

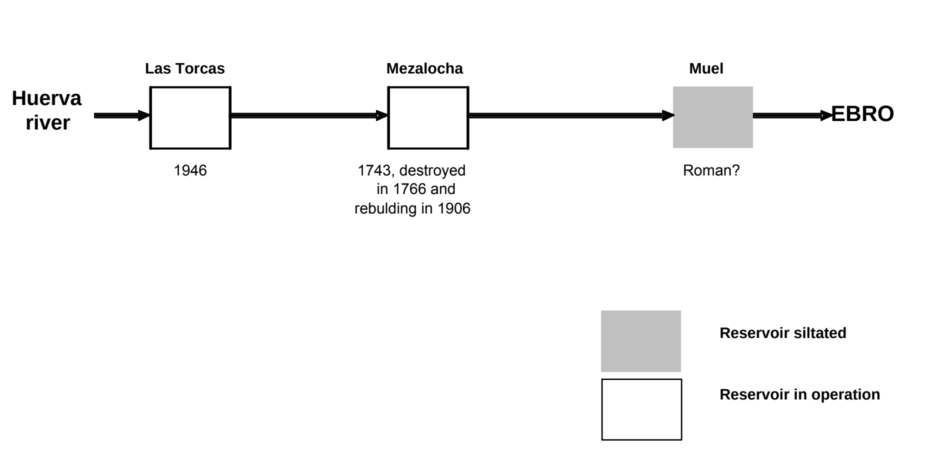


Figure 1. Distribution of reservoirs along the Huerva catchment.

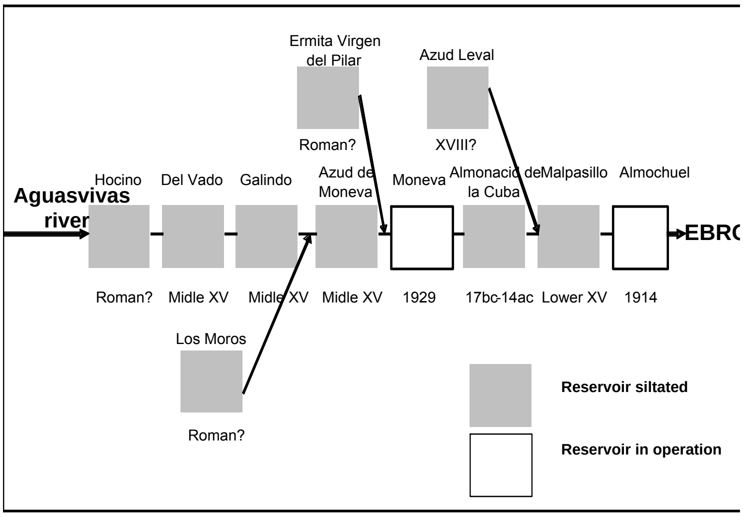


Figure 2. Distribution of reservoirs along the Aguasvivas catchment.

In both catchments is possible to observe the presence of several reservoirs building in both since Romans times until the half of the XX century. Siltation in Roman reservoirs was used for estimate accumulation rate and erosion rate during their life times (Silva *et al.,* 2004). A methodology to quickly and efficiently assess sediment yield in Ancient reservoirs and try to relate temporal sediment yield variability key to natural background and human impact during Roman times had been designed (Hinderer *et al.,* 2004). The area of study (Figure 3), is characterized by an intensive occupation by different groups. In 2000 years the area had been occupied by Iberics, Celtics, Romans, Goths, Visigoths, Muslim and finally conquest by the Catholics Kings in the 1118 by Alfonso the Fighter. The main city in the region during Roman times Caesaraugusta (Zaragoza) was founded the year 24 b.C. and important change in the planning of the land in the middle Ebro occurred during the August period (Suetonio, 121). The area has been affected like several Aragon villages for the migration to the big cities producing dropping of the population.

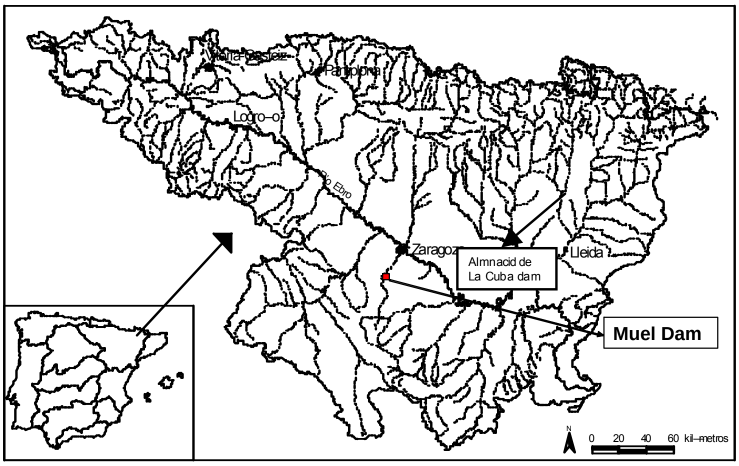


Figure 3: Ebro basin shown the location of the studied sites.

Variables involving in the sediment yield calculation are volume of sediments, area of the reservoir, cathment area, a period of time until the dam siltation and the density of the sediments. Sediment volume and area of the reservoir is easy to get when a bathymetric map of the reservoir is provided. In other case a bathymetric and dating study is required. On Table 1 the relation with the variables involving in the sediment yield calculation and the methods apply for assess is showed.

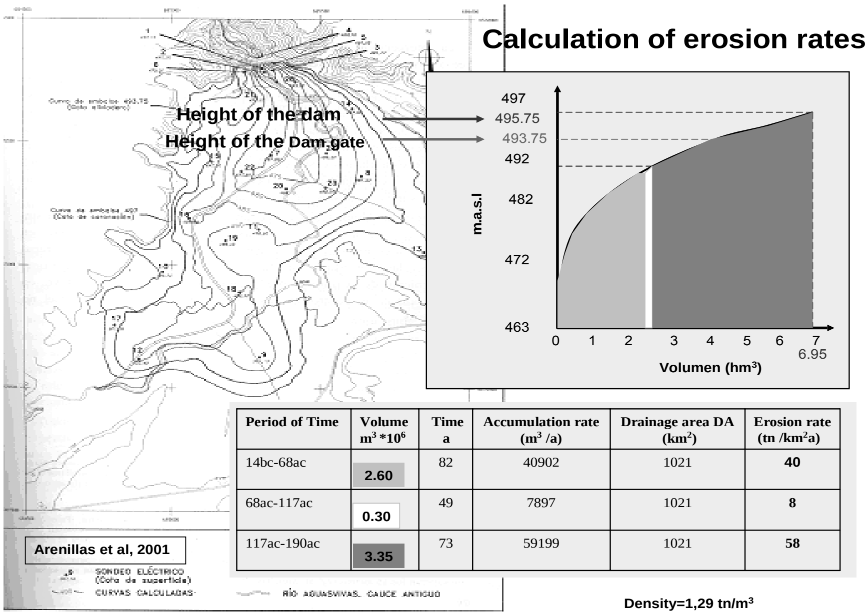
Table 1. Relation with the variables involving in the sediment yield calculation and the methods apply for assess.

Variable	GPR	^{14}C	Mag	GIS	Logging	Historical & Archeological compilation
Volume						
Are						
Dating						
Density						

The almonacid de la Cuba Roman Dam

The Almonacid de la Cuba dam is located in the Aguasvivas River (Figure 2). As one of the world biggest dam in the world before the construction of the Kurit dam building in Iran in the XIV century (Arenillas Parra, 2001). The dam is complete silted.

A bathymetric reconstruction was elaborated using VES, sounding and dating in sediments and dam wall (Arenillas Parra, 2001). Volume of sediments deposited in the reservoir during Roman times was calculated from the characteristic curve of the reservoir which relates the elevation of the reservoir area (m) and the volume (hm^3) (Figure 4).



Period of Time	Volume $m^3 * 10^6$	Time a	Accumulation rate (m^3/a)	Drainage area DA (km^2)	Erosion rate (tn /km^2a)
14bc-68ac	2.60	82	40902	1021	**40**
68ac-117ac	0.30	49	7897	1021	**8**
117ac-190ac	3.35	73	59199	1021	**58**

Fig 4: Sediment yield calculation for the Almonacid de la Cuba roman dam.

Dating information from previous work in Almonacid de la Cuba (Arenillas Parra *et al.,* 2001) estimated in base of ^{14}C dating in sediments and walls of the dam that it was building during the age of the Roman imperator August, the capacity was increased during the Neron epochs, and a complete siltation would be occurred during the II Century during the epochs of Trajanus, from them the dam was used by the Muslim and during the Middle age like a derivation water channel. Based in this results sediment yield form those different periods were calculated and are showed on Table 2 (Silva *et al,* 2004). Archeomagnetic methods were applied in the exposed sediment sequence of the reservoir by cutting of the actual river pathway (Pueyo *et al.,* 2001). Unfortunately they obtain a date of 2000 B.P. in the base of the gravel bed. This not reliable result will be due to contaminated samples (Emilio Pueyo and Blas Balero, 2004 personal communication).

The Muel Roman dam

The Muel dam is located in the Muel village over the Huerva River, a tributary of the Ebro River (NE Spain) (Figure 1). The dam wall of 3m high and 60m long is very well preserved, and the reservoir is completely silted (Fatas, 1964).

A combination of geophysical, sedimentological techniques, ^{14}C AMS dating and historical compilation allowed the estimation of sediment yield during the life of the dam. Ground Penetrating Radar was applied in order to obtain the thickness of the sediments accumulated in the reservoir, and to decide the cores location. Sections were carried out using a 25 MHz antenna. Penetrating depths was up to 40m and allow the identification of the infill materials (Figure 5). A drill campaign was carried out close to the dam site and cores were recovered (Garcia Vera, 2004). The bottom of the dam infill provided an age of 2020±30 ^{14}C yr B.P indicating that the dam was constructed during the I century b.C. corresponding with the period of Augustus. A second sample located 4,25m over the first one, showed an age of 1880±30 ^{14}C yr B.P.

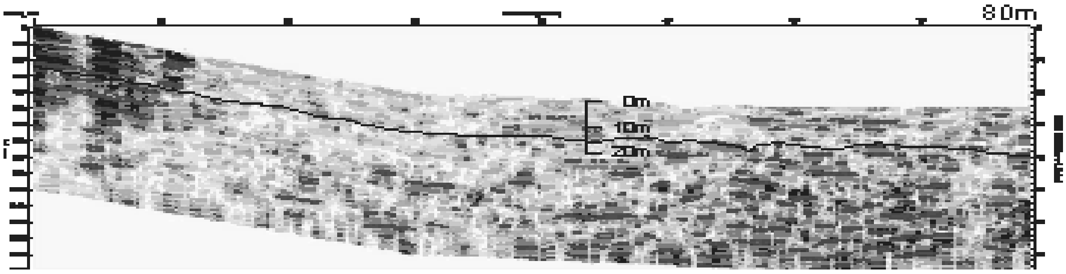


Figure 5: GPR sections were correlated with the core (M-2) located over the lines.

Assuming a constant rate of sedimentation of 1.9 mm/year the dam was complete silted during the VI century A.C. Oral traditions suggest that the Mosque was build over the silted dam under the orders of the moor king Muza in the year 869A.D. In 1770, a Catholic Hermitage was built over the Mosque ruins. Sedimentation of the Muel Dam is coherent with other historical data from Roman Reservoirs in Spain. Based in this results sediment yield form this period was calculated and is showed on Table 2.

Table 2: Accumulation and sediment yield calculation from the Muel Roman Dam.

Period Time	**Mass (Tn)**	**Years (a)**	**Accu. rate (Tn/a)**	**Drainage area (km^2)**	**SS yield ($vol/km^2/a$)**
15aC-572 AD	$4.3*10^6$	587	7197.8	461	16

Discussions

Sediment yield values calculated at the Muel dam seems to be lower (up to 5 times) that those calculated in the Almonacid de La Cuba dam. At the same time seem to be lower (10 times) sediment yield values provided (Avendaño-Salas *et al.,* 1998) from the present day reservoir in Las Torcas and Moneva (Figure 6). Differences in sediment yield will be related to important change in land use or change in climate conditions.

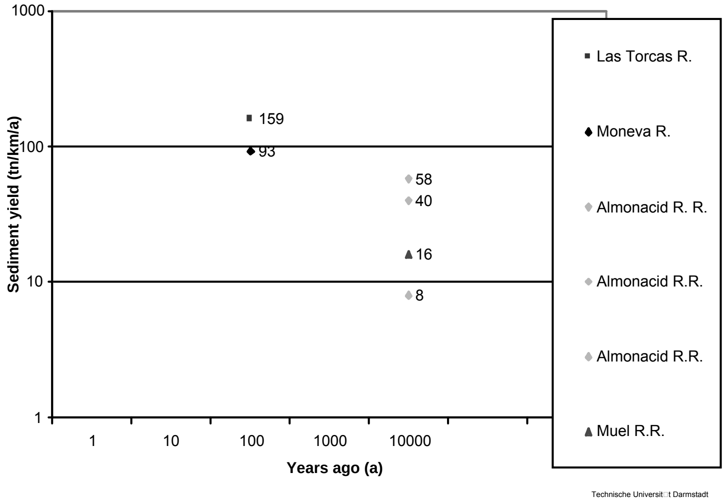


Figure 6. Comparison of Sediment Yield Vs Time in present ay and Roman reservoir in Aguasvivas and Huerva basin.

Information about climate and land use in Roman time would be related with the results of the sediment yield form Roman times. Is difficult to identify to which process climate or human activities influence erosion-sedimentation in past environment. Important studies in the area taking in account the climate and land use change had been carried out in the area of study (Gutierrez-Elorza, and Peña-Monné, 1998). From there the period of the Roman occupation coincided with a warm and dried period and a increasing of the valley and slope incisions. Besides the information in sediment yield obtained from Roman times, siltation in ancients water reservoirs will provided interesting information about climate and land use changes in the selected catchments.

The political, cultural and economic rise of Roman Civilization was paralleled by increased and unregulated deforestation to acquire raw material and land exploitation for farming (Brückner, 1996 and 1992). Equally damaging to soil erodibility was uncontrolled grazing of animals (Hughes and Thirgood, 1980).

Sediment yield values are usually used for water reservoir assessment and like catchment scale process indicator. A major obstacle to assess future impacts on climate and humans activities s the lack of suitable timescales over which long-term processes and system properties may be observed (Dearing and Jones, 2003).

Sediment yield had been calculated from recent reservoirs since the middle XX century and providing a maximum range of 100 years (Figure 7). The use of sediment yield values from ancient dams would provide a rank of at least 2000 years; this fact would be used to understand past hydrological variability (Oldfield, 2003) and improve the assessment of actual and future water reservoirs in the selected catchments.

Archaeomagnetic methods would provide important catchments environmental information about land use and climate change like periods of drought or flooded events in the cores and additionally a "continuing" dating along cores. Magnetic susceptibility of anthropogenic sediments and soils could be viewed as a proxy for past climates (Tite and Liningtond, 1975). Sediment cores from the reservoir were subject to archaeomagnetic methods in cooperation with Emilio Pueyo from the Geological and Mining Institute of Spain, and Prof Dr. Andrés Pocoví from the Departament of Endogen Geodynamic the University of Zaragoza.

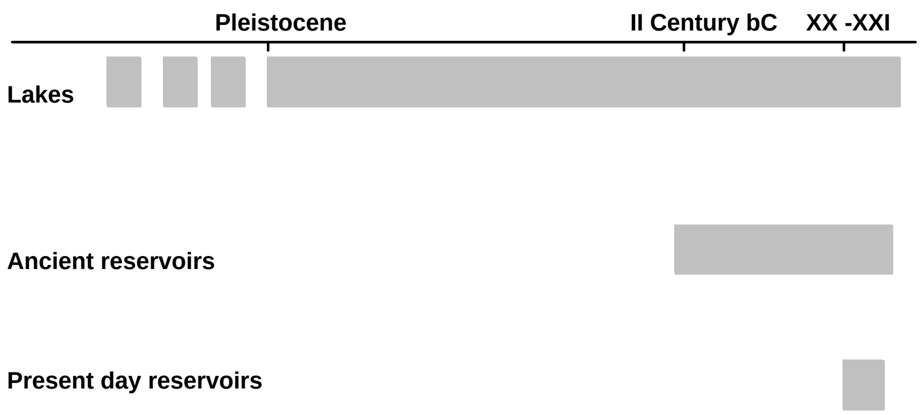


Figure 7. Comparison of temporal time life of some water related environment.

Testing is being carried out actually and we are optimistic with the preliminary results. This method would provide important catchments environmental information about land use and climate change like periods of drought or flooded events in the cores and additionally a "continuing" dating along cores.

References

Arenillas Parra, M., 2003. Dams from the Roman era in Spain. Analysis of design forms. http://traianus.rediris.es/

Arenillas Parra, M., Daz-Guerra, C., Cortes Gimeno, R., Beltrán Llorís, J.M.,Viladés Castillo, J.A., Sesma Muñoz, J., Utrilla Utrilla, F., Laliena Corbera C., 2001. *La presa romana de Almonacid de la Cuba. Ministerio del ambiente,.* Confederacion hidrologica del Ebro. Zaragoza, 316p.

Avendaño-Salas, C., Sanz Montero, Cobo-Rayán, R., 1998. *State of the art of reservoir sedimentation management in Spain*. International Workshop and Symposium on reservoir sedimentation management.

Brückner, H., 1986. Man´s impact on the evolution of the physical environment in the Mediterranean Region in Historical times. *GeoJournal* 13(1), 7-17pp.

Brückner, H., 1992. Human-Induced erosion process in Mediterranean Countries evidences from Archeology, Pedology and Geology. *GeoOekoplus,* III: 97-110pp.

Castillo Barranco, J.C., Arenillas Parra, M., 2000. *Las presas romanas en españa propuesta de inventario*. I Congreso Nacional de Historia de las Presas, Mérida, Spain.

Dearing, J., Jones, R., 2003. *Coupling temporal and spatial dimensions of global sediment flux through lake and marine sediment records*. Global and Planetary Change 39, 147-168pp.

Díez-Cascon Sagrado J., Bueno Hernández F.(2003). *Las presas y embalses en España, Historia de una necesidad.Ministerio de Medio Ambiente*. Dirección Gral. de Obras Hidráulicas y Calidad de las Aguas, 2003. 445 p.

Einsele, G. & Hinderer, M., 1997. Terrestrial sediment and the lifetimes of reservoirs, lakes, and larger basins. *Geol. Rundsch.* 86: 288-310.

Fatas, G., 196). *Notas sobre el dique romano de Muel*. Caesaraugusta 21-22, 174-177pp.

Garcia Vera, M.A., 2004. *Determinación de la tasa de erosion del tramo bajo del río Huerva: Recopilacion de informacion básica.* Thecnical report. Umpl. Confederación Hidrográfica del Ebro. Zaragoza, Spain.

Gutierrez-Elorza, M., Peña-Monné, J.L., 1998. *Geomorphology and late Holocene Climatic change in Northeastern Spain*. Elsevier, Geomorphologhy 23, 205-217pp.

Hinderer, M., Silva C., & Ries, J., 2004. *Erosion in zentralen Ebrobecken un Sedeiment-akkumulation in Talsperren. GeoLeipzig 2004, Geowissenschaften sichern Zukunft.* Schriftenreihe der Deutschen Geologischen Gesellschaft, Heft 34, p. 124.

Hughes, J.D., and J.V. Thirgood, 1982. *Deforestation in Ancient Greece & Rome collapse*. The Journal of forest History, 26(2), 196-208pp

Oldfield F., 2003. *Some personal perspectives on the role of Quaternary science in global change studies.* Global and Planetary Change 40, 3-9pp.

Pueyo,E., Mauritsch, H., Pocoví, A., Scholger, R., Millán, H., Valero, B., 2001. *Archaeomagnetostratigraphy of the filling of a Roman dam*. Preliminary report on Almonacid de la Cuba (NE Spain). AARCH-II Archaeomagnetism Applied to the Rescue of Cultural Heritages.

Suetonio (121) De vita Duodecim Caesarum. Translation of Alfonso Cuatrocasas, 2001. *La vida de los doce Césares*. Espasa, Spain. 508p.

Silva, C., Hinderer, M., Ries, J., 2004. *Catchments-scale erosion and sediment accumulation of reservoirs in the Ebro basin, a preliminary review.International Conference on River/Catchment Dynamics: Natural Processes and Human Impacts*. Abstracts book. Solsona (Catalonia, Spain) 15-20 May, 2004

Tite, M.S., Linington, R.E. (1975). *Efects of climate on the magnetic sussesptivility of soils*. Nature 256, 565-556.

Degradation of the riparian wetlands in the Lake Victoria basin - Yala swamp case study

Thuita Thenya[1], Reiner Wassmann[2,3], Louis Verchot[3], David Mungai[1]

[1]Department of Geography & Environmental studies, University of Nairobi, P. O. Box 30197-00100, Nairobi, Kenya
[2]Centre for Development Research (ZEF), University of Bonn, Walter Flex Str. 3, D-53113, Germany
[c]World Agroforestry centre (ICRAF), P. O. Box 30677-00100, Nairobi, Kenya
[3]Institute for Meteorology and Climate Research, Kreuzeckbahnstraße, 19, D-82467 Garmisch-Partenkirchen, Germany
Email :tthenya@yahoo.co.uk

Abstract

Land degradation is as a result of broad range of scales and factors, which include biophysical, climatic, demographic and socio-economic. The aim of this paper was to provide an analysis of wetland utilisation, ecosystem degradation and their effect on the Lake Victoria ecosystem. This involved analysis of socio-economic and remote sensed data. The main sources of wetland degradation in the Lake Victoria basin were identified as (i) farming activities, (ii) grazing and macrophyte harvesting and (iii) coupled with catchment degradation-deforestation. These factors were closely related to the demographic dynamics and unsustainable land utilisation practices. Socio-economic data provided valuable insight on the pattern of wetland utilisation and possible sources of degradation pressure. For example, there is high dependence of the local indigenous livelihood directly on the swamp for subsistence needs including farming, grazing and income generation. Farming is the most important wetland utilisation activity, which takes 95% of the households wetland land holding mainly for subsistence use. In addition, there has been progressive degradation of the catchment area through deforestation, overgrazing and low furrow period. This results in high sediments transport and other pollutants to the lake ecosystem due to the removal of buffering effect of the macrophytes in the swamp especially along river Nzoia systems. Remote sensing data indicated progressive opening of the swamp especially in the high population and more accessible northern side of the swamp. In conclusion, the unsustainable use of natural resources in the basin has had significant negative effect on the Lake ecosystem including water pollution siltation and increase of floating biomass.

Key words: Degradation, macrophyte, remote sensing, utilisation, wetland,

Introduction

Land degradation is a process which implies a reduction of potential productivity of the land (Hill *et al.,* 1995b). It is also considered to be a collective degradation of different components of the land such as water, biotic and soil resources (Hennemann, 2001). In broader sense land degradation is the alteration of all aspects of the biophysical environment by human actions to the detriment of vegetation, soils, landforms, water and ecosystems. According to Lal and Stewart (1985), soil degradation undermines the productive capacity of the ecosystem resulting for example in alteration of water status. Vegetation cover especially riverine vegetation act as natural filter of sediments minimising silt deposition in water bodies. It is for this reason that wetlands are valued for their ecological services like shoreline protection (Daily, 1997) and support to biodiversity among others (Barbier, 1993). In many countries especially in the developing world, local economies depend on wetland utilization such as reed harvesting, grazing, and farming especially in the dry regions (Adelaida, 2000). These activities often lead to change in wetland vegetation cover.

The increasing widespread lack of sustainable use of wetlands especially in the developing countries can be attributed to the lack of recognition of the traditional values of these wetlands, desire for modernisation (Panayotou, 1994; Maclean *et al.,* 2003) and failure of appreciation of ecological role. The most important sources of wetland degradation emanate from the drainage for agriculture, settlement and excessive exploitation by the local people. In addition, poor ecosystem management under increasing population generate high degradation pressure. Quite often there are numerous underlying factors to wetland degradation such as population increase (Ruthenberg, 1976; Binswanger and McIntire, 1987; Pingali *et al*., 1987). Other factors include international agreements, such as IMF, World Bank trade liberalization and related national policies that affects farmers' decisions on land use. Extreme weather events like drought and floods are also important (Turner *et al*., 1989). However, recent studies in East Africa on land use change have pointed out agriculture is the main cause of wetland degradation (Thenya, 2001; Mugisha, 2002; Torrion, 2002; Githaiga *et al.,* 2003; Reid *et al.,* 2004). However, wetland conversion to farming and settlements is not a new phenomenon. It is estimated that more than half of the wetlands in the world may have disappeared since the start of the 20th century, with most losses being in the developed countries (Barbier, 1993). However, wetland loss is on the rise in the developing countries as a result of conversions mainly for agricultural lands and urban settlements (Maltby, 1986; Farber and Constanza, 1987; Maltby, 1988; Mohamed, 1998; Torrion, 2002). Nonetheless dramatic losses have occurred over short space of time (Maltby, 1988).

The main objective of this paper was to provide a synthesis of wetland utilisation, ecosystem degradation and the effect on Lake Victoria aquatic ecosystem using the Yala swamp, a large swamp area located on the north-eastern edge of Lake Victoria, Kenya.

Ecosystem degradation and impact on aquatic ecosystem

The wetland degradation forces are mainly a combination of the demographic factors like population density, settlement, food deficit and institutional support failure (Hollis, 1990). Some of these wetland changes are within planned development scheme, like draining for rice farming on the upper part of the Senegal River (Halls, 1997; Drijver and Van Wetten, 2000). However, others are as a result of uncoordinated – and often illegal – activities of the local people (Panayotou, 1994). Demographic changes coupled with poor environmental policy have in various cases aggravated the degradation of the wetland resources. For example, unsustainable wetland utilization in Ethiopia has been attributed to the combined changes in policy, political regimes and demographic dynamics (Thompson, 1976; Wood, 1997; Wood *et al.,* 1998).

The Lake Victoria basin (LVB) faces far more complex social, economic, political and technical barriers than other transboundary lakes in the region (Duda, 2002). The environmental degradation of LVB over the last 3 decades, due to unsustainable use of natural resources has had significant effect on the Lake ecosystem. These include massive algal blooms, waterborne diseases, water hyacinth infestation and oxygen depletion (Odada *et al*., 2004). Changes in land-use pattern have lead fundamentally to spatial and temporal heterogeneity of the limnological characteristics thus influencing ecological structure and functioning of the aquatic ecosystems (Nogueira *et al*., 1999). Accurate estimation of the runoff can give an indication of land use changes and destruction of soil cover. Lake ecosystems are often downstream ecosystems that are affected by changes both in the catchment areas and fringing wetland areas. The latter are supposed to act as buffer to sediments and pollutants transport to the lake ecosystem. Studies on the adverse effects of cattle grazing in forests in the catchments areas of Rivers Yala and Nzoia have shown the infiltration capacity has greatly reduced in hardwood and softwood forests from 190 to 1.3 and 280 to 33 mm/hour respectively (JICA 1980 and 1991). The presence of forests and soil cover reduces the likelihood of serious soil erosion as the root system physically binds the soil while the canopy and ground litter physically protect the top soil by intercepting potentially destructive heavy rainfall and reduce the resulting runoff. Interception by the overhead canopy spreads the flow of water to the ground over longer period of time and allows more time for infiltration into the soil, which recharges ground water. The effect of extensive destruction of the soil cover including forest and reduced fallow period hamper infiltration and encourage high rivers discharge. There has been a general increase in river flow discharge over the years as well as an increase in rainfall run-off possibly resulting from the degradation of the catchment. The average river discharge both in the Yala and Nzoia rivers have increased over the years, for example their discharge in the period 1990-2000 was higher than for the period 1950/60 (Sangale *et al.,* 2001).

The different sub-catchment in the lake basin contributes varying amounts of sediments to the Lake Victoria depending on the local settings. For example, although the total land cover under cultivation in the basin in Kenya and Tanzania is comparatively smaller, the two countries contribute relatively higher sediments to the Lake Victoria (Table 1). This is due to their close location of the catchment relative to Lake Victoria and the step topography (Table 1). In contrast, Uganda has smaller sediments contribution through the river system since it is at the out flowing direction. However, in spite of the distant location of the Rwanda and Burundi catchments, the high percentage of land under cultivation contributes relatively high percentage of sediments and other pollutants into the lake ecosystem.

Table 1: Agricultural characteristics of the Lake Victoria basin (catchment land area (1000 ha) (Source: Odada *et al*., 2002).

Countries in the basin	Cultivated	Non-cultivated	Total	Percentage land under Cultivation
Kenya	1470	3400	4870	30.18
Uganda	1400	2100	3500	40.00
Tanzania	1500	5540	7040	21.31
Rwanda	930	1130	2060	45.15
Burundi	670	640	1310	51.15
Total	5970	12810	18780	31.79

According to Kaufman *et al*., (1997) anthropogenic activities have contributed to the changes in the limnology of the Lake Victoria. Records indicate that Lake Victoria was stable during the pre-colonial period (Hecky, 1993). Changes in the limnology of the lake started surfacing in the 1900 a period coinciding with cultivation, industrialization and rapid population growth whereby sewage and other domestic wastes increased. By 1920's nitrogen and phosphorous levels had started increasing due to anthropogenic inputs and blue green algae started to appear in the lake (Seehausen, 1996). According to Machiwa, (2001) rivers Simiyu and Kagera with heavy agricultural activities in the catchment contributed significantly to siltation load in the Lake Victoria. The two rivers' sediment loads were found to be 5674 tonnes/day and 2111 tonnes/day for Simiyu and Kagera respectively. Phosphorous levels indicated that river Simiyu had the highest phosphorous load of 32.99 tonnes/day (Total Phosphorous) and 0.4486 tonnes/day soluble reactive phosphorous. While River Kagera had values of 1.6475 tonnes/day (Total phosphorous) and 0.1515 tonnes/day soluble reactive phosphorous.

A major driving force towards wetlands degradation in Kenya is the expansion of farming communities and settlement schemes, especially in the Arid and Semi-Arid Lands (ASALs) (Mohamed, 1998; Thenya, 2001). In the Lake Victoria Basin (LVB), the use of wetlands for subsistence farming and to raise income is relatively high (Johnson *et al.,* 2000; Mugo and Shikuku, 2000; Gichuki *et al.,* 2001; Kairu, 2001; Thenya *et al.,* Vol. I *In prep*). This acts as a significant source of the swamp conversion pressure and subsequent degradation. In addition, the high population density in the LVB and around the Yala swamp exerts both direct and indirect pressures on the swamps in the basin. Lake Victoria has one of the high population densities globally which exerts high pressure on land utilisation (ICRAF, 2000). However, the role of population depends on specific settings in a given region. These settings include population density (Pender, 1998), land availability (Jones, 1988), type of land use (Schelhas, 1996), among others. Recently some 10,000 ha in the Yala swamp were leased out by the government to a private company for large-scale farming reducing the area available for cultivation and grazing by the indigenous community. This arrangement is bound to increase the degradation pressure especially in the absence of wetland management plan. The overall effects of these wetlands degradation by the farming community are reflected in reduced buffering effect of the wetland and subsequent siltation of the Lake Victoria.

Study area

Biophysical and hydrological dynamics conditions

Several of the rivers approaching the Lake Victoria form extensive wetlands along the lakeshore. These wetlands are made up of mixed grass and papyrus swamps with scattered stand of arborescent swamp forest. Vegetation cover dynamics in these wetlands responds significantly to changes in flooding regime. In the case of the Yala swamp, flooding is triggered by high rainfall in the upper catchment areas leading to high discharge of the Yala and Nzoia rivers. Annual rainfall in LVB encompasses a bimodal pattern with 'long rains' from March to April and 'short rains' from Oct. to Nov. The Yala/ Nzoia catchment has high precipitation in the Northern highland (1,800-2,000 mm per annum) and low in the South-Western lowlands (800-1,600 mm per annum). With the average rainfall around lowland Yala swamp being approximately 760mm.

Yala swamp is located in Lake Victoria Basin (LVB). The swamp is a deltaic wetland dominated by species of the genus Cyperus among them *C. papyrus, C. dives, C. exaltatus* and *C. distans*. It is located between rivers Yala and Nzoia, 0^0 07' N – 0^0 01' S / 33^0 58' – 34^0 15' E (Figure 1). Its formation was as a result of backflow of water from Lake Victoria as well as flooding of the Rivers Nzoia and Yala. River Nzoia has the second highest discharge at 115 m^3/s after River Kagera with 260 m^3/s. The swamp is mainly fed by river Yala that flows right through the swamp with a small contribution from river Nzoia in the North-eastern section of the swamp. The wetland comprises 30,000 ha; with the distances between the opposite edges extending for 15-25 km (Hughes and Hughes, 1992; M'mayi *et al*., 1997). This ecosystem also encompasses three lakes these are Lakes Sare, Namboyo and Kanyaboli (Figure 1). These lakes contain some endemic haplochromine fish species, some of which are no longer found in Lake Victoria and are in acute danger of extinction (Kaufman and Ochumba, 1993). Likewise, the wetland provides vital habitats for various species of birds, mammals, reptiles and fish (Mavuti, 1989; Hughes and Hughes, 1992).

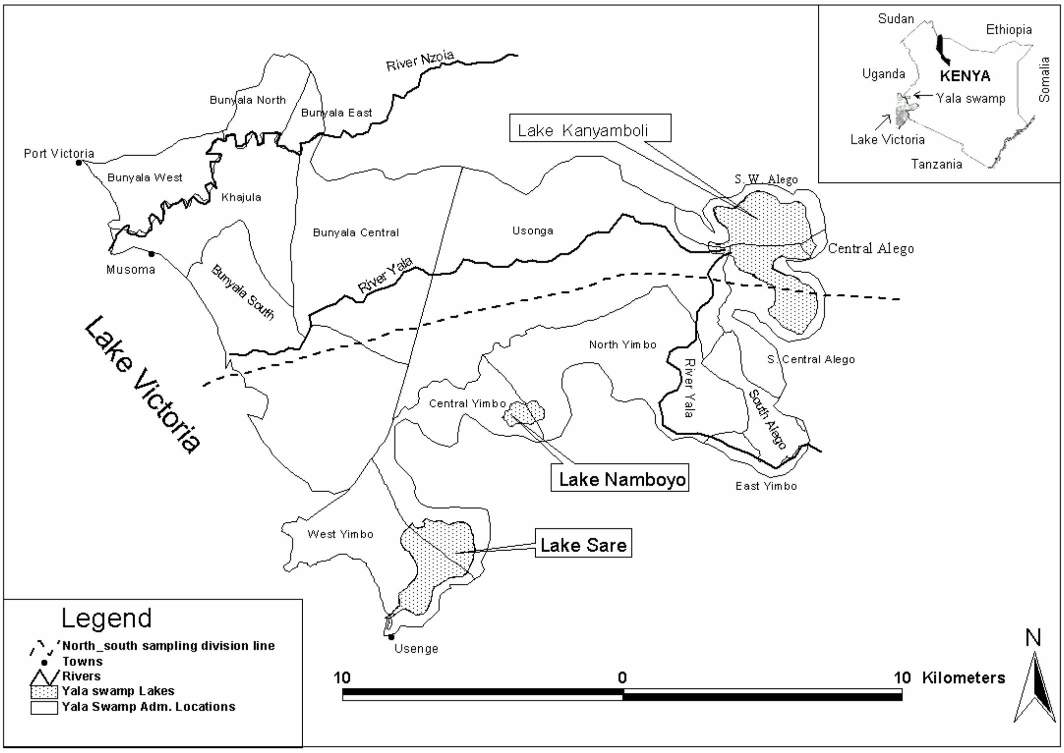


Figure 1. Yala swamp, associated lakes ecosystems and administrative Locations.

Socio-economic status

The Lake Victoria basin comprises one of the densest rural population globally with estimated 1,200 persons per km^2 (Hoekstra and Corbett, 1995). The population is characterised by high out migration to urban areas in search of jobs (GoK, 1994). This is attributed to the fact that the household are generally poor with livelihood mainly derived from subsistence farming and indigenous livestock. The local communities are traditionally dependent on the local wetlands for their livelihood including vegetable collection and grazing among others (Abila, 1998; Gichuki *et al.*, 2001). There is also a strong reliance on the macrophytes both for building, fishing gears, beehives as well as a source of income in this area (Odak, 1987; Ogutu 1987; Kareri 1992 Hoekstra and Corbett, 1995; ICRAF, 2000). Farming in the swamp in particular is an important activity (Mugo and Shikuku, 2000) and takes place seasonally in form of semi-permanent cultivation. The declining soil fertility around the swamp coupled with high population acts partly as driving force to wetland conversion. However, pervasive poverty hinders sustainable use of the land resources leading to considerable land degradation, sedimentation and nutrient run-off, which contributes to the swamp degradation. The swamp has also undergone considerable changes through coordinated government initiated farming activities. Until the mid 1960s the Yala swamp covered a total of 17,500 ha as natural swamp. Between 1965 and 1970, 2,300 ha were reclaimed for farming by the Ministry of Agriculture (MOA) (Aloo, 2003). Although, the Lake Basin Development Authority (LBDA) had earmarked two other Phases for reclamation, with Phase II constituting 6,000 ha and a Phase III 9,200 ha (Mavuti 1989; Mavuti 1992; Ocholla, 1997). Recently, in year 2002, some 10,000 ha, including a large section of phase 1 LBDA 2,300 ha drained section was leased out for rice farming to a private company threatening the indigenous use. Communities inhabiting both the catchment areas and the area around Yala swamp are highly dependent on natural resources mainly farming and grazing.

Materials and methods

This involved two approaches, (i) analysis of satellite images and (ii) socio-economic data on wetland utilisation.

Satellite imageries

A combination of Landsat MSS and Landsat ETM satellite images were used to assess land cover dynamics in the Yala swamp over duration of 28 years. The details of these two images were as follows;

Table 2. Details of the MSS and ETM used in the study

Landsat Images	Date acquired	Spatial resolution - m	Path	Row
MSS	5th Feb. 1973	57	183	060
ETM	2nd Feb. 2001	28.5	170	060

The two images were already orthorectified to the Universal Transverse Mercator (UTM 36M) projection, with WGS84 datum. Since both of these images were taken during the dry season, spectral changes due to the phenological differences of the vegetation were expected to be lower than changes due to the land cover transformation.

The choice of the two images was based more on the availability of the same season images from the Global Land Cover Facility (GLCF) (U. S. Geological Survey, 1973 and 2001) to allow temporal comparison. In addition, the absence of clouds was also an important criteria in the selection of these two images. The problem of availability of cloud-free images in the tropical region is very common and has been recorded by several scientists (Jha and Unni, 1982; Ducros-Gambart and Gastellu-Etchegorry, 1984; Pilon *et al*., 1988 and Alwashe and Bokhari, 1993).

Remote sensing data processing

Comparative land cover changes in the Yala swamp were derived from two satellite images that is the 1973 (5th Feb.; Multi-spectral scanner-MSS) and 2001 (2nd Feb.; enhanced thematic mapper-ETM), which were obtained from U. S. Geological Survey, (1973 and 2001). Initially the two images were subjected to relative radiometric corrections and registered to each other. Thereafter to facilitate comparison between the two images, the year 2001 image with pixel size of 28.5 m was degraded to 1973 image pixel scale of 57m in ERDAS using the nearest neighbour resampling technique (Lillesand and Kiefer, 1987). This approach is frequently used to facilitate comparison of satellite images with different spatial resolution (Turner *et al.,* 1989; Benson and Mackenzie, 1995; Wu *et al.,* 2002; Saura, 2003). A hybrid classification procedure was employed in the images analysis. This method is more able to analyse and visualize the class definitions than either type of classification can provide independently (Kloer, 1994). Thereafter, the resulting land cover classes were subjected to post classification analysis. This included accuracy assessment using confusion (error) matrix based on the field observations and land use change matrix analysis to assess land cover transfer.

Socio-economic survey

Primary data was collected at household level between May 2003 and April 2004. The survey covered all the 15 administrative units (Locations, see Figure 1) in the area surrounding the swamp yielding a total of 336 questionnaires. Since the focus was on the communities within close proximity to the swamp, the survey was restricted to a distance of 5km from the swamp boundary. This followed a similar approach used by Abila, (1998) in the Yala swamp. The household data collected included household structure, wetland utilization and seasonal utilisation dynamics. Local enumerators, familiar with the local dialect were trained and used as interviewers to avoid possible misunderstandings or loss of information that are prone to interpreter-mediated surveys. The data collected was analysed using SPSS.

Results

Wetland utilisation

Socio-economic factors in the Yala swamp were very important in explaining the wetland degradation. For example, farming is an important activity, which takes 95% of the swamp land holdings and contributes more than 70% of domestic food requirements. In addition, the declining soil fertility around the Yala swamp and the small land holding size in a high population density area generate a high swamp conversion pressure. Land in the wetland is acquired through self-allocation and later passed on through inheritance along kinship line. This happens since the swamp has no management strategy and is thus prone to over exploitation and degradation including sensitive areas like along the water body fridges.

While the other land uses that is grazing, furrow and sand harvesting had relatively small proportions on both sides of the swamp. There has been a gradual rise in the wetland farming with population increase over time, which dates back to the 1920 on both sides of the swamp. The reasons for starting to farm in the wetland were quite varied but almost the same on both sides of the swamp. The most important reason was to supply subsistence food. Other important reasons attracting people to the wetlands included water availability, high crop yield, favourable farming climate. However, the need to raise subsistence income through cultivating in the wetland was low, constituting only 30%. This emphasis the point that farming was mainly geared towards the supply of subsistence domestic food, with the need to raise income being a secondary activity. Most of this farming took place along the edge of the swamp and gradually edged inwards. According to the survey, the most prioritised future development by the local community was draining the swamp for farming with fishing and conservation taking a low priority. However, this preferred option overlooks other numerous benefits from the swamp like grazing, thatch material, fishing as well as ecosystem services like filter functions, carbon sequestration and species conservation.

Yala swamp provides numerous products to the local indigenous communities. These include thatch and craft material from the macrophytes, brick making soil, sand and domestic fuel. Others are water for the animals and domestic use, vegetables, fish, forage and grazing grounds. Note however that wetland utilisation is mainly for subsistence with minimal commercialisation, which is mainly geared towards domestic food supply, shelter and subsistence income. Macrophyte provides an important source of thatch material for the local indigenous communities where most of the houses were semi-permanent on both sides of the swamp. On average, approximately 50% of the houses had thatched roofs with 70% of the houses having earth walls. Analysis of some determinant of roofing material on both sides of the swamp indicated a strong relationship between the type of roofing and the education level of both the husband $x^2_{(20)}$ 118.84 > 37.57 at p= 0.01 and the wife $x^2_{(20)}$ 118.84 > 37.57 at p= 0.01. This implies that the prevalence of low education level in the area has a link to the presence of high percentage of semi-permanent houses. Low education level also means minimal alternative sources of income hence, the community members are unlikely to devote the small income gained to shelter improvement relative to other consideration such as education and clothing among others. This emphasizes the high dependences on the wetland resources with minimal alternative.

Livelihood support and income sources

Major sources of income and livelihood are restricted to four major activities in the Yala swamp. These include farming (70%), business (10%), fishing (6%) and local employment (4%) in that order. The relatively favourable land gradient in the northern side encourages more farming leading to more swamp degradation. In addition, in the northern side of the swamp, brick making is an important activity. Although fishing is relatively low on this side since the three major satellite lakes are located in the southern side. Minor sources of income (4%) included hunting and remittances from relatives which are shown under auxiliary. Business and fishing increases during the dry season to cater for the shortfall in farming and low local employment.

Various types of crops are grown in the wetland across the seasons on both sides of the swamps. These include bananas, maize, millet, yams, oil crops (groundnuts, simsim), sugarcane, kales, cowpeas, onions and cassava, which are common in the LVB. Several business undertakings are used to supplement local income especially in the dry season. These include mainly sale of crops including those cultivated in the swamp, trade in thatch material and game meat as well as boat making. Other items traded included fish, water vending, sale of local brews like chang'a, sale of building poles, pots, vegetables, milk, millet and running of kiosks. This was also supplemented by casual jobs like security guards and working in other people's farm for a pay. Comparing wetland income to other sources outside the swamp, more than 52% of the respondents on both sides of the swamp acknowledged that the swamp provides more opportunity. This was strengthened further by the fact that 82% of the households interviewed had farming land in the Yala swamp, which emphasised the importance of wetland opportunities. The education level of the husband was also important in determining the contribution of income from the wetland $X^2_{(10)}$ 25.49 > 23.2 at p = 0.01. Wetland activities are labour intensive and require a large input from men relative to women contribution to realise benefits.

Swamp land cover change

The outcome of satellite image analysis show progressive increase of farming area from 1,564 ha in 1973 to 5,939 ha in 2001 (Figure 2, Table 1). This increase was mainly at the expense of macrophyte community (papyrus-phragmites-typha community-sedges-papyrus), which decreased from 7,180 ha to 4,999 ha (Table 3). This supports the results of socio-economic survey which indicated that farming is an important activity and takes a large share of wetland land holdings.

Table 3. Land cover change between 1973 and 2001.

Land cover	Total - 1973	Total - 2001
Open water	2672.06	2200.94
Silt water	569.32	542.99
Conversions	1564.23	5939.84
Bushes-phoenix	1652.60	2462.60
Sedges-papyrus	7180.20	4999.93
Sedges-latifolia	3959.87	3435.64
Bushes-sesbania	4343.60	2359.96
Total	21941.89	21941.89

The minimal change that was noted in silt spatial coverage was most likely due to the increase of floating vegetation mass. However, the reduction in open water is a clear evidence of this vegetation mass expansion. Other changes include expansion of vegetation communities (bushes-phoenix community), which often appears as coloniser both in the disturbed areas or the less flooded eco-types, increased over the same period. The most extensive land cover change in the swamp was observed along the edge of the swamp and in particular along River Nzoia (Figure 2, Figure 3, *A* & *B*) comparing the 1973 and the 2001 images. This section also experienced comparatively extensive silt deposition along the Lake Victoria shoreline comparing the 1973 and the 2001 shoreline area (Figure 2; *see A,* Figure 3). Other changes that were attributed to silt deposition include cutting of a gulf to the north of River Nzoia by floating vegetation accumulation, which was prominent in 1973, (Figure 3, *see B*). The presence of floating macrophytes such as water hyacinth *Eichhornia crapssipes,* which is common in Lake Victoria might as well have contributed to the accumulation of silt due to the reduced silt dispersion by water waves especially in the sheltered areas along the shoreline. No such plume was recorded on the mouth of Yala River (Figure 3, see *C)*, which flows through the swamp and have minimal farming activities especially along the river mouths.

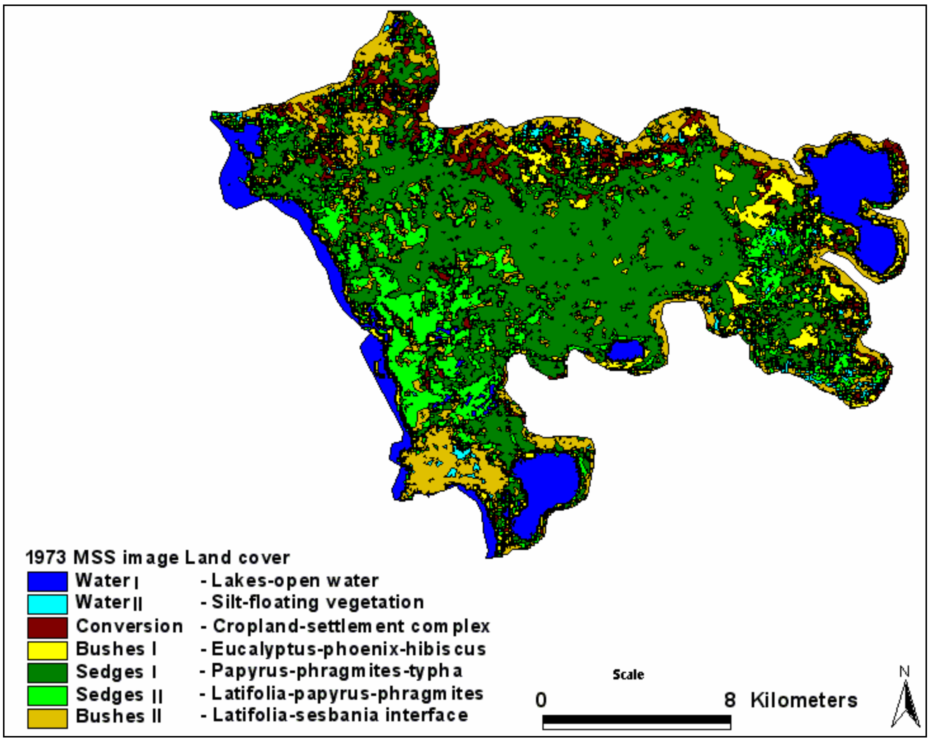


Figure 2. Land cover classes in 1973 (Image date 5th February 1973).

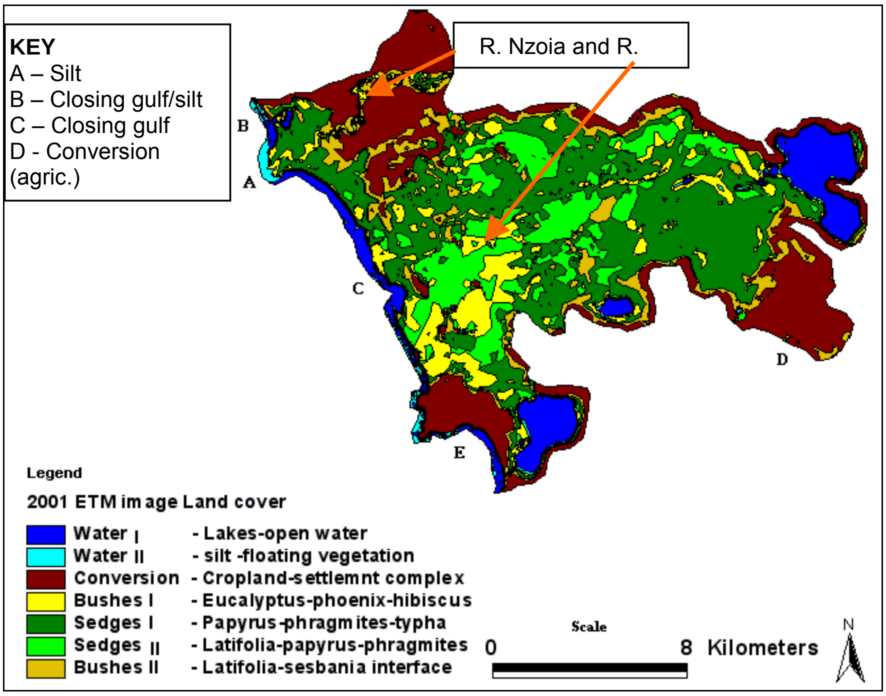


Figure 3. Land cover classes in 2001 (Image date 2nd February 2001).

Although the conversions to the northwest of Lake Sare area (Figure 2 and *see E* Figure 3) were due to the local community farming activities. The changes noted along the southern section of Lake Kanyaboli (*D*), were as a result of combined government policy initiatives in the early 1970s to drain this part of the swamp. This area was later taken up by the small-scale farmers. However, these two changes had no major effect in term of siltation of the water bodies system especially due to the absence of river canal.

Discussion and conclusions

Analysis of socio-economic and remote sensing data, point strongly at increasing farming trend over the years as the main cause of wetland degradation. Recent studies in East Africa on anthropogenic impact on wetlands point out at subsistence agriculture as a major driving force of wetland degradation (Githaiga *et al*., 2003). For example, between 1967 and 1990, Lake Naivasha lost over 3000 ha papyrus to agricultural activities expansion (Bemigisha, 1998). Similarly, long-term (1959-2000) studies in Uganda also point out at 50% rise in wetland farming as being attributed to subsistence farming. The shift in demographic factors like increased demand for, farming area in the Yala swamp and the need to raise income from the wetland led to more wetland degradation (Thenya *et al*., *In prep*). According to Mfundisi (2005), the total converted area between 1984 and 2001 was 6,333 ha over 17 years at a rate of 375.5 ha per year. Comparatively the total converted areas between 1973 and 2001 was 5639 over 28 years at a rate of 156 ha/ year. Although the discrepancies between the two analyses can be attributed to analysis and time of acquiring the image in 1984, the rate of change is much higher after the 1980s. This can be attributed to increased demand for farming land. This period was also associated with macro-economic changes related to IMF structural adjustments that have been associated with deforestations in developing countries (Merten *et al.,* 2000).

There is high dependence on the Yala swamp resources especially for farming by the local communities. This trends has been observed in other wetland ecosystem in the Lake Victoria basin (Mugo and Shikuku, 2000; Gichuki *et al*., 2001), which contributes primarily to land cover change including wetland degradation. The main disturbances, which emanates from the expansion of cultivation activities by the local communities. These changes include reduction of macrophyte coverage and increase of bushes as observed from 2001 image. Otieno *et al.,* (2001) made similar observation in terms of vegetation cover change using 2001 images, and linked the variability to farming activities as well. Similar observations in reduction of macrophyte coverage have also been made in Lake Naivasha fringing swamp coverage, which was also directly related to the expansion of farming communities (Bemigisha, 1998). These changes in vegetation land cover extend to the catchments areas, which contributes to increased river discharge according to JICA, (1980) and Sangale *et al.,* (2001). The catchments areas have over the years experienced extensive deforestation, over grazing and low annual cropping Sangale *et al.,* (2001). These disturbances are compounded by demographic changes, especially population increase, demand for farming land and the failure of resources management institutions.

Among the three East Africa state Kenya has the largest contribution of nutrient (nitrogen and phosphorus) load into the lake from urban waste and runoff, which is more than double the amount contributed from the other two countries. However, Tanzania has a higher load of industrial waste than the other two countries (COWI, 2002). In particular, River Nzoia on the Kenyan sub-basin have the largest catchment in the Lake Victoria basin and since it flows through denuded landscape it contributes the largest percentage of nitrogen and phosphorus at 3340 and 946 ton/year. According to Odada *et al*, (2004), the largest contribution of nutrients emanates from river flow and run-off, which constitute more than 50% with atmospheric contribution making 40%. This means that direct land activities mainly farming is responsible for nutrient load into the lake. This is made possible because of the degradation of the fringing swamps like Yala and Nyando swamp reducing their buffering capacity.

The effects of both the swamp degradation through farming and repeated macrophyte harvesting was clearly reflected in the accumulation of silt along the water bodies in particular along Lake Victoria shoreline. Previous records on water quality data in Lake Victoria indicate that the Secchi transparency index has progressively declined especially along the shores of Lake Victoria between 1930s and 2005. In the 1930s the values ranged between 0.5-7.9 meters (m) according to (Worthington, 1930), 1.5-2.8 m, Mavuti and Litterick, (1992), 0.6-2.3 m, Calamari *et al*., (1995) and Njuru (2001) 0.6-2.8m. The shoreline has approximate 0.7m secchi disc index compared to open lake of 2.9 m (Njuru and Hecky, 2005). These changes have also been attributed to the effect of catchment degradation, which has overtime increased rivers discharge and soil erosion to the downstream wetland ecosystem with reduced buffering effect (ICRAF, 2000; Sangale *et al.,* 2001). This has lead to the deposition of nutrient rich soil from the catchments deforested areas, which would have otherwise been trapped by the macrophytes. In (2000), ICRAF noted high contribution of nitrogen and phosphorous from catchments areas, both farming areas as well deforested areas. In contrast, the inland satellite lakes in the Yala swamp ecosystem do not show such turbidity increases due to the presence of macrophytes fridges along these water bodies.

The effect of sediment transport and macrophytes clearing has two pronounced effects in the north western section of the Yala swamp. First, the nutrients that are flushed into the water from both the degraded catchments and riparian farming areas encourage the proliferation of floating macrophytes such as water hyacinth among other floating macrophytes, which in turn reduce the dispersion of water from the shoreline and sheltered areas as observed at the mouth of River Nzoia. River Yala, which flows through the swamp, have no such plume at its mouth, which was also observed by Otieno *et al*., (2001). Secondly, the effect of combined cultivation and the deposition of soil from the catchment to the River Nzoia floodplain have also contributed to a shift in river Nzoia canal northward as it approaches the Lake Victoria. It has been noted that high sediments transport into Lake Victoria basin has also been attributed to the physical characteristics of Kenya drainage basins. These include larger average annual rainfall, average slope and sediment transport capacity than the other sub-basins of LVB (Odada, *et al*., 2004). These factors encourage transportation of high sediments amounts especially where there is reduced or sparse vegetation land cover. .

Conclusion

- The absence of ecosystem management plan in the Yala swamp and unclear land ownership in the swamp contribute to unsustainable land management
- The causes of land use degradation in the Yala swamp are diverse, but the demand to meet local livelihood needs in terms of food supply and income are important factors
- The clearing of macrophyte especially along river systems reduce the buffering effectiveness of the macrophyte in filtering the water
- Deforestation and un sustainable land use in the catchment contribute to increased sediment transport
- This calls for concerted efforts to restore the ecosystem and halt the Lake water siltation

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References

Abila, R., 1998. Utilisation and economic valuation of the Yala swamp wetland, Kenya. *In.* Gawler, M. ed. *Strategies for wise use of wetlands: best practices in participatory management*. Proceedings of 2nd International Conference on Wetlands and Development, (November 1998, Dakar, Senegal). Wetlands International IUCN, WWF Publications No. 56, Wageningen, The Netherlands.

Adelaida, K. S., 2000. *The wetlands of sub-Saharan Africa, their value and management*. Botany Department, University of Dar es Salaam, Tanzania.

Aloo, P. A., 2003. Biological diversity of the Yala swamp lakes, with special emphasis on fish species composition, in relation to changes in the Lake Victoria Basin (Kenya): threats and conservation measures. *Biodiversity and Conservation* 12, 905–920

Alwashe, M. A., and Bokhari, A. Y., 1993, Monitoring vegetation changes in Al Madinah, Saudi Arabia, using Thematic Mapper data. *International Journal of Remote Sensing*, 2, 191-197.

Benson, B. J., Mackenzie, M. D., 1995. Effects of sensor spatial resolution on landscape structure parameters. *Landscape Ecology* 10, 13-120.

Binswanger, H. P., McIntire, J., 1987. Behaviour and material determinants of production relations in land abundant tropical agriculture. *Economic Development and Cultural Change* 36, 73-99.

Calamari, D., Akech, M. O., Ochumba, P. B. O., 1995. Pollution of Winam Gulf, Lake Victoria, Kenya: A case study for preliminary assessment. *In.* Njuru, P. G., 2001. *An overview of the present status of Water Quality of Lake Victoria, Kenya. A limnological perspective*. Proceedings of Lake Victoria Environment Management Programme (LVEMP), First Regional Scientific Conference 3rd-7th December 2001, Kisumu, Kenya.

Daily, G. C., ed, 1997. Nature's services: Societal dependence on natural ecosystems. In: Ozesmi, S. L., Bauer, M. E., 2002. Satellite remote sensing of wetlands. *Wetland Ecology and Management* 10 (5), 381-402.

Drijver, C. A., Van Wetten, J, C, J., 2000. Sahel wetlands 2020. *Water and environment: Key to Africa's development.* Centre of Environmental Science (CML), University of Leiden, The Netherlands.

Ducros-Gambart, D., and Gastellu-Etchegorry, J. P., 1984, *Automatic analysis of bitemporal*

Duda, A. 2002. Restoring and protecting the African Great Lake Basin ecosystems –lessons from the North American Great Lakes and the GEF. In: *The East African Great Lakes: Limnology, Palaeolimnology and Biodiversity.* Odada E. O. and Olago D. O. (eds). Advances in Global Change Research, Kluwer Academic Publishers, pp. 537-556.

Farber, S., Constanza, R., 1987. The economic value of wetlands systems. In Mohamed, M. A., 1998. Mitigating the effects of intensive agriculture on wetlands: the Case of Saiwa Wetlands, Kenya. *In.* Gawler, M. ed. S*trategies for wise use of* *wetlands: best practices in participatory management.* Proceedings of 2nd International Conference on Wetlands and Development, (November 1998, Dakar, Senegal). Wetlands International IUCN, WWF Publications No. 56, Wageningen, The Netherlands.

Gichuki, J., Guebas, F. D., Mugo, J., Rabuor, C. O., Triest, L., Dehairs, F., 2001. Species inventory and the local uses of the plants and fishes of the Lower Sondu Miriu wetland of Lake Victoria, Kenya. *Hydrobiologia* 458, 99–106.

Githaiga, J. M., Reid, R., Muchiru, A., van Dijk, S., 2003. *Survey of water quality changes with land use type, in the Loitokitok Area, Kajiado District, Kenya*. Land Use Change Impacts and Dynamics (LUCID) Project Working Paper 35; Nairobi, Kenya: International Livestock Research Institute.

Government of Kenya, 1994. *District development plan*. Siaya District. Government Printers, Nairobi.

Green, G. M., Sussman, R, W., 1990. Deforestation history of the eastern rain forest of Madagascar from satellite images. *Science* 2, 212-215.

Halls, A. J., (Eds.), 1997. *Wetlands, biodiversity and the Ramsar Convention: the role of the convention on wetlands in the conservation and wise use of biodiversity*. Ramsar Convention bureau, Gland, Switzerland.

Hoekstra, D., Corbett, J., 1995. *Sustainable agricultural growth for highlands of east and central Africa: prospects to 2020*. Paper presented at the EcoRegions of the developing world: a lens for assessing food, agriculture and the environment to the year 2020. In ICRAF, 2000. Improved land management in the Lake Victoria basin: linking land and lake, research and extension, catchment and Lake Basin. First Technical Report, Start-up Phase, July 1999 to June 2000. ICRAF and Ministry of Agriculture and Rural Development.

Hollis, G. E., 1990. *Environmental impact of development on wetlands in arid and semi-arid lands. In: Wetlands and Integrated River Basin Managemen*t. Experiences in Asia and the Pacific. United Nations Environment Programme, Nairobi.

Hughes, R. H., Hughes, J, S., 1992. *A directory of African wetlands.* IUCN, Gland, Switzerland and Cambridge, UK, xxxiv+820pp, 48 maps.

ICRAF, 2000. *Improved land management in the Lake Victoria Basin: linking land and lake, research and extension, catchment and Lake Basin*. First Technical Report, Start-up Phase, July 1999 to June 2000. ICRAF and Ministry of Agriculture and Rural Development.

Jha, C. S., and Unni, N. V., 1994, Digital change detection of forest conversion of a dry tropical Indian forest region. International *Journal of Remote Sensing*, 13, 2543-2552.

JICA, 1980. National water master plan, stage 1 study report. *In.* Sangale, F., Okungu, J., and Opango, P., 2001. *Variation of flow of water from Rivers Nzoia, Yala and Sio into Lake. Victoria. LVEMP, First Regional Scientific Conference 3rd-7th December 2001, Kisumu, Kenya*.

JICA, 1991. National water master plan. Republic of Kenya, Ministry of Water Development. *In*. Sangale, F., Okungu, J., and Opango, P., 2001. *Variation of flow of water from Rivers Nzoia, Yala and Sio into Lake. Victoria*. LVEMP, First Regional Scientific Conference 3rd-7th December 2001, Kisumu, Kenya.

Johnson, T. C., Kelts, K., Odada, E., 2000. *The holocene history of Lake Victoria*. AMBIO: A Journal of the Human Environment 29 (1), 2-11.

Jones, J. R., 1998. Colonization in Central America. *In.* Kok, K., 2004. The role of population in understanding Honduran land use patterns. *Journal of Environmental Management* 72, 73-89.

Kairu, J. K., 2001. *Wetland Use and Impact on Lake Victoria, Kenya Region*. Lakes and Reservoirs. Research and Management 6, 117 - 125.

Kareri, R. W., 1992. The sociological and economic values of Kenya's wetlands. In Crafter, S. A., Njuguna, S. G and Howard, G. W. eds., 1992. *Wetlands of Kenya*. Proceedings of the Kenya Wetlands Working Group Seminar on Wetlands of Kenya, National Museums of Kenya, Nairobi, Kenya 3-5 July 1991. Vii + 183pp.

Kaufman, L., Ochumba, P. B., O.1993. Evolutionary and conservation biology of cichlid fishes as revealed by faunal remnants in northern Lake Victoria. *In.* Gawler, M. ed. S*trategies for wise use of wetlands: best practices in participatory management*. Proceedings of 2nd International Conference on Wetlands and Development, Dakar, Senegal. Wetlands International IUCN, WWF Publications No. 56, Wageningen, The Netherlands.

Kloer, B. R., 1994. *Hybrid parametric/non-parametric image classification*. Paper Presented at the ACSM-ASPRS Annual Convention, Reno, Nevada.

Lillesand, T. M., Kiefer, R. W., 1987. Remote sensing image interpretation. 2nd Edition. John Wiley, Chichester and New York.

Maclean, I, M, D., Tinch, R., Hassal, M., Boar, R., 2003. *Social and economic use of wetland resources: A case study from Lake Bunyonyi, Uganda*. Centre for Social and Economic Research on the Global Environment, University of East Anglia (CSERGE), Working Paper ECM 03-09.

Maltby, E., 1986. *Waterlogged wealth: Why waste the world's wet places?* IIED, London.

Maltby, E., 1988. *Global wetlands – history, current status and future*. UNEP.

Mavuti, K. M., 1989. Ecology of the Yala swamp. *Resources* 1, 16 - 19.

Mavuti, K. M., 1992. An account of some important freshwater wetlands of Kenya. *In*. Crafter, S.A., Njuguna, S. G and Howard, G. W. eds., 1992. *Wetlands of Kenya*. Proceedings of the Kenya Wetlands Working Group Seminar on Wetlands of Kenya, National Museums of Kenya, Nairobi, Kenya 3-5 July 1991. Vii + 183pp.

Mavuti, K. M,. Litterick, M. R., 1992. Composition, distribution and ecological role of zooplankton community in Lake Victoria, Kenyan waters. *In.* Njuru, P. G., 2001. *An overview of the present status of Water Quality of Lake Victoria, Kenya: A limnological perspective*. Proceedings of Lake Victoria Environment Management Programme (LVEMP), First Regional Scientific Conference 3rd-7th December 2001, Kisumu, Kenya.

Mertens, B., Sunderlin, W. D., Ndoye, O., Lambin, E. F., 2000. Impact of macroeconomic change on deforestation in South Cameroon: integration of household survey and remotely-sensed data. *World Development* 28 (6), 983-999.

M'mayi, P., Katua, S., Mbarire, S. K., eds., 1997. *Buffering capacity of Lake Victoria wetlands project and the sustainable utilization and management of wetlands and wetlands products pilot project. Proceedings of the induction workshop of the wetland management component of Lake Victoria Environment Management Programme, National Environment Secretariat.* Ministry of Environment and Natural Resources.

Mohamed, M. A., 1998. Mitigating the effects of intensive agriculture on wetlands: the case of Saiwa wetlands, Kenya. In: Gawler, M. ed., S*trategies for wise use of wetlands: best practices in participatory management.* Proceedings of 2nd International Conference on Wetlands and Development, (November 1998, Dakar, Senegal). Wetlands International IUCN, WWF Publications No. 56, Wageningen, The Netherlands.
Mugisha, S., 2002. *Root causes of land cover/use change in Uganda: An account of the past 100 years. Land Use Change Impacts and Dynamics (LUCID) Project Working Paper 14.* Nairobi, Kenya: International Livestock Research Institute.
Mugo, F, W., Shikuku, O., 2000. *Improved land management in the Lake Victoria Basin: linking land & lake, research and extension, catchment and Lake Basin. Natural resources use, constrains and possible intervention. The case for Ombaka Wetlands Focal Area in Nyando District*. World Agroforestry Centre-ICRAF, Nairobi.
Njuru, P. G., 2001. *An overview of the present status of Water Quality of Lake Victoria, Kenya: A limnological perspective. Proceedings of Lake Victoria Environment Management Programme (LVEMP).* First Regional Scientific Conference 3rd-7th December 2001, Kisumu, Kenya.
Ocholla, J., 1997. Buffering capacity activity in the Yala swamp: In M'mayi, P., S. Katua, and S, K. Mbarire, eds., 1997. *Buffering capacity of Lake Victoria Wetlands project and the sustainable utilization and management of wetlands and wetlands products pilot project.* Proceedings of the induction workshop of the wetland management component of Lake Victoria Environment Management Programme, National Environment Secretariat, Ministry of Environment and Natural Resources.
Odada E. O., Olago, D. O., Kulindwa, K., Ntiba, M., and Wandiga, S., 2004. Mitigation of Environmental Problems in Lake Victoria, East Africa: Causal Chain and Policy Options Analyses. *Ambio* Vol. 33 No. 1–2.
Odak, O., 1987. Materials culture and design. *In.* Kareri, R. W., 1992. *The sociological and economic values of Kenya's wetlands*. *In.* Crafter, S.A., Njuguna, S. G and Howard, G. W., eds., 1992, *Wetlands of Kenya*. Proceedings of the Kenya Wetlands Working Group Seminar on Wetlands of Kenya, National Museums of Kenya, Nairobi, Kenya 3-5 July 1991. Vii + 183pp.
Ogutu, M.A., 1987. Fisheries and fishing, Pages 86-105. *In.* Republic of Kenya, 1987. S*iaya District Socio- Cultural Profile*. Ministry of Planning and National Development and Institute of Agricultural Studies, University of Nairobi, Kenya.
Otieno, M, N., Odenyo, V, O, A., Okeyo, J, B., 2001. *The capabilities of multispectral Landsat TM for mapping spatial landscape characteristics of a tropical wetland. The case of Yala swamp, Kenya*. In proceedings of LVEMP, First Regional Scientific Conference 3rd-7th December 2001, Kisumu, Kenya.
Panayotou, T., 1994. *The economics of environmental degradation: Problems, causes and responses. In: Pattanayak, S., Kramer, R. A., 1999. Pricing ecological services: Willingness to pay for drought mitigation in eastern Indonesia*. Land Economics.
Pender, J., 1998. Population growth, agricultural intensification, induced innovation and natural resource sustainability: An application of neoclassical growth theory. *Agricultural Economics*. 19, 99-112.
Pilon, P. G., Howarth, P. J., and Bullock, R. A., 1988. An enhanced classiffication approach to change detection in semi-arid environments. *Photogrammetric Engineering and Remote Sensing,* 12, 1709 -1716.
Pingali, P., Bigot, T., Binswanger, P. H., 1987. *Agricultural mechanisation and the evolution of farming systems in Sub-Saharan Africa* (pp. 216). Baltimore, MD, USA: The John Hopkins University Press. (A World Bank publication).
Reid, R. S., Gachimbi, L. N., Worden, J., Wangui, E. E., Mathai, S., Mugatha, S., M., Campbell, D. J., Maitima, J. M., Butt, B., Gichohi, H., and Ogol, E., 2004. *Linkages between changes in land use, biodiversity and land degradation in the Loitokitok Area of Kenya. Land Use Change Impacts and Dynamics (LUCID) Project Working Paper 49 Nairobi, Kenya*:. International Livestock Research Institute.
Ruthenberg, H., 1976. Farming systems in the tropics, 2nd ed. Oxford, England: Oxford University Press.
Sangale, F., Okungu, J., Opango, P., 2001. Variation of flow of water from Rivers Nzoia, Yala and Sio into Lake Victoria. *Proceedings of Lake Victoria Environment Management Programme (LVEMP), First regional scientific conference 3rd-7th December 2001, Kisumu, Kenya*.
Schelhas, J., 1996. Land use choice and change: intensification and diversification in the lowland tropics of Costa Rica. *Human Organization* 55, 298-306.
Saura, S., 2003. Effects of remote sensor spatial resolution and data aggregation on selected fragmentation indices. *Landscape Ecology* 19, 197-209.
Thenya, T., 2001. Challenges of conservation of dryland shallow waters, Ewaso Narok Swamp, Laikipia District Wetlands, Kenya. *Hydrobiologia* 458, 121-130.
Thompson, K., 1976. Swamp development in the head waters of the White Nile. *In.* J Rzoska ed. *The Nile Biology of an Ancient River*. Dr. W. Junk BV, Publisher. The Hague.
Torrion, J., 2002. *Land degradation detection mapping and monitoring in the lake Naivasha basin Kenya*. International Institute for Geo-Information Science and Earth Obervation Enschede, The Netherlands.
Turner, M. G., O'Neill, R. V., Gardner, R. H., Milne, B. T., 1989. Effects of changing spatial scale on the analysis of landscape pattern. *Landscape Ecology* 3, 153-162.
Wood, A., 1997. *Wetlands in highlands: The search for sustainable use of valley bottom swamp in the highlands of South-West Ethiopia.* Department of Geographical and Environmental Science, University of Huddersfield, UK.
Wood, A., Hailu, A., Abbot, P., Dixon, A., 1998. Sustainable management of wetlands in Ethiopia: local knowledge versus government policy. *In.* Gawler, M. ed. *Strategies for wise use of wetlands: best practices in participatory management, Proceedings of 2nd International Conference on Wetlands and Development, (November 1998, Dakar, Senegal).* Wetlands International IUCN, WWF Publications No. 56, Wageningen, The Netherlands.
Wood, C. H., Skole, D., 1998. Linking satellite census and survey data to study deforestation in the Brazilian Amazon. *In.* B. Mertens, W. D., Sunderlin, O., Ndoye, Lambin, E. F., 2000. *Impact of macroeconomic change on deforestation in South Cameroon: integration of household survey and remotely-sensed data*. World Development 28 (6), 983-999.
Worthington, E. B., 1930. Observations on the temperature, hydrogen-ion concentration, and other physical conditions of the Victoria and Albert Nyanzas. *In.* Njuru, P. G., 2001. *An overview of the present status of Water Quality of Lake*

Victoria, Kenya: A limnological perspective. Proceedings of Lake Victoria Environment Management Programme (LVEMP), First Regional Scientific Conference 3rd-7th December 2001, Kisumu, Kenya.
Wu, J., Jelinski, D. E., Luck, M., Tueller, P. T., 2002. Multiscale analysis of landscape heterogeneity: scale variance and pattern metrics. *Geographical Information Science* 6, 6-19.

Problems of a small reservoir: A case study of Makwaye Lake in the Savanna Region of Northern Nigeria

Balarabe, M. L. and J.K Balogun

Department of Biological Sciences, Ahmadu Bello University, Zaria, Nigeria
Email:mlbalarabe2001@yahoo.com

Abstract

Climatic variation and human activities have intensified over the years around catchment basins of lakes in the savannah region of Northern Nigeria. The rainfall distribution pattern has been erratic coupled with threats of desertification and population growth. Deforestation, bush burning, overgrazing, intensive farming and indiscriminate use of agrochemicals are on the increase. These have detrimental impact on the aquatic ecosystems. Water retention capacity, physico-chemical and biological characteristics of reservoirs depend on the catchment characteristics, human activities and climatic regime. The implication of these have resulted to increased siltation, reduction in water storage capacity, nutrient build up, eutrophication and upsurge in aquatic weeds infestation thereby threatening the water quality status and its productivity. The paper highlights the ecological impacts of human activities and possible remedies.

Key words: Aquatic, Ecosystem, Savanna, Makkwaye lake, Nigeria

Introduction

In Nigeria over 12.5 million hectares of inland waters have been identified (Ita *et al.,*, 1985). Nigeria's landmass is characterized by distinctive geographical profile and climatic regimes, which determine rainfall duration pattern, nature of vegetation and seasonal characteristics of different areas. The vegetation ranged from the tropical rainforest of the southern coastal areas to the savanna in the North. The savanna is zoned into Guinea, Sudan and Sahel, which accounted for about 554,400Km^2 or 60% of the total landmass of the country.

In most parts of sub-humid regions of the world such as the savanna region of Northern Nigeria, the unreliability of rainfall has led to the development of surface water storage devices such as dams and lakes for agriculture, livestock watering, fisheries and domestic use. Reported estimates have shown that over 80% of the existing surface-area of reservoirs in Nigeria are located in the Savanna zones (Ita *et al.,*, 1985) where in addition to their recorded potentials of boosting fish production they are actively used for dry season and irrigation farming (Mbagwu and Ita, 1994).

The ever-increasing population of developing countries like Nigeria has necessitated the intensification of agricultural activities with the aim of accelerating food production. The bulk of grain production in the country takes place in the savanna. The general ecology of the Savanna region of Northern Nigeria is faced with climatic changes probably due to global warming and intense human activities. These resulted to erratic rains (drought) coupled with advancing desert encroachment where 15 to 75% of the areas are facing the threats of desertification. The region is also faced with rapid population growth rate with increasing demand of water, agricultural land and fuel wood. These accounted for massive deforestation, bush burning, overgrazing and intense cultivation for both rain-fed and irrigation farming. Changes in climatic patterns and human activities are of growing concern on the environmental quality of the region.

Water retention capacity depends on the amount of rainfall received, size of the water body, topography, nature of soils, evaporation, seepage, and extent of use and other induced human activities. Deforestation, farming activities, grazing, and bush burning enhances accelerated silting, addition of large quantities of nutrients, chemicals, pesticides and organic matter into water bodies through surface run-off (Gopal *et al.,*, 1981). Ecosystems in their natural state consist of integrated and self-regulating community of organisms in harmony with the physical and socio-cultural environment in a self-sustaining fashion. Water quality management requires the understanding of the relationship between biotic and abiotic components of the aquatic environment (Cosser, 1988). Monitoring the biological and physico chemical characteristics of reservoirs with respect to their productivity for short or long term trends is important because it enables appropriate decisions and planning to be made in order to minimize deleterious effects (Stent,1981). Marshall and Maes (1994) highlighted among others the need for a review in information on ecological status of African freshwater environment, present potentials, problems and priorities in research so as to give guidelines for their conservation and management.

Water and food security are the most important issues in the arid and semi-arid regions of the world (Tolba et al, 1992). The peculiarity of Nigerian Savanna with respect to existing linkages between population upsurge, socio-cultural values and environmental conditions of watershed catchments will provide an insight on the present status of small water bodies with respect to protection, rational utilization and their sustainability. The aim of this paper is to assess the changes that have occurred over the last fifteen years in a small medium sized lake based on limnological study as reported by Balarabe and Oladimeji (1991 and 1992), Balarabe *et al.,* (2000) and the re-survey of the dry season of 2002/2003. The paper also attempts to explain the possible causes and suggests ways to arrest further deterioration.

The study area

Makwaye (Ahmadu Bello University) farm lake, is located in Zaria (Lat 11^0 28^1 N, Long 7^0 48^1E) (Figure 1). It was impounded in 1966 for irrigation of the University farm, livestock watering and other domestic uses. It has a total surface area of 950,000m^2. The catchment of the water shade is 1841.22 hectares. The land use of the catchment basin as reported by Kowal and Omolokun (1970) are as follows:-

- 6.1% is residential.
- 69.% is farmland of which 22% is occupied as experimental farm
- 17.1% is uncultivated land under natural fallows
- 5.1% is poor land used for range grazing by migrating herdsmen.

Part of the eastern side of the basin is intersected by a railway embankment, which does not interfere with the natural drainage (Kowal, 1970). About 30 tons of inorganic fertilizers is used annually at the experimental farm along with 200 to 250 litres of herbicides (Gramazone, Premagrin, Cesapril) and about 150 litres of Cymbush pesticides. The crops grown mainly are maize, sorghum, cowpeas and cotton. Substantial number of livestock consisting of cattle, goats, sheep, pigs, poultry and rabbits are reared in the farm.

Climate and soil

The climate is characterized by well-defined wet and dry season. The wet season lasts from the beginning of May to early October, August being the wettest month. Mean annual rainfall is about 1080mm. The dry season (Nov-April) lasts for about six months, December/January are the coldest months while April /March is the hottest with average temperature of up to 40^0C. The soils of the study area are part of tropical ferruginous soils characterized by low organic matter content, cation exchange capacity and sandy clay texture (Iguisi, 2002). During the long dry season the vegetation is subjected to overgrazing, bush burning and fuelwood cutting which leave the soil bare prone to erosion.

Methodology

Temperature, pH and electrical conductivity were measured in the field with Hanna pH 210 and Hanna EC214 conductivity meter. Water transparency was determined with Secchi disc of 25cm diameter. Other parameters were measured according to standard methods described by Lind (1979) and APHA (1980). The results are expressed as seasonal averages.

Results

The results of the physico-chemical parameters are shown in Table 1. pH, dissolved oxygen and temperature did not change significantly over the years.

There is wide variation in water hardness, Nitrate – Nitrogen (NO_3-N) and Phosphate –Phosphorus (PO_4-P). Water depth also showed a significant variation as was exhibited by reduction in Lake volume. Species diversity of aquatic macrophytes exhibited a trend in succession over the years as shown in Table 2.

Table 1: Dry Season Physico-Chemical Parameters of Makwaye Lake (Mean values ± s.e.m).

PARAMETER	1987/88	2002/2003
Temperature (^{0}C)	21.5± 3.50	22.3± 4.10
Water depth (m)	1.88±0.30	1.21±0.31
Secchi transparency (cm)	42.3±18.3	63.2±13.8
pH	7.45±0.19	7.33± 0.18
Conductivity (µmhoscm^{-1}	54.67±7.79	71.00±8.17
Water hardness (mg1^{-1}$CaCO_3$)	33.33± 2.42	55.42±8.68
Dissolved oxygen (mg1^{-1} $CaCO_2$)	6.57±0.82	6.63±0.32
NO_3-N (mg1^{-1})	1.97±0.26	4.13± 0.37
$P0_4$-P (mg1^{-1})	0.061±0.03	1.28±0.16

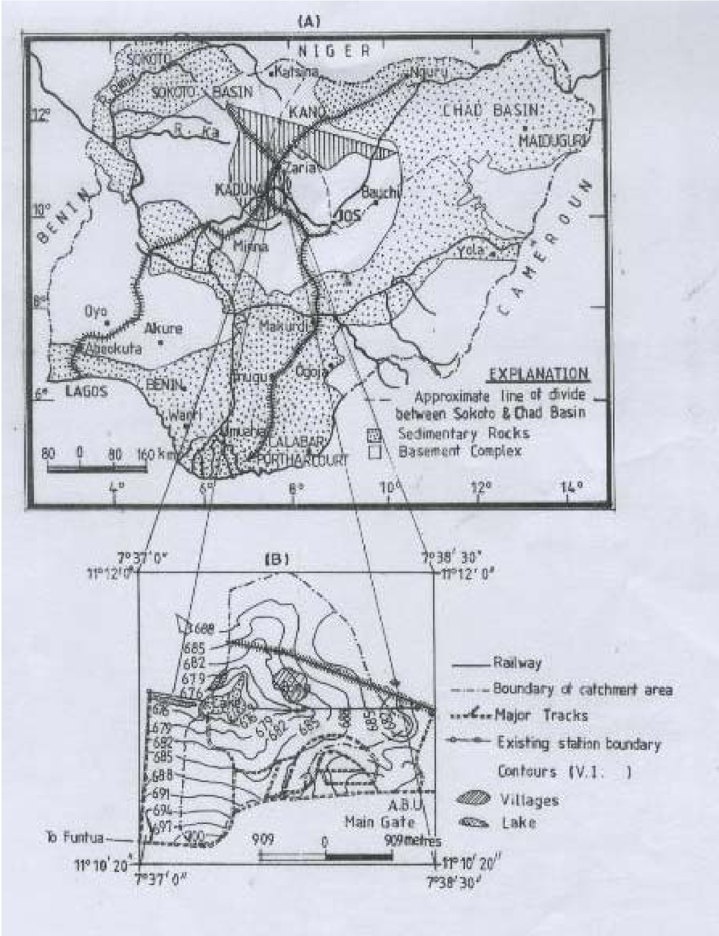


FIG. 1 : MAP OF NIGERIA SHOWING LOCATION OF THE STUDY AREA. (A), (B) MAGNIFIED
Source: KOWAL AND OMOLOKUN, 1970

Table 2: Aquatic macrophytes distribution in makwaye lake between 1987/88 and 2002/2003.

1987/88	**2002/2003**
Nymphea lotus	*N. lotus*
Nymphaea micrantha	*N. micrantha*
Typha australis	*T.australis*
Pistia stratoites	*Pistia stratoites*
Polygonum lanigerum	*Polygonum lanigerum*
Salvinia nymphellula	*Salvinia nymphellula*
	Phragmites karka
	Najas pectinata
	Helistropium ovaliforium
	Mimosa pigra
	Oryza longistaminata
	Maricus alternifolia

The aquatic macrophytes vegetation in 1987/88 was dominated by 5 different species. *Nymphaea lotus* and *Pistia stratoites* were the most dominant especially around the southwestern edges of the lake. *Tyhpa autralis* was not prominent except around the periphery of the western side. By 2002/2003, thirteen dominant species of both floating emergent and terrestrial plants were found inhabiting both littoral and surface area of the lake. Over 20% of the entire lake surface was dominated by aquatic weeds during the tail end of the dry season of 2002/2003. The distribution is concentrated in the pools along the exposed south western edges of the lake where the entire margins were dominated by *Typha autralis* while *Nymphaea lotus*, *Salvinia nymphellula* and *Pistia* are concentrated in the pools and the surface of the receding lake. The lake volume has receded over the years. This was exhibited by the gradual reduction in average depth of the lake. Similarly major nutrients i.e. NO_3-N and PO_4-P have shown remarkable variation over the years.

Discussion

Deforestation, human and livestock population have increased over the years around the catchment basin. About 24.2% of the catchment, which was left as fallow and poor lands for grazing, is now under intense cultivation. In addition to this as temporary water bodies around Zaria dry up, livestock tend to concentrate around the lake basin for both watering and grazing on the sprouting vegetation along the lake basin. Livestock watering make significant water demands, especially in semi-arid areas where surface water resources are scarce and long dry seasons are experienced (Ndege, 1996). Over three decades ago, Fry (1975) attributed the vegetation changes around Zaria (study area) was induced by the Sahelian drought of 1967 to 1974 as a result of human and livestock pressure.

The increasing rate of deforestation and land clearance in Africa, combined with poor agricultural practices and unrelenting growth of human population are the most serious threats to small water bodies (Marshall and Maes, 1994). In East Africa, the main factors that contribute to stress are population growth, irrigation, livestock watering, drought, deforestation, poor land management and pollution from human and industrial activities (Ndege, 1996). Water volume fluctuations due to seasonal changes at any given time determine both physico–chemical and biological characteristic of Savanna lakes. Reduction in lake volume is mainly due to high temperature, wind, evaporation, seepage, water abstraction for livestock watering and irrigation. In arid lakes evaporation is the dominant mechanism by which water is lost (Schindler, 1991).

Apart from rain-fed cropping, irrigation has intensified over the years throughout the entire lake edges leading to massive use of fertilizer and subsequent siltation due to deposition of eroded soil materials into the lake via surface runoff. Accumulation of organic matter in the sediments, associated changes in physico-chemical parameters, and reduction in water volume are some of the major limnological changes associated with reservoir ageing. Deposition of sediment loads to water courses results in faster sedimentation of reservoirs and their gradual extinction (Tolba et al, 1992). The ecological and use characterization of lakes drainage react in different ways to changes occurring in their drainage basin (Chmilewesky *et al.,*, 1995). The proliferation of aquatic weeds could be associated with the ecological succession from lacustrine to terrestrial environment attributed to heavy siltation, water level fluctuations due to annual draw down and increased nutrient enrichment leading to accelerated eutrophication.

In addition to proliferation of aquatic weeds, the average depth of most reservoirs within 100km radius from Makwaye lake have reduced by about 50% of their previously reported mean depth due to siltation and sedimentation (Iguisi & Abubakar, 1998; Gbem *et al.,*, 1999; Balogun *et al.,*, 2000; Oniye *et al.,*, 2000). Dutsen ma dam that was impounded along with others have completely silted up Oniye and Balarabe (2000).

With respect to other water quality parameters, dissolved oxygen, pH and temperature were within favourable range acceptable to fisheries production. However, based on Exchangeable Sodium Percentage (ESP) of Makwaye Lake, the water is unsuitable for irrigation because of high sodium concentration (Balarabe *et al.,*, 2000).

Recommendation

In order to ameliorate the ageing process and siltation of Makwaye and other similar lakes in the Savanna there is the need to:

i. Initiate community involvement in sustainable agricultural practice and protection of water resources.

ii. Massive reforestation especially around the lake edges

iii. Mechanical clearance of aquatic weeds through traditional communal effort.

iv. Construct sediment traps and buffer zone (i.e. non farming) within the immediate vicinity of the lake

v. Stakeholders forum and community based water resources management for the protection of water.

vi. Sustainable farming systems and livestock management by creating designated livestock watering points, so as to stop trampling of soil and feacal deposition around the lake margins.

vii. Community awareness and collective joint protection measures involving all stakeholders along the catchment.

Conclusion

There is the need to develop sustainable protection of measures of the aquatic resources in the savanna, and alternative energy source to reduce deforestation and a country wide desertification control measures. There is the need to develop catchment protection measures, sustainable agricultural practice and community based water resources management and development. The rapid population growth, changes in rainfall distribution pattern, diverse human activities and water stress in the Savanna region of Northern Nigeria require urgent attention to develop sustainable aquatic resources management so as to safeguard the livelihood of human and livestock requirements.

References

APHA, 1980. American Public Health Association. Standard methods for the Examination of water and wastewater. Washington DC 15th Edition 1076pp.

Balarabe, M.L and Oladimeji, A.A (1991. *Some metal concentration in the surface water and sediment of Makwaye Lake (ABU) Farm) lake, Samaru Zaria, Nigeria.* A paper presented at the National Conference on water quality monitoring and status in Nigeria. 11th to 18th October 1991 Durbar Hotel Kaduna.

Balarabe, M.L and Oladimeji, A.A, 1992. *Physicochemical Limnology of Makwaye (Ahmadu Bello University(Farm. Lake, Samaru Zaria, Nigeria.* A paper presented at the seventh annual conference of the National Association of Aquatic Sciences (NAAS) held at the University of Ilorin, Kwara State Nigeria 25th to 28th October, 1992.

Balarabe, M.L and Oladimeji, A.A and Abubakar, S.M, 2000. An assessment of surface water quality of Makwaye farm lake in Samaru Zaria. *In.* J.A Falla, K. Ahmed, M.A Liman and A Maiwada eds. *Issues in Land administration and development in Northern Nigeria*. Proceedings of the National workshop on land administrating development in Northern Nigeria.

Balogun,J.K, Auta, J. Balarabe, M.L and Bako, S.P, 2000. *Fisheries Management and development. Consideration for small reservoirs: a case study of Zaria reservoir.* A paper presented at the 15th Annual Conference of the fisheries Society of Nigeria (FISON, held at university of Jos, Nigeria 19th to 25th March 2000.

Chimilewesky, T.J., Stanisland, R and Sielweiz, B, 1997. Changes in ecological relationship in a group of eight shallow lakes in the Polesie Iyebeiske region, Eastern Poland) over forty years. Hydrobiologia:342/343:285-295.

CIFA, 1995. Committee for Inland Fisheries in Africa *Gbem, T.T.*

Cosser, P.R, 1988. Macroinvetebrate Community Sructure and Chemisty of organically polluted Creek in South East Queens land. *Aust. Mar.Freshwater Res* 39:671-683.

Fry, C.H, 1975. *Arid Zone Elements in Northern Guinea Savanna* Avi fauna-A Reply Savanna vol. 4 No.2:197-199.

Gbem, T.T., M.L. Balarabe, S.P Bako, J. Auta and J.K Balogun, 1999. *A study of the changes in physical characteristics of a small man-made fresh water ecosystem in relation to aquatic vegetation: A case study of Borno lake, Zaria* (1987-1993. Proceeding of large and small scale irrigation schemes in Africa.15th to 19th November 1999. Abuja, Nigeria. pp307-315.

Gopal, B., Goel, P.K., Sharma, K.P and Rivedy, Q.K, 1981. Limnological study of a Freshwater reservoir, Jamuwa Ranggarh (Jaipur) reservoir, *Hydrobiologia* 83:287-294.

Iguisi, E.O., 2002. An assessment of temporal variation in soil loss within a hydrological year. *The Zaria Geographer.* Vol. 15:125-134.

Ita, E.O, Sado E.K, Balogun, J.K., Pandogari, A. and Ibitoye, B., 1985. *Inventory Survey of Nigeria Inland waters and their fishery resources*. A preliminary checklist of Kainji Lake research Institute Technical report series No.14:51pp

Kowal, J., 1970. The hydrology of a small catchment basin at Samaru, Nigeria. II. Assessment of main component of water budget. *Nigeria Agric Journal* Vol 7:41-52

Kowal, J. and Omolokun, A.O, 1970. The hydrology of a small catchment basin at Samaru, Nigeria I. Seasonal fluctuations in the height of ground water table. *Nig. Agric. Journal* Vol. 7. No. 1:27-40.

Lind, O.T, 1979. *A Handout of Limnological Methods:* C.V Mosby Co. St. Louis 199pp

Marshall, B and Maes, M, 1994. Small water Bodies and their fisheries in southern Africa. CIFA Technical Paper 29.

Mbagwu, I.G and Ita, E.O, 1994. Pesticides use in the sub-humid Zones of Nigeria: Implications for Conservation of Aquatic Resources. *Environmental Conservation,* Vol 21, No:314-219. iii. Strain, water demand and supply directions in the most stressed water systems of eastern Africa.

Ndege, M.M, 1996. Water Management in Africa and the Middle East: Challenges and opportunities. E. Racheal', E. Rathseber and D. Brooks Eds. Inter Dev. Res. Centre pp 153-165

Oniye, S.J and Balarabe, M.L, 2000. *Impact of Human activities on some reservoirs and Rivers: Implications to fisheries.* A paper presented at the 15th annual conference of the fisheries society of Nigeria (FISON) held at University of Jos, Nigeria 19th to 25th March 2000.

Schindler, D.W, 1991. Lakes and Oceans as a functional model in fundamentals of Aquatic Ecology. *In.* Barnes, R.S.K and Mann, K.H., eds. *Blackwell Scientific Publication Second edition* pp. 91-107.

Stent R.F, 1981. Grafman Water: A discussion of Limnological events during the period 1974-1978. *Freshwater Biology* 11:567-578.

Tolba, M.K., El-Kholy., D., El-Hinnawi, E., Holdgate, M.W, Micheal, D and Munn. R.E, 1992. *The World Environment 1972-1992.* Two decades of challenge. Pp 83-104.

Spatial and historical changes of indigenous *O. mortimeri* following introduction of exotic *O. niloticus* in Lake Kariba

Chifamba, Portia Chiyedza

University of Zimbabwe, Department of Biological Sciences, P O Box MP 167, Mount Pleasant, Harare, Zimbabwe
Email:pcchifamba@yahoo.co.uk

Abstract

Gill net catches taken between October 2003 and December 2004, at nine sites in the Sanyati basin of Lake Kariba were analysed for patterns in the distribution of the introduced *Oreochromis niloticus* and the endemic *Oreochromis mortimeri*. Some environmental variables at the site were measured in order to test if these were correlated to fish catches. *O. niloticus* and *O. mortimeri* catch at each site differed depending on the environmental conditions and the distance from the fish farm. Near the farm the ratio of *O. mortimeri* to *O. niloticus* was 1:27.4 whereas the area furthest from the farm the ratio was 1: 0.5 showing the effect of dispersal from the point of introduction. The ratio in the area close to the fish farm had a mean ratio of 1:0.1 in the period September 1993 to June 1994. *O. niloticus* became dominant in the catch from the area close to the farm after a period of 10 years. Fish catch statistics from the area near the fish farm showed a significant exponential increase ($p < 0.001$) in the ratio between *O. niloticus* and *O. mortimeri* from 1993 (when it was first observed in the catch) and 2001. It is very probable that the introduced species will continue to increase in the areas further from the farm whist displacing the indigenous species at this fast rate. The catch per setting of *O. niloticus* was higher in the area close to extensive shallow marginal water possibly because these areas offered good nursery grounds. There was no correlation between fish catches and the overall slope of the area, macrophytes and the type of sediments, an indication that both species were able to tolerate a range of environmental conditions.

Key words: *O. niloticus*, impact, Lake Kariba, *O. mortimeri*.

Introduction

For decades the cichlid *O. mortimeri,* restricted in distribution to the Middle Zambezi River and the Luangwa tributary (Skelton 1993), was important in the Lake Kariba fishery. In 1962 this species contributed 35% to the artisanal fishery catches from Sinazongwe area. Later in 1974, it formed 38% of catches from Sanyati to Sengwa basins (Kenmuir 1984). This species contributed 16% to the Index of Relative Importance in the experimental fishing during the period from 1969 to 1991 (Karenge 1992). Formation of Lake Kariba following the close of Kariba dam in 1958 had enabled rapid expansion of populations of this species that favours lacustrine waters.

Introduction of *O. niloticus* into Lake Kariba, through escape from farming units in the early 1990s, was a potential threat to this species. Elsewhere in Africa, the introduced *O. niloticus* became important in fishery (Balirwa 1992; Vos *et al.,* 1990). Reduction in biodiversity due to *O. niloticus* introduction was observed in Lake Victoria (Ogutu Ohwayo, 1990) and Lake Luhondo (Vos *et al.,,* 1990). In 1992, *O. niloticus* was first recorded in catches from Lake Kariba. A study carried out in Lake Kariba, Sanyati basin in 1993-4 showed that the species had become established (Chifamba, 1998). At that time, *O. niloticus* made only a small contribution to catches (0.4% by weight) in the area of introduction, adjacent to Kariba town. Not a single specimen was recorded from the catches from the fishing villages of the Sanyati Basin then.

Because of diet overlap and superior growth performance on fish farms and success in other lakes in which it was introduced, competitive advantage over native species was predicted. Chifamba (1998) and Mhlanga (2000) indicated a diet overlap between *O. niloticus* with *O. mortimeri* in Lake Kariba. In Lake Victoria, growth and mortality rates have been estimated (Getabu, 1992) and found to be higher than in native lakes. Hence it was important to monitor the species composition and assess the impact of the introduction on the native species particularly on *O. mortimeri* that had a similar niche. The situation is an opportunity to observe competition in the natural environment.

This study aimed at determining the impact of the introduced *O. niloticus* on the native *O. mortimeri*. The historical changes in the catches of *O. mortimeri* and *O. niloticus* were examined to determine any changes in the abundances of these species due to their interactions. To assess dispersal of *O. niloticus* the spatial distribution of the two species with respect to the farming unit was also studied. To assess habitat overlap and hence potential competition for space and some environmental attributes were correlated to fish catches.

Methods

The study was based on field surveys carried out throughout the Sanyati basin as well as data from the Experimental Gill Netting Programme collected by Lake Kariba Fisheries Research Station (LKFRI) at Lakeside site (Songore, 2002).

The field data collection

Nine sites in the Sanyati basin (Figure 1) were sampled in 2003 and 2004 in order to study the habitat and fish distribution. Three sites Prawn Farm, Nyanyana and Zebra Island were located in the fish farming area. Tsetse, Nyamhunga and Monga sites were in the artisanal fishing area whilst Changachirere, Spurwing and Fothergill were in a non-fishing area and were furthest from the farming operations.

Fish samples were collected using gill nets ranging from 25mm to 178mm. Data on the characteristics of the shoreline (slope, bottom type, estuary, open lake shore, distance from the *O. niloticus* farming units) were recorded. The distance from the fish farm was estimated as a straight from a map. The shoreline and bottom type was described as rocky, sandy or silty from material sampled with a mud grab. The distance from the shore was estimated from the point with 0.5m water depths to the water's edge.

Data analysis

Pearson Correlation analysis was used to correlate fish catch data, for the two species, to environmental variables. To establish the relationship between the species a ratio between the catches of the species was calculated. These ratios were used to analyse the trends in time and space.

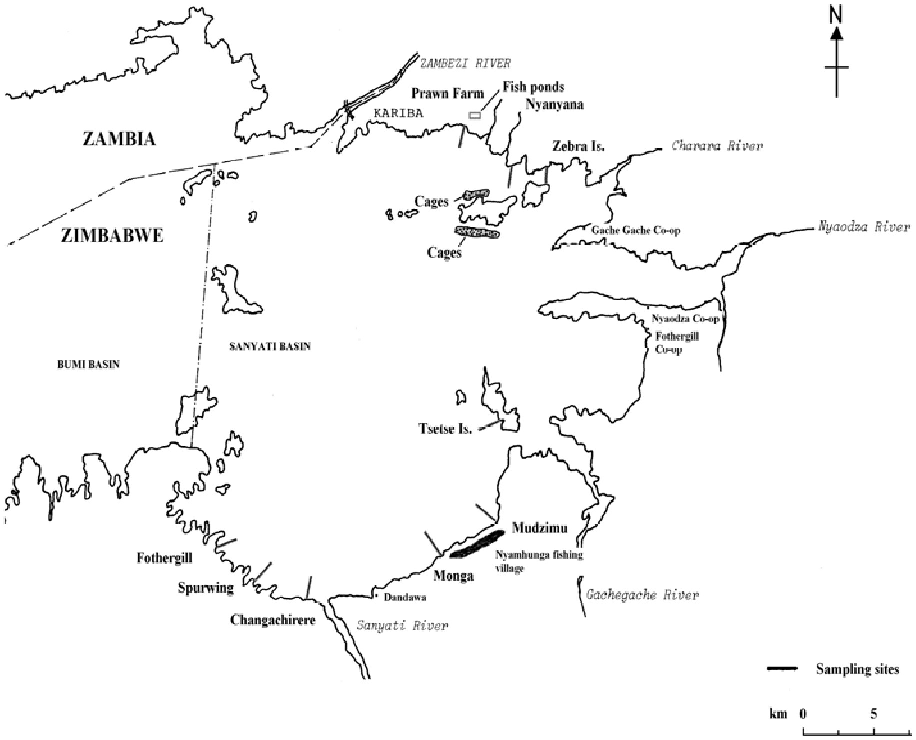


Figure 1. A map of the Sanyati basin of Lake Kariba showing the position of fishing sites, fishing villages, the fish ponds on the fish farm and cages.

Results

Distribution in the Sanyati basin

A spatial difference in the catches of both *O. niloticus* and *O. mortimeri* was observed (Figure 2a). Three sites had high catches of *O. niloticus*: Prawn farm close to the fish farm and Nyamhunga and Monga in an artisanal fishing area. No *O niloticus* was caught at site furthest from the fish farm. This trend was opposite to that of *O. mortimeri.* Near the farm (three sites combined) the ratio of *O. mortimeri* to *O. niloticus* was 1:27.4 whereas the non-fishing area furthest from the farm the ratio was 1: 0.5.

The distance from the fish farm explained a significant amount (77%) of variation in the distribution of *O. mortimeri* (Table 1). Ratios between *O. mortimeri* and *O. niloticus* catches in 2004 and 2005 combined, had a significant exponential relationship with distance from the fish farm (Figure 2b), as described by equation (1).

O. mortimeri/O. niloticus catches = $0.9887e^{0.0214x}$ -1 (R^2 = 0.69; $p<0.05$) (1)

The distribution of *O. niloticus* was significantly correlated to water turbidity and distance from the shoreline (Table 1). Oxygen was negatively correlated to the distribution of both species though the relationships were not significant. Sites with high catches Mudzimu and Monga, were characterized by low Oxygen and low pH, high turbidity and long distance from the shore (Figure 3).

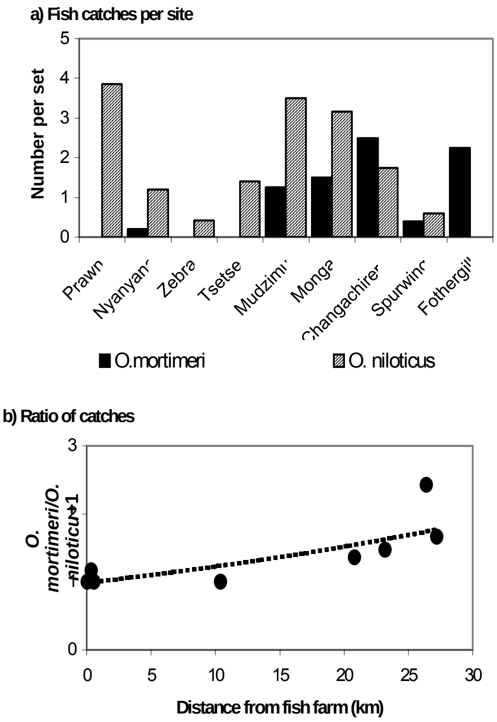


Figure 2. Fishing site differences in a) catches of *O. niloticus* and *O. mortimeri* and b) ratios of catches.

Table 1: The Pearson Correlation Coefficients for the relationships between oxygen, turbidity, distance from shore, distance from farm and catches of *O. mortimeri,* and *O. niloticus.* (* = p < 0.5, ** = p < 0.01); 2 tail significance levels)

Variable	*O. mortimeri*	*O. niloticus*
Oxygen	-0.54	-0.55
Turbidity	0.02	0.69*
Distance from shore	0.03	0.69*
Distance from farm	0.77*	-0.10

Historical catches from Lakeside

During the period 1993 to 2004, the catches of *O. mortimeri* from the LKFRI Experimental Gill Netting Programme increased to a peak in 1995 (Figure 4a). A steady decline followed until the catches were reduced to zero. Within the same period the *O. niloticus* catches increased with a sudden rise in catches in 1998. The Pearson Correlation Coefficient of the relationship between *O. mortimeri* and *O. niloticus* catches were R = -0.73 (p < 0.05). The ratio of *O. niloticus* and *O. mortimeri* during the period increased exponentially (Figure 4 b). The relationship was described by equation (2).

O. niloticus/O. mortimeri catches = $1E\text{-}262e^{0.3025x} - 1$ ($R^2 = 0.86$; $p < 0.00$) (2)

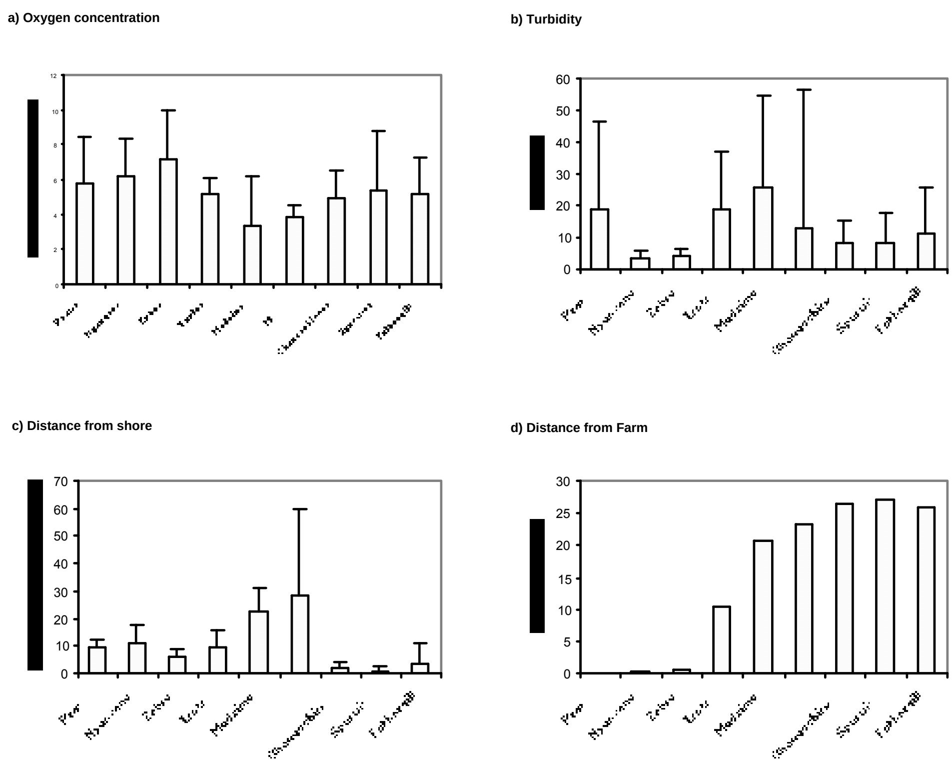


Figure 3. Environmental variables measured at each sampling site in the Sanyati Basin a) Oxygen concentration, b) turbidity, c) distance from shoreline and d) distance from fish farm.

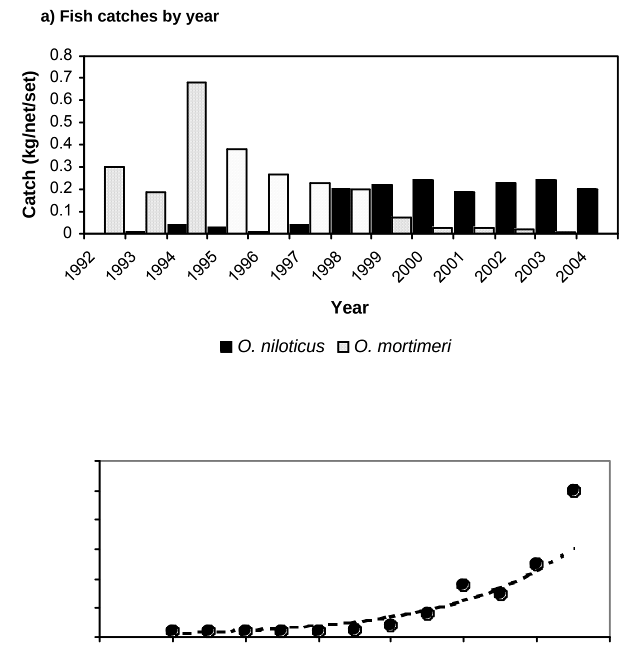


Figure 4. Trend in a) catches of *O. niloticus* and *O. mortimeri* and b) ratio of the catches at Lakeside sampling site between 1992 and 2004.

Discussion

Results showed that the introduced *O. niloticus* was replacing *O. mortimeri* in Lake Kariba. As proof, high catches of *O. mortimeri* were taken from areas where *O. niloticus* catches were low. Also there was high negative correlation between catches of these two species taken in the different areas ($R = -0.73$ ($p < 0.05$). Historical data from one site showed a similar trend where *O. niloticus* replaced *O. mortimeri* over time. At the Prawn farm site the mean ratio of catches of *O. mortimeri* and *O. niloticus* in the period September 1993 to June 1994 was 1:0.1 (Chifamba 1998). Yet in the 2003 & 2004 sampling not even a single specimen of *O. mortimeri* was caught in 8 settings of gill net at the same site. Furthermore, a more intensive Experimental fishing Programme (50 settings) came up with no specimen of *O. mortimeri* in 2004. This displacement was rapid and occurred over 10 years in the area close to the fish farm. Exponential models were used to describe the spatial and temporal rate of the change in the proportion of each species. Observations at Prawn Farm and Lakeside site indicated that it was not possible for the two species to coexist. Therefore it is likely that *O. niloticus* would replace *O. mortimeri* in areas that these two species currently coexists.

There are many similarities between *O. niloticus* and *O. mortimeri* that make competition probable. Competition for food is likely since diet similarities of two species in Lake Kariba have been reported (Chifamba 1998, Mhlanga 2000). The competitive exclusion principle predicts that if one of the species has competitive advantage over the other this leads to the extinction of the other. *O. niloticus* is believed to generally have high growth rates (Fryer & Iles, 1972) giving it competitive advantage. Competitive advantage of *O. niloticus* has now been demonstrated in Lake Kariba as in other water bodies.

In some water bodies that *O. niloticus* was introduced the species became important in the fishery replacing the native species as is happening in Lake Kariba (Goudwaard *et al.,* 2002, Schwank 1995, Balirwa 1992, Ogutu Ohwayo 1990, Vos *et a.l* 1990). Goudwaard *et al.,* (2002) provided evidence to support that species was the main cause for the disappearance of native tilapiine cichlids from Lake Victoria. In the Kafue River, a tributary of the Zambezi River, increase in *O. niloticus* was associated with a decline in native *Oreochromis andersonii* (Castelnau) and *Oreochromis macrochir* (Schwank 1995).

Both *O. niloticus* and *O. mortimeri* can tolerate a range of environmental conditions. Macrophyte abundance and type as well as bottom type were not correlated to catches. The only environmental factors correlated with *O. niloticus* catches were turbidity and distance from the shoreline. These areas were shallow and may have many suitable areas for breeding and nursery sites. In Lake Kyogo, where the species was also introduced spawning took place in areas with sandy or muddy bottoms (Twango 1995). In the same paper, *O. niloticus* was reported to thrive in water less than 10 deep as well as in areas of low oxygen levels. The tolerance to wide range of environments means that there are few boundaries to the distribution of the introduced species.

The study is a record of adverse impact of species introduction that threatens the native endemic fish species with extinction. It also demonstrates the importance of impact assessment before fish introductions to avoid loss of native species. In the case of Kariba it is very probable that the introduced species will continue to increase in the areas further from the farm whist displacing the indigenous species at a fast rate.

Acknowledgements

This research work was carried out using a Research Grants from the International Foundation for Science. Lake Kariba Fisheries Research Institute provided data from their Experimental Gill netting Programme. Sampling was done from the University Lake Kariba Research Station who provided staff, research vessel and other facilities. I am grateful to all my colleagues who assisted in the collection, and analysis of the data.

References

Balirwa J. S., 1992. The evolution of the fishery of *Oreochromis niloticus* (Piesces Cichlidae) in Lake Victoria. *Hydrobiologia* 232, 85-90.

Chifamba P. C., 1998. Status of *Oreochromis niloticus* in Lake Kariba, Zimbabwe following its escape from fish farms. In: *Stocking and introduction of fish.* (ed Cowx I G) pp. 267-273. Fishing News Books.

de Vos L. Snoeks L. & van den Audenaerde D. T. (1990) the effects of *Tilapia* introductions in Lake Luhondo, Rwanda. *Environmatal Biology of Fishes.* 27, 303-308.

Fryer G. and Iles T. D., 1972. *The cichlid fishes of the Great Lakes of Africa: Their biology and evolution.* Oliver and Boyd, London.

Goudswaard P. C., Witte F. & Katunzi E. F. B. (2002) The tilapiine fish stocks of Lake Victoria before and after the Nile perch upsurge. *J. Fish Biol.* 60

Karenge L. P., 1992. *Inshore fish population changes at Lakeside, Kariba between 1969 –1991.* M. Phil Thesis, University of Bergen p 54

Kenmuir D. H. S., 1984. Fish population changes in the Sanyati Basin , Lake Kariba, Zimbabwe. *S. Afr. J. Zool.* 19, 194-209.

Mhlanga L., 2000. The diet of five cichlid fish specie from Lake Kariba Zimbabwe. *Trans. Zimb. Scient. Assoc.* 74, 16-21.

Ogutu-Ohwayo R, 1990. The reduction of fish species diversity in Lakes Victoria and Koga (East Africa) following human exploitation and introduction of non-native fishes. *J. Fish Biol.* 32 (Suppl. A) 207-8).

Schwank E. J., 1995. The introduced *Oreochromis niloticus* is spreading on the Kafue floodplain, Zambia. *Hydrobiologia* 315, 143-147.

Skelton P. 1993. *A complete guide to the Freshwater fishes of Southern Africa*. Southern Book Publishers. p388.

Songore N., 2002. *Fish Diversity Development in Lake Kariba Zimbabwe form 1960 to 2001.* M.Phil. Thesis University of Bergen Norway. p84

Twango T., 1995. Impact of fish species introductions on the tilapias of Lakes Victoria and Kyoga. In: T. Pitcher & P. J. B. Hart, eds. *The impact of species changes in African lakes.* pp 45-55 Chapman & Hall, London.

Factors affecting the spatial and temporal distribution of Nile Perch in Lake Victoria

Albert Getabu

Kenya Marine and Fisheries Research Institute, P.O Box 1881, Kisumu, Kenya
Email:agetabu@yahoo.com

Abstact

The study was conducted to assess the stock of Nile perch in Lake Victoria and limnological factors controlling its distribution and abundance. Hydro acoustic data collected biannually from August 1987 to August 2001 is used to asses the spatial and temporal distribution of biomass and yields. Data on key environmental parameters collected concurrently with hydro acoustic data is used to describe the observed spatial and temporal distribution. The mean total acoustic biomass index was 2.17 x 10^6 t (range 1.0 x 10^6 – 2.5 x 10^6 t) corresponding to a standing crop of 31.0 t km^{-2} of which Nile perch constituted 59.3% of the total. The maximum sustainable yield from hydro acoustic surveys was 700, 000 t. The mean biomass index of Nile perch was 1.29 x 10^6 t which corresponds to a standing stock of 21.3 t km^{-2}. Spatial and temporal differences in the standing crop were observed between north and south and between shallow and deep water. Fish were mainly concentrated inshore. There were no commercially viable stocks in deeper offshore regions. Densities there were 3 - 4 times lower than inshore. Fish densities in the offshore region were relatively higher in August compared to February. Standing crops of August were correlated. There was an inconsistent pattern of correlation between those of August and February. The standing crop of fish correlated with dissolved oxygen concentration, chlorophyll-a, water depth and Secchi disk transparency. The implication of the research finsdings for management and food security for fishing communities is discussed.

Key words: Lake Victoria, Nile Perch, limnology, spatial and temporal distribution

Introduction

The Lake Victoria ecosystem is continuously changing in response to climate change and anthropogenic activities in its catchment. The latter provide material inputs into the lake in form of nutrients and pollutants, respectively. These have strong influence on the food web, fish abundance and yields. This study illustrates the effects of environmental variables on spatial and temporal distribution of *L. niloticus* in Lake Victoria. Data on dissolved oxygen and chlorophyll a concentrations, Secchi disc transparency, temperature, depth and conductivity was analyzed and interrelationships drawn with the standing crop of *L. niloticus* using backward stepwise regression.

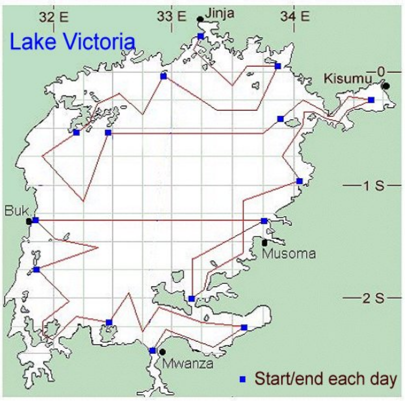
The spatial and temporal structure of fish populations in the water column, success of development and survival of egg and larval stages growth, mortality, fecundity, recruitment, migration and feeding are controlled by both short term and long-term environmental variations (Laevastu and Hayes, 1981; Laevastu, 1993, Merigoux *et al.*,, 2001). For example the seasonal stratification cycle of lakes has a strong influence on the distribution of materials such as nutrients, dissolved gases and solids as well as organisms in the water column. This in turn controls the natural productivity and fish distribution in the water column. Investigations of fisheries in the tropics seldom include adequate coverage of the limnology or lower trophic levels which support fish populations (Melack, 1976).

Different fish species are adapted to different ranges of environmental variables and hence show different spatial and temporal distributions in nature. A number of studies demonstrate how physical and chemical factors influence the distribution of fish in the water column. Observations by Rudstam and Magnuson (1985) show that vertical gradients of dissolved oxygen and temperature occur in lakes during stratification and that these are correlated with fish distributions. Suthers and Gee (1986) observed that the stickleback and the yellow perch, fishes closely related to *L. niloticus* exhibit diel and seasonal behavioural responses to changes in dissolved oxygen in water. They moved away from areas of low oxygen to areas of high oxygen. Studies in Lake Superior indicated that the smelt occurred in zones characterized by shallow warm waters (Heist and Swenson, 1983). Various other studies demonstrate strong relationships between primary production and fish yields (Melack, 1976; Bachman *et al.*,, 1996; Hakanson and Boulion, 2001).

This paper deals with the effect of environmental variables on the spatial and temporal distribution of *L. niloticus* in Lake Victoria. Data on temporal and spatial distribution of dissolved oxygen and chlorophyll *a* concentrations, Secchi disc transparency, temperature and conductivity is analyzed and variations between different zones of the lake are presented. Interrelationships are drawn between the standing crop of *L. niloticus* as the dependent variable and the environmental variables as the independent variables using backward stepwise regression. It is demonstrated that environmental factors affect the spatial and temporal distribution and abundance of *L. niloticus* in Lake Victoria. Environmental studies should therefore be integrated with stock assessment studies to provide an understanding on how they contribute to the distribution, abundance and behaviour of fish stocks.

Materials and methods

Acoustic data was collected using the Simrad EY 500 scientific echo-sounder with a hull mounted transducer of frequency 120 kHz and a 9° beam angle. Two cruise track designs were used Figs 1a and b. The first design in August 1999 concentrated inshore with fewer offshore transects. The second had uniform lake coverage and was used twice yeasrly between February 2000 and August 2001. Calibration was conducted twice at the start and end of the survey. Target strength experiments were conducted on live L. niloticus in an. octagonal cage, of side length 0.76 m wide by 1 m deep made of mosquito netting. Two standard targets: a 23 mm diameter copper sphere (TS = - 40.4 dB) between August 1999 and February 2001, and a 33.2 mm tungsten carbide sphere (TS = - 40.6 dB) in August 2001 were used. Limnological sampling was conducted twice a year in February and August – September during February 2000 – August 2001 using a Seabird E – 19 CTD profiler equipped with a Wetstar fluorometer. Sechi depth transparency was measured using a 30 cm diameter white disk.



(a)

(b)

Figure 1. Cruise tracks used for lake wide hydro-acoustic survey in August 1999 (a) and February 2000 - August 2001(b)

Results

The mean total acoustic biomass for all fish species was 2.17 x 10^6 tonnes (range: 1.9 x 10^6 - 2.5 x 10^6 tonnes), corresponding to a standing crop of 31.0 t km^{-2}, of which *L. niloticus* constituted of 59.3% of the total. The harvestable biomass from the total hydro-acoustic biomass estimate was 716 000 tonnes. The mean biomass for *L. niloticus* was 1.29 x 10^6 tonnes, which corresponds to a mean standing crop of 21.3 t km^{-2} which corresponds to an harvestable biomass of .426,000 t. There was no significant trend in the variation of the biomass over the sampling period Figure 2.

Nile perch was observed to concentrate in high densities particularly in inshore areas (Figures 3a – e; Figure 4) near river mouths and around major islands. Higher densities of juvenile Nile perch were also observed in same places. This makes the species liable to overexploitation by fishermen. Spatial and temporal differences in the standing crop were observed between shallow and deep water. There are no commercially viable stocks in the deeper offshore region as densities there are 3 - 4 times lower than those inshore.. Standing crops of February surveys were correlated as well as those of August surveys.

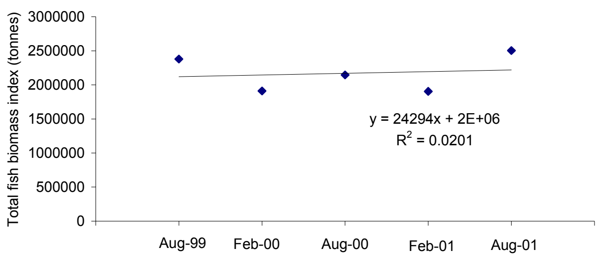
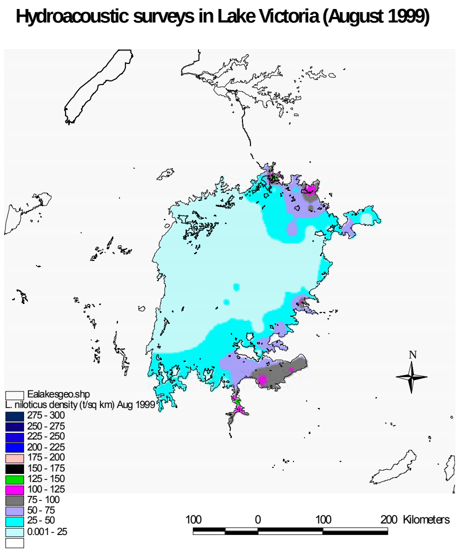
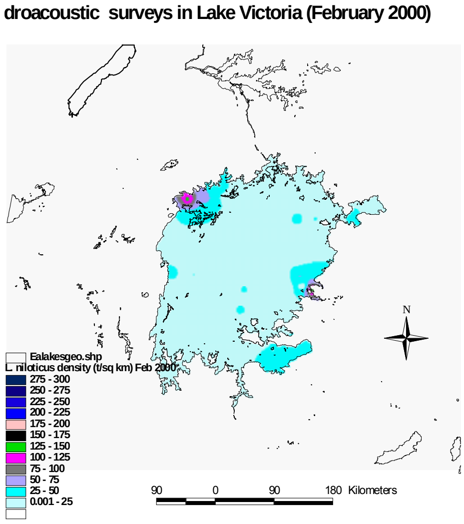


Figure 2. Total biomass of the Nile perch, *Lates niloticus* in Lake Victoria estimated during lake wide hydro acoustic surveys August 1999 – August 2001.



a.



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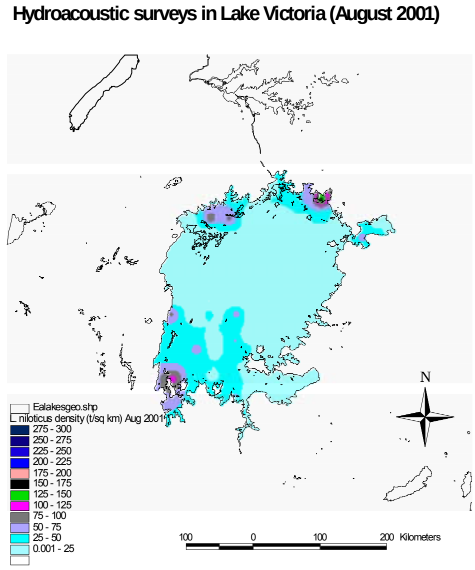
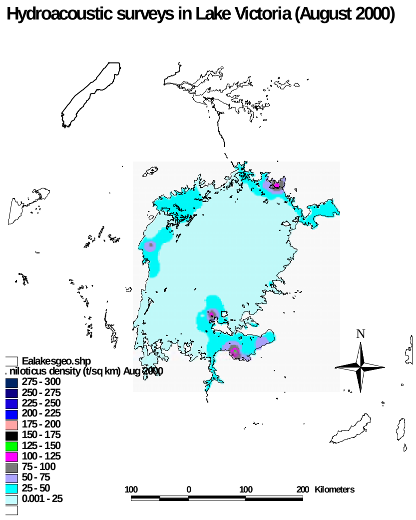
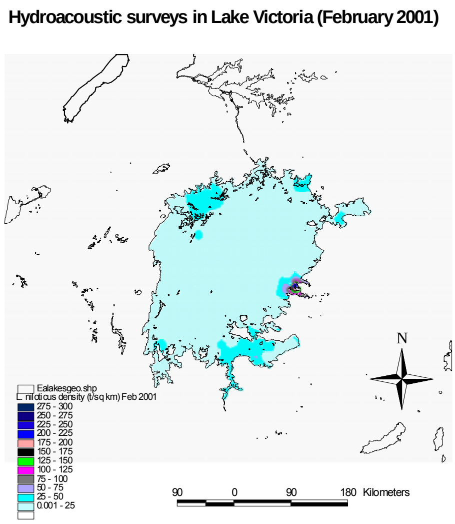


e.



c.



d.

Figure3a-e: Spatial and temporal distribution of the biomass of the Nile eprch lates niloticus in lake Victoria during periods August 1999-August 2001.

There was an inconsistent pattern of correlation between those of February and August. Environmental conditions in the lake were seasonal. In August, the water column was well mixed and in February, it was stratified. Correlations between limnological parameters and density (t km^{-2}) of *L. niloticus* varied from survey to survey. Different limnological parameters were correlated with density during different surveys (Table 1). The strongest positive correlation was between density and dissolved oxygen concentration(R^2 = 0.62, n = 45); followed by density with chlorophyll *a* concentration (R^2 = 0.57). The density of was negatively and significantly correlated with depth (R^2 = -0.75) and Secchi disc transparency (R^2 = -0.59). The proportions (R^2) of the density explained by the regression ranged from 0.595 to 0.608 indicating that besides variables examined, the other unexplained variation (39.2 – 40.5%) is due to other environmental, fishery related factors and measurement errors not considered in this study.

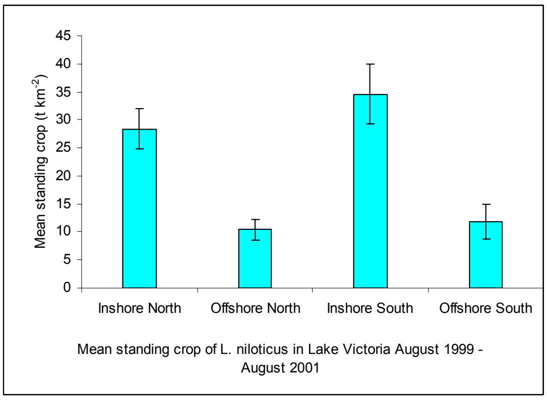


Figure 4. Differences in the densities of Nile perch between south and north inshore sectors respectively. The 1 degree latitude is used to define north and south of the lake.

The need for monitoring hydrographic conditions for use in interpreting the abundance and distribution of fish is emphasized.

Discussion

Echograms obtained during hydro acoustic surveys show that echo traces of *L. niloticus* were mainly located in mid-water layers above the thermocline during both February and August in the deep offshore region. This is the reason why bottom trawls deployed in deeper areas do not catch any *L. niloticus* (Mkumbo, 2002). In February, this could be due to hypoxic conditions in the hypolimnion coupled with the fact that some of its food preys constituting of haplochromines and *R. argentea* are located in mid-water. In August, the only reason why it was located in mid water could be feeding and may be migration since there were no hypoxic conditions at the bottom. The extent to which the fish was distributed was dependent on the position of the thermocline. The distribution in deep water was pelagic while it was observed in the entire water column, with most of it near the bottom inshore.

Table 1. Correlation between density of *Lates niloticus* and limnological parameters in Lake Victoria during February 2000.

		Density	Depth	Oxygen	Chlor. *a*	Secchi	Temp.	Cond.
Pearson Correlation	Density	1.000	-.752	.567	.621	-.594	.406	.159
	Depth	-.752	1.000	-.683	-.766	.790	-.547	-.420
	Oxygen	.567	-.683	1.000	.662	-.426	.444	.340
	Chlor. *a*	.621	-.766	.662	1.000	-.664	.536	.330
	Secchi.	-.594	.790	-.426	-.664	1.000	-.635	-.469
	Temp.	.406	-.547	.444	.536	-.635	1.000	.442
	Cond.	.159	-.420	.340	.330	-.469	.442	1.000
Sig. (1-tailed)	Density	.	.000	.000	.000	.000	.003	.148
	Depth	.000	.	.000	.000	.000	.000	.002
	Oxygen	.000	.000	.	.000	.002	.001	.011
	Chlor. *a*	.000	.000	.000	.	.000	.000	.013
	Secchi.	.000	.000	.002	.000	.	.000	.001
	Temp.	.003	.000	.001	.000	.000	.	.001
	Cond.	.148	.002	.011	.013	.001	.001	.

Key: Density - Standing crop of *L. niloticus* (t km^{-2}) Cond. - Conductivity (µS cm^{-1})
Chlor. a - Chlorophyll a concentration (µg l^{-1}) Temp. - Temperature (°C)
Secchi. - Secchi disc depth transparency (m) Significance level ≥ 0.5.

During February, when the offshore lake region was completely stratified, most of the *L. niloticus* were found in inshore areas which were not stratified and which were well oxygenated. Fish densities were much lower in the open water than during the mixing period in August. Earlier studies indicated that *L. niloticus* prefers areas with high oxygen concentrations (>5 mg l^{-1}) (Fish, 1956; Schofield and Chapman, 1999). It is for this reason that densities of both juvenile and adult *L. niloticus* were highest in the inshore areas where oxygen concentrations are high. The spatial and temporal distribution pattern around the islands in February was different from that of August periods. The islands where densities were higher than the surrounding areas in February were different from those of August.

This demonstrates the spatial and temporal variability of the limnological parameters during the two periods. The regions around the islands are shallower and well oxygenated than the surrounding deeper waters.. This is because such island areas constituted well-oxygenated refugia from the prevailing hypoxic conditions in the neighboring deeper waters. During the August surveys, the presence of high densities of *L. niloticus* around the islands could not be attributed to hypoxic conditions in the neighbouring deeper areas since the lake was well mixed. Other factors for example the distribution of sufficient prey organisms or spawning could contribute to the observed distribution. This demonstrates the importance of the islands in the biology of the species in the lake.

L. niloticus moves from the deeper deoxygenated parts of the lake into the well-aerated shallow regions and the riverine environment and are not accessible for the acoustic survey. This could be the reason why February surveys estimates are biased downwards. The movement of *L. niloticus* from deeper offshore waters into shallow inshore waters and into the areas surrounding islands exposes it to intense fishing pressure by the fishermen. In the Kenyan sector, where *L. niloticus* first established itself, stocks have declined drastically due to a combination of high fishing effort and deteriorating environmental conditions. The high fishing mortality and the deteriorating environmental conditions in the Kenyan sector are thought to have resulted in the reduction of the catch rates of *L. niloticus* in the Nyanza Gulf. As a result, the percentage contribution of *L. niloticus* to the overall demrsal biomass has reduced.

More recent changes in the fish community brought about by predation by *L. niloticus* and over fishing could aggravate the low oxygen conditions by altering nutrient flows and distributions within the lake (Kaufman, 1992). The species predated on algivorus haplochromines rendering the algal segment of the food web unutilized. This has resulted into deposition of organic matter at the bottom of the lake which contributes to anoxia in deeper layers. Thus, the current observed distribution of *L. niloticus* has been partly caused by the long-term ecological effects of the species itself in the lake.

Since the last four decades, there has been an increasing population density in the lakes catchment. This has increased anthropogenic inputs into the lake in form of nutrient rich organic matter from sewage disposal, agricultural runoff and industrial wastes. This also increased decomposing organic matter at the bottom thus enhancing eutrophication. These have had greater consequences in the distribution of *L. niloticus* in the shallower areas of the lake. The reduction of the size of the epilimnion i.e. the oxygenated layer means reduced area that can be occupied by *L. niloticus* and hence reduced biomass as the species keeps off low oxygen areas. The formation and persistence of low oxygen water mass creates the potential for rapid change in dissolved oxygen concentration changes near the bottom. If a distance of 10 m displaces the anoxic boundary vertically, it can travel tens of kilometers across the lake bottom because of it low bottom slope (Newell, 1960). Even near shore surface waters are rapidly deoxygenated during upwelling events along steep sided shorelines. For example, the Kenyan waters in the northeastern sector which have steep slopes have had increasing frequency of fish kills the majority of which are *L. niloticus* (Ochumba and Kibaara, 1989). Studies by Ochumba (1995) show the devastating effects anoxic conditions have had on *L. niloticus* in the lake. The species experienced high mortality as a result of upwelling hypoxic water on the eastern shoreline area to the south east of Mfangano Island. Observations during this study indicate that this area has low fish densities most of the time. This is because the shoreline of the area is steep and any upwelling has disastrous effects on fish inhabiting it. Further, southeast of the region where fish kills predominated by *L. niloticus* were found, is an area where always there are good catches of *L. niloticus*. This area is shallow and the slope of the shoreline is not steep. It constitutes the River Kuja delta. Such areas constitute the shallow areas and riverine environments where *L. niloticus* takes refuge during hypoxia in the deeper parts of the lake.

The distribution pattern of chlorophyll a concentration followed the same pattern as the distribution of *L. niloticus* density and dissolved oxygen concentration. High chlorophyll a concentration in the inshore areas and around the islands were correlated with high standing crops of *Lates niloticus*. Studies in other lakes and marine environment indicate that fish densities are high where chlorophyll a concentrations are high (Melack, 1976; Oglesby, 1977 and Bachman *et al.,,* 1996). High primary productivity supports high secondary production on which fish depend. Studies by Bachmann *et al.,*, (1996) indicate that total fish biomass per unit area was positively correlated with among other things chlorophyll a concentration in Florida lakes. A number of studies support the expectation that an increase in productivity at the base of the food web in a lake should translate into an increase in the abundance of fish (Melack, 1976; Oglesby, 1977; Downing *et al.,,* 1990; Lowe-McConnell, 1997).

Observations during this study indicated that total fish standing crop index was highest in the inshore Tanzanian sector of Lake Victoria. This implies that the southern part of the lake is more productive than the northern part. This is in agreement with limnological findings showing more concentrated nutrients in the southern sector of the lake. This is thought to be due to the southward pile up of the nutrients, resulting from wind action against a downward–northern tilt of the thermocline (Kitaka, 1971).

The regression analysis of data produced by the current research has indicated that fish densities in the lake are linearly and positively related to temperature. Higher fish densities were observed in areas where temperature was high, i.e. in the shallow inshore where the mixing makes the temperature uniform throughout the water column.

Another factor, which was linearly related to the density of *L. niloticus,* was conductivity. Conductivity was highest in inshore areas. This could be due to a number of reasons for example due to the ease of the mixing of the entire water column, a factor that brings up bottom deposits into the water column, thus increasing the conductivity. The other factor probably responsible for high conductivity in inshore areas is anthropogenic discharges from the major rivers draining into Lake Victoria. Erosion of the catchment enhanced by poor agricultural practices deposits eroded material rich in electrolytes, which increase conductivity in the inshore areas. Hydrodynamic processes (currents and waves) then redistribute the materials within the inshore areas thus increasing the conductivity. Some of the electrolytes are important plant nutrients. These increase primary productivity in inshore areas, thus also improving the food resource base for fish and account for the observed high densities of *L. niloticus* observed there. Therefore, density is only associated with conductivity and not caused by it-the real cause is the abundant food supply resulting from the high nutrient concentrations in inshore areas.

Conclusion

The concentration of Nile perch in certain areas during different seasons makes it liable to overexploitation by fishermen once they obtain the knowledge about the distribution. This means that conservation and management measures should be put in place to protect the species during periods it concentrates in such areas. It is further demonstrated that environmental factors affect the spatial and temporal distribution and abundance of *L. niloticus.* Environmental studies should therefore be integrated with stock assessment studies to provide an understanding on how they contribute to the distribution and abundance and behaviour of fish stocks.

References

Bachman, R.W., L.J. Bradly, D.D. Fox, M.H. Hoyer, L.A., Bull and D.E., Canfield Jr, (1996). Relations between trophic state and fish in Florida (USA) lakes. *Can. J. Fish. Aquat. Sci*. 53, 842 - 855.

by measuring echo abundance of o-group fish. *FiskDir. Skr. Ser. Havunders,* 13, 47 -

Downing, J.A., C. Plante and S. Lalonde, 1990. Fish production correlated with primary productivity, not the morphoedaphic index. *Can. J. Fish. Aquat. Sci*., 47: 1929 - 1936.

Dragesund, O. and S. Olsen (1965). On the possibility of estimating year-class strength

Fish, G.R. (1956). Some aspects of the respiration of six species of fish from Uganda. *J. Exprt. Biol*. 33: 186-195.

Fish, G.R. (1957). A seiche movement and its effect on the hydrology of Lake Victoria. Fish.

Hakanson, L. and V. Boulion (2001). Regularities in primary production, secchi depth and fish yield and a new system to define trophic and humic state indices for lake ecosystems. *Int. Rev. Hydrobiol*., 86 (1), 23 - 62.

Hecky, R.E., F.W.B. Bugenyi, P. Ochumba, J.F. Talling, R. Mugidde, M. Gophen and L. Kaufman (1994). Deoxygenation of deep Lake Victoria. *Limnol. & Oceanogr.* 39(6) 1476 - 1481.

Heist, B.G. and W.A. Swenson (1983). Distribution and abundance of rainbow smelt in western Lake Superior as determined from acoustic sampling. *J. Great Lakes Res*., 9(3), 343 - 353.

Kaufman, L. (1992). Catastrophic change in species-rich freshwater ecosystems: the lessons of Lake Victoria. *Bioscience* 42(11), 846-858.

Kitaka, G.E.G. (1971). An instance of cyclonic up welling in the southern offshore waters of Lake Victoria. *Afr. J. Trop. Hydrobiol. Fish*. (2)1, 85-92.

Laevastu, T. (1993). Marine climate, weather and fisheries. Fishing News books, Hartnolls Ltd., Bodwin, Cornwall, 204pp.

Low-McConnell, R.L. (1997). EAFRO and after: a guide to Key events affecting fish communities in Lake Victoria (East Africa). *South J. Sci*. 93: 570 - 574.

Melack, J.M. (1976). Primary productivity and fish yields in tropical lakes. *Trans. Amer. Fish. Soc*. 105 (5), 575 - 580.

Merigoux, S., S. Doledec and B. Statzner (2001). Species traits in relation to habitatvariability and state: neotropical juvenile fish in floodplain creeks. *Freshwat. Biol*. 46, 1252 - 1267.

Mugidde, R. (1993). The increase of phytoplankton primary productivity and biomass in Lake Victoria (Uganda). *Verh. Verein. Limnol*. 25, 846 - 849.

Newell, B.S. (1960). The hydrology of Lake Victoria. *Hydrobiologia* 15, 363 - 383.

Ochumba, P.B.O. (1995). Limnological changes in Lake Victoria since the Nile perchintroduction. Pages 33 - 43. *In* T.J. Pitcher and P.J.B. Hart (Eds.) The impacts of species changes in African lakes. Chapman and Hall Publishers, Fish and Fisheries Series 8, 601pp.

Ochumba, P.B.O. and D.I. Kibaara (1989). Observation on the blue green algal blooms in the open waters of Lake Victoria, Kenya. *Afr. J. Ecol.*27: 23 –24.

Oglesby, R.T. (1977). Relationships of fish yield to lake phytoplankton production and morphoedaphic factors. *J. Fish. Res. Bd. Can*., 34, 2271 - 2279.

Oguttu-Ohwayo, R. (1995). Diversity and stability of fish stocks in Lakes Victoria and Kyoga. Pages 59 - 82. *In* T.J. Pitcher and P.J.B. Hart (Eds.) The impacts of species changes in African lakes. Chapman and Hall Publishers, Fish and Fisheries Series 8, 601pp.

Rudstam, L.G. and J.J. Magnuson (1985). Predicting the vertical distribution of fish populations: Analysis of Cisco, *Coregonus artedii*, and yellow perch, *Perca*

Schofield, P.J. and L.J. Chapman (1999). Interaction between Nile perch, *Lates niloticus* and other fishes in Lake Nabugabo, Uganda. *Env. Biol. Fish*. 55, 343 - 358.

Suthers I.M. and J.H. Gee (1986). Role of hypoxia in limiting diel spring and summer distribution of juvenile yellow perch, (*Perca flavescens*) in a Prairie marsh. *Can. J. Fish. Aquat. Sci*., 43: 1562-1570.

Talling, J.F. (1957). Some observations on the stratification of Lake Victoria. *Limnol* and *Oceanogr*. 3 (1), 213 - 221.

Talling, J.F. (1966). The annual cycle of stratification and phytoplankton growth in Lake Victoria (East Africa). *Int. Rev. ges. Hydrobiol*., 51 (4), 545 - 621.

Status and root causes of biodiversity loss in the eastern Rift Valley lakes, Kenya

Nathan N. Gichuki[1], Helida A. Oyieke[2] and Taita Terer[2]

[1]University of Nairobi, Department of Zoology, P.O. Box 30197, Nairobi, Kenya
[2]National Museums of Kenya, Center for Biodiversity, P.O. Box 40658-00100, Nairobi, Kenya
Email: E-mail-ngichuki@uonbi.ac.ke

Abstract

The Eastern Rift Valley traverses Ethiopia, Kenya and Tanzania and constitutes a major geographical feature in eastern Africa. It forms an internal drainage basin with wet highlands on either side and a chain of 20 lakes on its floor. The lakes support globally important biodiversity both individually and as a chain. Recognizing its biodiversity significance, the seriousness of the threats and the need to stop biodiversity loss, Ethiopia, Kenya and Tanzania developed a regional project with the support of Global Environment Facility between 1999 and 2003. The development objective of the project was to ensure conservation and sustainable use of biodiversity in the eastern rift valley lakes. To achieve this goal, all-inclusive consultations and problem analysis both at national and site levels was conducted in Kenya. A series of stakeholder consultation workshops and informal meetings were held in Nairobi and at Lakes Turkana, Bogoria and Magadi for northern, central and southern rift valley lakes respectively. Expert working groups were also set up to gather, analyse and synthesize data and information on biodiversity status and trends, human impacts, constraints to conservation and sustainable use, community/ institutional participation and capacity building needs, and the impacts of hydrological and climate changes on biodiversity. The above analysis revealed that the biodiversity of the lakes is threatened by watershed degradation, soil erosion, pollution and unsustainable exploitation of biological and physical resources. In this paper the root causes of biodiversity loss in the Kenya's Rift Valley Lakes are identified using the information available from the authors' own studies in the country as well as from national consultation process and thematic group reviews. The paper also provides an analysis of issues pertaining the management of biodiversity in trans-boundary areas as well as underscoring the significance of a holistic approach to lake basin management, including regional co-operation among states.

Key words: Biodiversity, lakes, rift valley, Kenya

Introduction

The Eastern Rift Valley, which traverses Ethiopia, Kenya and Tanzania constitutes a major geographical feature. Although most of the valley has characteristic arid and semi- arid climatic conditions, it forms an internal drainage basin with a chain of 20 lakes, which are fed by streams, springs and underground water seepage from wet highlands on the Eastern and western shoulders of the valley. The system of lakes and the floodplains of affluent streams and rivers support globally important biodiversity both individually and as a chain.

In spite of the importance of Eastern Rift Valley Lakes in conservation of biodiversity, including valuable genetic resources and in supporting livelihoods and economies of the local communities, there has been growing pressure which can be linked to global climatic changes, human population growth, patterns of resource exploitation and policy response or policy failures. Studies on the lakes have been scattered both in time and space and therefore there is no single document that brings together and analyses the findings of those studies. This paper brings together current information on the status and root causes of biodiversity loss in Kenya's Eastern Rift Valley Lakes.

Materials and methods

Study area and scope

The Rift Valley system in Kenya extends about 900 km from the Turkana depression in the north to the Magadi- Natron depression in the south (Figure 1). It contains a chain of fresh water and saline lakes of different origin, morphometric configuration and productivity status. Its drainage system covers a total area of 126,910 km^2 and lies within 4º30' N to 2º 00' S and 35º 30' E to 37º E. The major lakes comprise Turkana, Baringo, Bogoria, Nakuru, Elementaita, Naivasha and Magadi while some of the minor ones include Kamnarok, Kwenia, Kabongo, Kijirtit, Solai, Ol Bollossat and Logipi. The variability in topography, geology and climatic conditions as well as the volcanic landscape in which the lakes occur impart to each lake its unique physical, chemical and biological characteristics. It is thought that most the current lakes are remnants of larger lakes that occupied the floor of the valley during the last pluvial period (Beadle 1974, Hopson 1982).

Data collection and analysis

In this study, several approaches were used to collect and analyse data. The first approach was to review the state of knowledge of Rift valley lakes in Kenya by gathering, documenting and collating existing information on biodiversity values, threats, constraints to conservation and management, and the existing institutional and policy frameworks reviewed. In order to execute the task, five technical teams each consisting of five experts were established in June 1999. The team gathered information on: Biodiversity status and trends (Mutangah *et al.,,* 2000), Impacts of human activities on landscape and natural resources (Ndede *et al.,,* 2000), community/institutional participation and capacity building needs (Ogutu *et al.,,* 2000), constraints to conservation and sustainable use (Koyo *et al.,,* 2000) and impacts of hydrological and climate changes on biodiversity of the Kenyan Rift Valley Lakes (Ondieki *et al.,,* 2000).

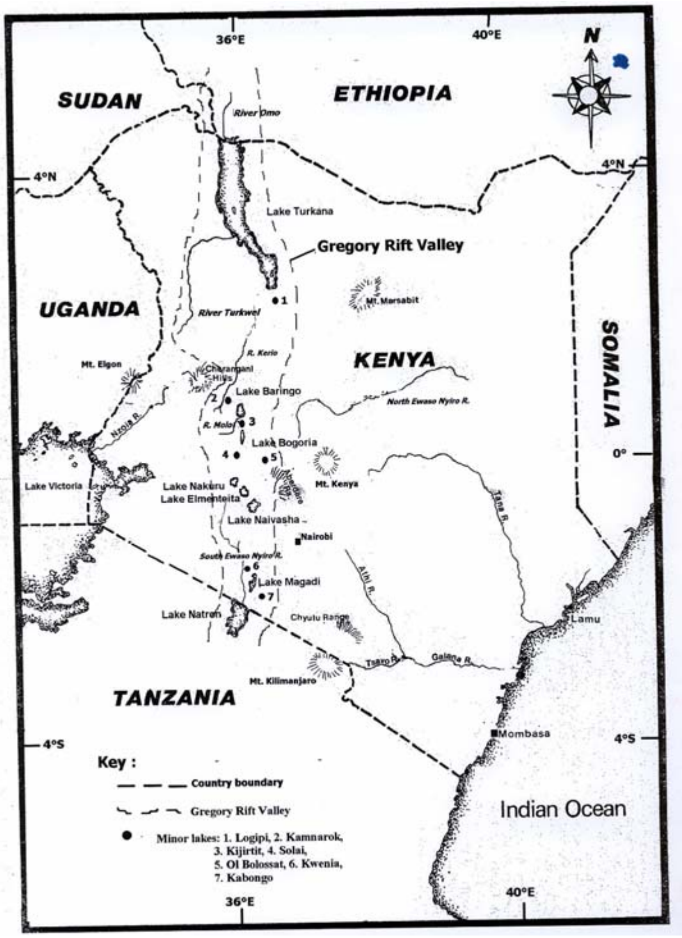


Figure 1. Hydrographic map showing lakes of the Kenyan Rift Valley.

Running parallel to the literature review activities were national stakeholder consultation workshops (Gichuki *et al.,*1999a, Gichuki *et al.,*1999b, Njuguna *et al.,*1999: unpublished data). Three workshops were conducted in the Northern Rift (to cover Lake Turkana & Logipi), Central Rift (to cover Lakes Bogoria, Baringo, Nakuru, Naivasha and minor Lakes) and South Rift (to cover Lakes Magadi and Natron including Ewaso Ngiro South river basin). The main aim of these workshops were to involve local communities and other stakeholders in identifying socio- economic and conservation issues relating to the lakes in the regions (Njuguna *et al.,,* 2000).

The analysis and synthesis of the information gathered from June 1999 to January 2000 through literature review, national consultations and stakeholder workshops identified the lakes' biodiversity values, threats, constraints to conservation, existing institutional framework and national policies that impact on the lakes. Conservation and management initiatives were evaluated site by site for all major and some minor Rift Valley lakes. This site-specific approach permitted the determination of baseline investments that had been made (or were being made) in Rift Valley lakes (Njuguna *et al.,* 2000, NMK 2001). The whole project was coordinated by a national planning and management committee drawn from key institutional stakeholders of biodiversity resources in the Rift Valley Lakes of Kenya.

The development of a strategic action programme (SAP) entailed converting identified biodiversity problems and constraints, research gaps, root causes of biodiversity loss and community development priorities into actionable activities and achievable outputs. This was carried out in 2000 by a task force of experts at regional level (Njuguna S. 2000) and national level (Ndede and Bennun 2000). This paper outlines the status of biodiversity, root causes of biodiversity loss and policy options needed to address them in the eastern Rift Valley lakes in Kenya.

Results

Status of biodiversity

The diversity of landforms and climatic conditions led to the evolution a wide range of habitats and diverse flora and fauna in the rift valley lakes. The biological diversity of Rift Valley lakes and wetlands comprises: microorganisms, macroinvertebrates, vertebrates and macrophytes. The status and trends of that biodiversity were reviewed by Gichuki & Terer (1999) and Mutangah *et al.,* (2000).

Rift Valley Lakes support unique micro-biodiversity whose composition and structure varies with water quality (freshwater or alkaline water) and other physico-chemical characteristics of each individual lake or wetland. Studies of soda lakes Bogoria, Elmentaita and Nakuru have showed that the lakes have a population of over 10^6 bacteria per millilitre of water (Mwatha 1991, Jones et al 1994). The blue-green algae (Spirulina plantensis) is the most abundant microflora. In Lake Magadi, photosynthetic bacteria, such as Ectothiorhodospira contribute to primary productivity. Grant & Mwatha (1989) and (Mwaura 1981) found nitrogen – fixing bacteria being associated with roots of papyrus and Salvinia molesta. Phytoplankton community in Lakes Turkana, Baringo and Naivasha is dominated by blue–green algae while the zooplankton community is dominated by rotifers, crustaceans, protozoa, copepods and cladocerans (Mavuti 1983 & 1992, Kalliquest 1987). The phototrophic eukaryotes (diatoms) are common in the soda lakes and in fact contribute significantly to the productivity in Lakes Elementaita and Nakuru.

Macroinvertebrate animals also form a significant part of aquatic biodiversity in the rift valley lakes. Lake Naivasha is poor in macro-invertebrate diversity compared to other fresh water tropical lakes (Clark *et al.,* 1989). The species considered threatened in Lake Naivasha include seven species of Oligochaetes. Similarly, some insect groups such as Dipteran *Limnochironomus sp*., Hemipteran *Corixa mirandilla*, Tricopteran *Ecnomus menneli* and dragon flies (Order Odonata) have become fairly uncommon during the last 30 years. Lake Turkana is known to have sponges, bryozoan species, gastropod bivalves, astracods, hydracarines and annelids (Cohen 1986). Hughes & Hughes (1992) have indicated that the lake also harbours *Bellamya unicolor* and Cl*eopatra pirothi*. Vareschi (1992) reported the abundance of pollution resistant chironomid dipterans in Lakes Bogoria and Nakuru. In Lake Oloiden, Lox (1977) and Clarke et al (1989) reported the presence of tubificids in addition to five orders of insects. In contrast, Lakes Magadi and Elementaita and most of the minor lakes have limited macroinvertebrate information.

The rift valley lakes host a small number of vertebrate animals, including reptiles, fishes, amphibians, birds and mammals. Lake Turkana has a population of up to 14,000 Nile crocodile, *Crocodylus niloticus*, Monitor Lizard *Varamus niloticus* and two other endemic species of herptiles: Turkana Hinged Terrapin *Pelusios broadleyi* and unnamed frog (Bufo sp.) Other species include the Nile Soft-shelled Turtle *Trionyx tringuis* which is site specific and the Helmeted Terrapin *Pelomedusa subrufa* whose status is unknown (Mutangah *et al.,* 2000). Other lakes are relatively poor in reptile species although crocodiles exist in lakes Baringo and Kamnarok in Kerio Valley.

The total fish species richness of the Rift valley lakes stands at 64 species, 48 of which occur in Lake Turkana. The major fish groups are groups Cichlids (dominated by genus *Tilapia,* Haplochromines, Siluriforms (the catfishes), Perciformis (Perches) and Cypriniformis (the carps). So far, fish species have been recorded in 8 lakes, namely, Turkana, Baringo, Naivasha, Kamnarok, Kijirtit, Nakuru, Oloidien and Magadi. These fishes occur in diverse environments, ranging from freshwater to hyper saline conditions. The cichlids are the most successful fish species and form an important fishery resource in the fresh water lakes.

Kenya's Rift Valley is home to over 500 species of birds, 20% of which are water birds while the rest inhabit marshes, woodlands and wooded grasslands adjacent to the lakes. There is no bird species that is endemic to Kenya's Rift Valley lakes but the lakes support four globally threatened species namely: Lesser Flamingos *Phoeniconaias minor*, Madagascar Squacco Heron *Ardeola idae*, Lesser Kestrel *Falco naumanni* and Grey-crested Helmet Shrike *Prionops poliolophus*. The lakes support over two million Lesser flamingos but their numbers show conspicuous variations. Githaiga (2004) linked occasional die offs of flamingos to food shortage, lack of drinking water, bacterial toxicity and heavy metals.

The four species of mammals that regularly inhabit the wetlands of the Rift Valley lakes are the African Hippopotamus *Hippopotamus amphibious*, Giant marsh rat *Mycocaster coypu*, Black-necked Otter and Marsh Mongoose. Hippos are abundant in Lakes Naivasha and Baringo. These large mammals graze on the lake shores and influence plant growth and regeneration. The Giant marsh rat was introduced in Lake Naivasha in 1930s to control the marsh vegetation but it has since spread to many other water bodies in central Kenya.

The soda lakes have no emergent vegetation and their shoreline is either muddy or rocky and the adjacent land is covered by salt tolerant grass species: *Paspalidium geminatum* and *Sporobolus spicatus.* These grasses cover the seasonally exposed shallows and provide important nurseries for fish. In Lake Turkana, *Salvadora persica* bushes cover central and south Islands while the western shore, especially in Ferguson's Gulf, is covered by *Prosopis juliflora,* an invasive alien shrub. *Cyperus papyrus* and *Typha domigensis* dominate the mouths of affluent rivers, especially in lakes Naivasha and Baringo. The Nile cabbage *Pistia stratiotes,* Water hyacinth *Eichornia crassipes* and water lilies *Nyamphaea caerulea* float on the surface of small bays. With 108 species of plants, Lake Naivasha has the riches aquatic plant community, whose structure and function have been well studied (Gaudet 1979). The main submerged plants in the lake are *Najas* p*ectinata*, *Ceratophyllum demersum* and *Potamogeton* species. The invasive water fern *Salvinia molesta* has invaded some the freshwater bodies in the rift valley and blocked the mouths of affluent rivers

Proximate causes of biodiversity loss

The biodiversity of the Rift Valley Lakes in Kenya faces threats from a wide range of sources but most can be attributed to human activities within the lake basins and their catchment areas (Oyieke (2000). The primary impact of human activities is to magnify the effects of natural processes, such as soil erosion and deposition, water pollution with sediments and introduce new threats to biodiversity. The root causes to biodiversity loss in the Kenyan part of the Rift Valley can be attributed to a number of factors.

Degradation of land: The degradation of the land can be linked to rapid human population growth and its increasing pressure on natural resources. The 1999 population census showed an upward trend in all districts within the Rift Valley Province (GoK 1999). In the arid north, Turkana district had a population density of 4 persons per Km^2 and a mean annual growth rate of 2.5%. In the wet central Rift Valley, Nakuru district had a population density of 191 persons per Km^2 and mean annual growth rate of 4.9%. In semi-arid Kajiado District, population density was 21 persons per Km2 with mean annual growth rate of 3.6%. These population data provide a clear indication of magnitude of the threat to biodiversity in different parts of the Rift Valley. In central rift, the greatest threat to native biodiversity has been the excision of natural forest and woodland for agriculture and expansion of human settlement. In the dry Turkana, Baringo and Kajiado districts, soil erosion due to overgrazing and clear-felling of trees in water catchment areas are serious threats to biodiversity of the Rift Valley Lakes. Soil erosion and its subsequent deposition are also serious problems in the deltas of Rivers Kerio, Turkwell, and Omo. Damage to Doum Palm trees along river Turkwell and on the western shores of Lake Turkana for cottage industry is a serious emerging threat to fish breeding habitats in Lake Turkana (Njuguna *et al.,* 1999: unpublished data). Deforestation and intensification of agriculture in Nguruman escarpment is affecting the quantity and quality of water supply to River Ewaso Ngiro and Lake Natron (Gichuki *et al.,,* 1998).

Pollution of aquatic systems: Pollution levels in rivers and lakes in the rift valley are strongly linked to the density of human settlements, commercial, industrial and agricultural activities. These activities are concentrated in central Rift Valley, where land use is most intensive. Pollution from agricultural chemicals such as fertilizers, pesticides as well as industrial and domestic effluent threatens the biodiversity of Lakes Nakuru, Baringo and Naivasha. Waste material from mining activities and agricultural chemicals in Kerio Valley and Omo valley in Ethiopia as well as growing human settlements associated with fishing, cottage industries, trade and tourism on the shores threatens biodiversity and fisheries in Lake Turkana (Abebe 1999, pers. obsev.). In south rift, Lake Magadi is polluted by waste material generated from soda ash factory and residential houses in the town. Agricultural chemicals, animal waste and sewage from Nguruman market flows into River Ewaso Ngiro and threatens biodiversity in Shompole swamp and Lake Natron (Gichuki *et al.,,* 2000).

Unsustainable exploitation of water and biological resources: Most of Kenya's Rift Valley semi-arid and hence freshwater is scarce. Except for Lake Naivasha, which is fresh, the other lakes contain water which is not suitable for domestic and farm use. Water abstraction from the Lake Naivasha and River Malewa in the catchment areas exceeds sustainable levels. This affects water balance and reduces water levels in the lake. Excessive extraction of water from Rivers Molo and Perkera has also serious negative effects on water supply to Lake Baringo. Similarly, water abstraction from River Ewaseges in Subukia has reduced water input into Lake Bogoria. Water supply into Lake Nakuru has also declined significantly due to intensification of agriculture in the Mau catchment. Diversion of water for irrigation along the course of rivers Omo and Kerio has drastically reduced water supply into Lake Turkana. Horticultural farming on Nguruman escarpment has expanded considerably and water use has increased almost fourfold since the 1970s. This has serious negative effects on biodiversity and availability of pasture for livestock in Shompole swamp. It also threatens the largest breeding habitat of flamingos in East Africa: Lake Natron (Gichuki *et al.,*2001). Inter-basin water transfers, as has been done from Naivasha basin to Nakuru basin, has significant ecological impacts on the biodiversity of the lakes affected. Construction dams on rivers flowing into the lakes, for instance, Turkwell Dam on River Turkwell and several dams on rivers flowing into Lake Baringo has caused major changes in the water flow regimes and consequently changes in biodiversity.

Introduction of invasive species: These are a number of exotic plants and animal species that have been introduced into the Rift Valley in order to meet social, cultural and economic needs of the people. The introduced species, however, have proliferated and are now a threat to indigenous biodiversity. For instance, exotic weeds such as the water fern *Salvinia molesta* and Water hyacinth *Eichornia crassipes* have at different periods colonized Lake Naivasha*.* Water pollution has encouraged the proliferation of aquatic weeds especially in Lake Naivasha (Gichuki *et al.,* 1997). The introduction of Louisiana Grayfish and Coypu rat (*Coypu coypu*) in the Rift Valley lakes in the 1930s has also had negative impact on ecological balance between species.

Ineffective conservation and management: The main constraints to biodiversity conservation and sustainable use in Kenya's Rift Valley lakes are infective policy for biodiversity conservation, poor knowledge base and application of the ecosystem principles in conservation, uncoordinated information management, inadequate mechanism for monitoring biodiversity, inadequate system of biological reserves for conservation and shortage of resources, including qualified personnel responsible for research and monitoring. These constraints reflect the general situation all along the Rift Valley and in Kenya as a whole.

Conclusion and recommendations

The Rift Valley Lakes of Eastern Africa harbour a vast array of organisms and complex ecological interactions. The lakes form a system of sites that individually and collectively support abundant physical resources and globally important biodiversity. The Eastern Rift Valley lakes are a global heritage that offers the last viable examples of past global ecological and evolutionally processes that have lead to the current distribution and functional patterns of aquatic systems in the eastern Africa. The existing biotic communities, lacustrine sediments and fossils provide a unique window to the ecological and evolutionary history of eastern Africa. Therefore there is urgent need for regional and national conservation efforts to safeguard the biodiversity values of these wetlands

The biodiversity of the lakes also holds potential global and national benefits. The abundance of heterotrophic bacteria and algae in the soda lakes, for instance, has recently attracted bioprospecting activities for pharmaceutical purposes and development in biotechnology. While these activities promise economic benefits in future, those benefits should be shared with the local communities, who are the custodians of these wetlands. Furthermore, the rift valley lakes occur in an arid or semiarid environment where local communities have limited livelihood options. Most of them are engaged in subsistence agriculture, traditional pastoralism and harvesting of biological resources, such as fisheries and plant products. Faced with the reality of a rapidly growing population and increasing poverty, and the negative consequences of natural resource exploitation, alternative livelihood options should be identified and developed as a matter of priority.

In conclusion, this study has established the key components of biodiversity and the factors that threaten that biodiversity in Kenyan Rift Valley Lakes. These root causes biodiversity loss are population pressure, unsustainable use of natural resources, economic policies that do not recognise the true value of biodiversity, inadequate knowledge and its application, and weak legal and institutional systems. Unless each one of these root causes is deliberately and effectively addressed, the valuable biodiversity of the rift valley lakes in East Africa will continue to be lost. This situation has serious consequences on social and economic development of communities that live and depend on these aquatic systems. The governments of Ethiopia, Kenya and Tanzania as well as local communities and development partners have the responsibility to ensure that this globally important biodiversity is conserved and used in a sustainable manner.

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References

Abebe Y.D., 1999. Cross-border issues in Lake Turkana and Omo River basin. *In.* Njuguna P.K. ed. *Proceedings of Lake Turkana Stakeholder Workshop*, pp 9-23. National Museums of Kenya, Nairobi.

Beadle L.C., 1932. Scientific Results of Cambridge Expedition to the E. African lakes 1930-31: the waters of some E. African lakes in relation to their fauna and flora. *J. Lin. Afr.* Soc. Zoology 38:157-211.

Beadle L.C., 1974. *The inland waters of tropical Africa*. Longman Group Ltd.

Butzer K.W., 1971. *Recent history of an Ethiopian delta.* University of Chicago Press.

Clark F. Beeby A Kirby P., 1989). A study of the macro invertebrate of lakes Naivasha, Olidien and Sonachi. Kenya Revenue Authority, *Hydrobiol.* Tropic. 22:21-23

Cohen A.S., 1986. Distribution and faunal associations of benthic invertebrates of Lake Turkana, Kenya. *Hydrobiologia* 141: 179-197.

Florenzano G., Silli C., Pelosi E. & Vincenzini M., 1985. *Cyanospira rippkae* and *Cyanospira capsulate*: a new filamentous cyanobacteria from Lake Magadi, (Kenya). *Arch, Microbiol.* 140: 301-306.

Gaudet J.J. 1979. Seasonal changes in nutrients in tropical swamp water. *J. Ecology*, 67: 953-981.

Gichuki N.N., Wakanene K. & Ndiritu G.G., 1997. *Biodiversity Profile of Nakuru District and its environs, Kenya.* Institutional Support for Protection of East African Biodiversity (UNO/RAF/006/GEF) National Museums of Kenya, Nairobi.

Gichuki, N. N., H. A. Oyieke & G. G. Ndiritu, 1998. Assessment and monitoring of wetlands for conservation and development in dry lands: A case study of Kajiado District, Kenya. *In.* C. M. Finlayson, Davidson N. C. & Stevenson N. J., eds. *Wetland Inventory, Assessment and Monitoring: Practical Techniques and Identification of Major Issues*. pp.166:99-107. Supervising Scientist Report, Brisbane, Australia.

Githaiga J.M., Mavuti K. & Karanja W., 1999. *Effects of hydrological regimes on Lesser flamingo Phoeniconais minor movements in the saline lakes of Kenya*. Proc. Intern. Conf. Shallow Tropical Lakes. KWSTI, Naivasha, Kenya.

Government of Kenya, 1989. *National Population Census 1989. Ministry of Planning and National Development*. Central Bureau of Statistics. Government Printer, Nairobi.

Government of Kenya, 1999. *Population Census 1999. Ministry of Planning and National Development, Central Bureau of Statistics.* Government Printers, Nairobi.

Grant W.D. & Mwatha W.E., 1989. Bacteria from alkaline, saline environments. *In. Recent Advances in Microbial Ecology*, Japan Scientific Press, pp. 64-67.

Hopson A.J., 1982. *Lake Turkana*. A report on the findings of Lake Turkana Project.

Howard, G. W. 1992. Introduction. *In.* G.L Kamukala, S.A. Crafter eds. *Wetlands of Tanzania* pp.1-6. IUCN, Gland, Switzerland.

Hughes H.P. and Hughes J.S., 1992. *Directory of African Wetlands*. IUCN, UNEP and World Conservation Monitoring Centre.

Jones B.J., Grant W.D., Collind, & Mwatha W. E., 1994. Alkaliphiles: Diversity and identification. *In.* F.G. Priest *et al.,*, eds. *Bacterial Diversity and Systematics* Plenum Press New York.

Kallquist T., 1987. *Primary production and Phytoplankton in Lake Baringo and Lake Naivasha, Kenya.* Norwegian Institute for Water Research Report.

Koyo A., Kariuki J., Kaburu H., Mungai I. P. & Gichuki M.C., 2000. Constraints to conservation and sustainable use of biodiversity in Rift valley lakes in Kenya: *In. Problem Synthesis and Analysis of Biodiversity Status and trends in Rift Valley Lakes of Kenya*, National Museums of Kenya, Nairobi.

Lamb H.H., 1961. Climate in the 1960's. *Geol. J.* 132: 183 – 212.

Litterick M..R., Gaudet J., Kalff J.& Melack J., 1979. *The Limnology of Lake Naivasha.* Proc. SIL – UNEP Workshop, December 16 – 23, Nairobi, Kenya.

Maltby, E., Dugan, P.J., Lefeuvre, J.C., 1988. *Conservation and Development. The Sustainable use of Wetland Resources.* IUCN ,Gland, Switzerland.

Mavuti K.M., 1983. *Studies of community structure, population dynamics and production of the limnetic zooplakton of a tropical Lake Naivasha, Kenya*. Ph.D Thesis, University of Nairobi.

Mavuti K.M., 1992. An account of important freshwater wetlands. *In.* S.A Crafter. S.G Njuguna & G.W. Howard, eds. *Wetlands of Kenya* pp.23-27. IUCN/Nairobi.

McCall G.J.H., 1967. *Geology of the Nakuru, Thomson's Falls – Lake Hannington Area. Government of Kenya.* Ministry of Natural Resources, Geological Survey of Kenya, Nairobi.

Melack J.K., 1976. Temporal variability of phytoplankton in tropical lakes. *Oecologia* 44: 1-7.

Mutangah J.G., Mwatha M.E., Kamau C., Nasirwa O. & Gichuru N., 2000. Biodiversity status and trends. In: *Problem Synthesis and Analysis of Biodiversity Status and Trends in Rift Valley Lakes, Keny.* National Museums of Kenya, Nairobi.

Muthuri F., 1992. Plant products from freshwater wetlands. *In.* S.A. Crafter, S.G Njuguna& G.W. Howard eds. *Wetlands of Kenya*. pp. 109-114. IUCN / Nairobi.

Muthuri F.M.1985 *The Primary productivity of papyrus (Cyperus papyrus) in relation to environmental variables*. Ph.D. Thesis, University of Nairobi

Mwatha W.E., 1991. *Microbial Ecology of Kenyan Soda lakes*. Ph.D Thesis, Univesity of Leicester, UK.

Mwaura F., 1999. A spatio-chemical survey of hydrogeothermal springs in Lake Elmentaita, Kenya. *Intern. Journ. Salt Lake Research*.

Mwaura F., 1992. *Some aspects of hydrochemistry, lake level and vegetation dynamics in the Lake Elmentaita basin, Kenya.* MSc. Thesis, University of Nairobi.

Mwaura F.B., 1981. *Nitrogen fixation in the papyrus swamps of Lake Naivasha.* MSc. Thesis, University of Nairobi, Kenya.

Ndede H., Situma C., Gitau S., Ndetei R. & Makenzi P., 2000. Impact of human activities on landscape and natural resources of the Rift valley lakes of Kenya. In *Problem Synthesis and Analysis of Biodiversity Status and Trends in Rift Valley Lakes of Kenya*, National Museums of Kenya, Nairobi.

Njuguna P.K., Terer T. & Gichuki N.N, 2000. *Conservation and sustainable use of biodiversity in the Eastern Rift Valley Lakes.* UNDP Task Force Report on Baseline Investment Studies in Rift Valley lakes, Kenya. National Museums of Kenya, Nairobi.

Njuguna S.G., 2000. *Conservation and Sustainable Use of Biodiversity in the Eastern Rift Valley lakes in Kenya, Tanzania and Ethiopia.* UNDP Regional Analysis and Synthesis Report, National Museums of Kenya, Nairobi.

NMK (National Museums of Kenya), 2001. *Conservation and Sustainable Use of Biodiversity in the Eastern Rift Valley Lakes: Kenya component.* Final GEF-UNDP Project Proposal (KEN 98/G41/A/1G/99), National Museums of Kenya, Nairobi, Kenya.

Ogutu A.Z., Becha H., Ndonye P., Kahata J. & Gatheru S., 2000. Institutional and community participation, and capacity building needs for conservation and sustainable use of biodiversity in Rift valley lakes of Kenya. *In. Problem Synthesis and Analysis of Biodiversity Status and Trends in Rift Valley Lakes of Kenya*, National Museums of Kenya, Nairobi.

Ondieki M.C., Awour O.V., Murimi K.S., Mwaura F. & Githaiga J., 2000. Impact of climatic and hydrological changes on biodiversity of Rift valley lakes. *In. Problem Synthesis and Analysis of Biodiversity Status and Trends in Rift Valley Lakes of Kenya*. National Museums of Kenya, Nairobi.

Oyieke H.A., ed. 2000. *Conservation and Sustainable Use of Biodiversity in the Eastern Rift Valley lakes. Problem Synthesis and Analysis Report (Kenya): Awassa, Ethiopia*. National Museums of Kenya, Nairobi.

Yuretich R.F. & Cerling T.E., 1983. Hydrogeo-chemistry of Lake Turkana, Kenya; mass balance and mineral reactions in an alkaline lake. *Geochem. Cosmochim. Acta* 47: 1099 – 1109.

Yuretich R.F., 1976. *Sedimentology, geochemistry and geological significance of modern sediments in Lake Rudolf (Turkana) in Eastern Rift Valley, Kenya*. Ph.D Thesis, University of Princetown, New Jersey, USA.

Assessment of the potential impact of biodiversity changes on waterfowl in Lake Victoria

Katebaka Raymond

Makerere University Institute of Environment and Natural Resources, (MUIENR). National Biodiversity Data Bank (NBDB), P. O. Box 7062, Kampala, Uganda
Email:katebakaraymond@yahoo.com

Abstract

To correctly assess the impact of biodiversity changes on waterfowl, it is necessary to utilize appropriate phenological, biodiversity and socio economic data. Where such data is scarce, it is possible to make initial assessments using less complex methods that require only the waterfowl as indicators of biodiversity loss. For such methods, the annual waterfowl census is assumed to consist of an economic growth term and biodiversity loss terms. The relative biodiversity loss in the lake is then expressed as function of the first moments to contribute the potential impact changes. The analysis indicates whether biodiversity loss on Musambwa islands has a positive or negative effect on economies in future. The method was tested using Musambwa waterfowl census since 1999-2005. The results show that for small future annual biodiversity changes have a negative effect on potential environmental changes, except for slight positive changes around 2004. If economic activities and biodiversity indicator loss continues to follow the changes observed in 2004/2005, then effective conservation measures are recommended.

Key words: biodiversity, waterfowl, Lake Victoria

Introduction

Lake Victoria is the largest lake in Africa and second largest fresh water lake in the world (second to lake Superior). It has a surface area of approximately 68,800km^2 of which Uganda, Kenya and Tanzania share. Uganda controls 45%, Kenya 6% and Tanzania 49% respectively. It stretches 412km from the north to south between latitudes 0^{o}30'N and 3^o 12'S and 355km from the west to east between the longitudes 31^{o}37' and 34^{o}53'E. The lake is situated at an altitude 1134m above the sea level. Its basin is about 2000000km^2 and covers parts of Rwanda, Burundi, Sudan, Uganda, Kenya and Tanzania. A number of activities which include, fishing, lumbering, transport, farming in and around the lake, mining to mention but few are the major economic activity to the surrounding countries. It also contributes to major sources of protein nutrition. It is estimated that the gross economic product of the basin in the order of \$3-4 billion annually and that if deterioration of the lake resulted in a 5% reduction in productivity of the region then the consequent loss would be about \$150 million annually Lake Victoria study (2000). Eventually any events like biodiversity changes on waterfowl may result in the lake to have significant impact on the environmental well being in the lake basin and surrounding region.

The threats facing the lake have caused considerable hardship for the population dependant on the lake for their livelihoods and have also reduced the biodiversity of the lake's fauna, most notably the phytoplanktons. As is often the case with eco-region problem, the challenges facing the lake do not reorganize jurisdictional boundaries. Addressing these issues effectively and in a sustainable manner calls for an ecosystem oriented approach that includes the cooperation among all the countries in the lake basin. Key to this approach is a common vision shared by all countries and communities to beneficial use of the lake and how to protect them. The availability of useful and timely data and information on the physical and biological state of the lake and socio economic factors in the basin will support the implementation of that vision. Such data and information have been gathered through several projects that have been undertaken (www.worldlakes.org).

Waterfowl as indicator species

To assess the potential biodiversity changes, waterfowl was considered as an indicator of activities or processes that take place in and around the lake especially on Musambwa islands in lake Victoria. These activities or processes ultimately control or affect biodiversity and productivity of living aquatic resources.

DeAngelis and Cushman (1990) have listed numerous casual chains that are predicted to have effects on marine, like fisheries in the event of biodiversity loss, climatic changes and others due to increased CO_2 concentrations. While there is no ideal 'indicator species', the advantages of fish eating birds as monitors of environmental health are considerable (Gilman *et al.*,, 1979; Peakall, 1988). This work has concentrated on the Grey headed gulls (*Larus cirrocephalus*), with Greater cormorants (*Phalacrocorax carbo*), as the second species; Long-tailed cormorants (*Phalacrocorax africanus*) as the third species and Little egrets (*Egreta garzetta*) as the fourth species as well as Lesser Black Backed Gulls (*Larus fuscus*) as the fifth species. The species have been chosen due to their significance to breed and roost on this site, nest colonially and this helps to obtain accurate total population count. Their populations are high on the food web and thus are likely to reflect the changes within it. More so gulls are more catholic feeders than cormorants and thus may have a greater ability to switch food sources if lower trophic levels are damaged.

The aim of the surveys is to determine whether monitoring changes of waterfowl as indicators of lake biodiversity contribute to conservation. This paper reports the results and discusses their implications to lake environment conservation.

Methods

Study area

Musambwa Islands is one of the Important Bird Areas in Uganda with the size of 0.8ha and located about 31° 47'E, 0047°S, 1,130m in Rakai District near the boarder of Uganda and Tanzania. It is composed of three rocky islets, about 3km offshore in the Sango bay. The largest is about 5ha and the next is about 3ha, whilst the smallest is just a rocky outcrop jutting out of the lake. The two larger ones are sparsely vegetated with shrubs dominated by erlangia sp and short weather beaten trees especially of Ficus species.

The shoreline has no fringing swamp or sandy beaches. Fishermen catching Nile perch (Lates niloticus) and Rastrineobola argentea use the larger island periodically. A group of fishermen have settled on the island (Byaruhanga *et al.,*, 2001).

Total counting

After arriving in the island, we first wait for five minutes for the birds to settle that is if they have been disturbed by engine sound. Accordingly, the waters surrounding the islands is shallow (<10m) deep; the islands are arbitrarily divided during every annual surveys. We use total count method, where every bird in a particular place is counted (Pomeroy, 1992). However the opportunities for making total counts are comparatively rare. More often the area is too large, the birds are too many or the habitat is not open enough. Then we have to rely on estimates instead. However in the case of Musambwa birds are too many. During counting we use motorboat, ride around the islands counting birds in order to ease the work and it is difficult to manage water waves and turbulences especially in months of June/July counts. Observations are made after counting is finished while riding around the island to record the activities taking place and count what was missed during the boat ride.

Results

The results of surveys are given by species that gave out data and are those ones, which were considered to have bigger congregations on the Island enough for analysis. In addition the bigger congregations can give a good prediction of the impact on the food web as they all feed on fish since it is their most preference. Considering the arithmetic mean of the actual counts, Figures 1 and 2 show the abundance and variation of species in January/February and June July season of surveys.

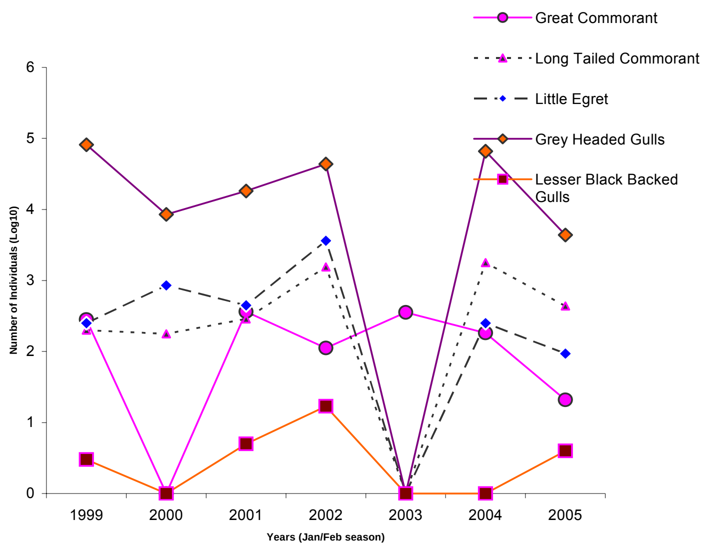

Figure 1. Annual waterfowl counts in the season Jan/Feb surveys.

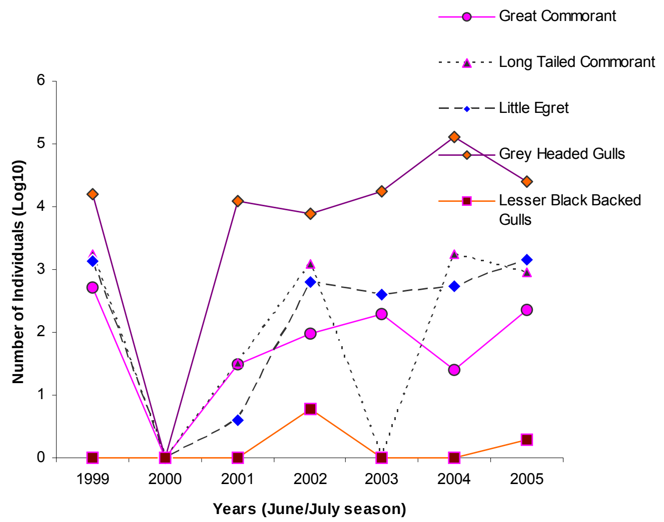

Figure 2. Annual counts in June/July season over the years.

Grey headed gulls *(Larus cirrocephalus)* is more abundant on the island with trend of significant difference of 5 < 6 in 1999. The trend lowered in 2000 but later increased in 2001-2002. There was a count observed in 2003. However, there was a significant rise in 2004 but the trend lowered again in 2005 in the season of January/February. In the season of June/July surveys had a scale of 4.91 but decreased to small scale and later became significantly stable in 2001-2004 with scale of 5.11 which however decreased to 4.39 in 2005. Little egrets *(Egreta garzetta)* was significantly stabe according to the curve in 1999-2002 with a scale of 2.4 and 3.56 later. The trend increased in 2004 to 2.4 but seems to fall in 2005 with a scale of 1.97 in the January/February season. In June/July season there was a significant decrease from 3.25 but this increased to stabilize in the next 5 years.

Long-tailed cormorants *(Phalacrocorax africanus)* had significant stability since 1999-2002, which however deteriorated in 2003. In the later years they were observed to have a significance of 3.25 and 2.64 in 2004 and 2005 years of January/February season. During the June/July season their counts decreased from 3.25, disappeared in 2000 but appeared significantly in later two years. In 2003 they were not counted but were counted in later two years of 2004 and 2005. Greater cormorants *(Phalacrocorax carbo)* were observed in 1999 but disappeared in 2000 and became significantly stable in later four years with a scale of 2.56 and below and 1.32 in 2005 in the January February season.

We have observed a number of activities during our surveys on the islands and these include:

i Fishing being the major economic activity-taking place on the island occurs with two kinds of fishermen in these islands: those who fish Nile perch (Lates niloticus) setting their nets far from the shore and only use islands for short period; and those who fish the sardine-sized Rastrineobola argentea, setting their nets close to the islands and they reside on the island.

ii Human settlement is another activity observed and is assumed that it has reduced the biodiversity of the island.

iii There has been stone collection as a minor activity noticed for some time in the year 2003, by some government officials.

Discussion

The paper has presented figures of annual waterfowl counts, observed fluctuations and calculated relative deviation of the species that are more abundant on the island. It is possible to predict these parameters by randomly selecting any year considering the economic growth factor. The grey headed gulls' relative deviation had a significant difference of P^{+}=0.3139 and P^{-}= 0.999 in 2004. This ratio shows low economic growth. Other species' relative deviation can be calculated.

The studies reported here show how birds can act as indicators of biodiversity change resulting from the impact of human activities and settlement on Musambwa island. Morrison (1986) has reviewed the usefulness of birds as environmental indicators: Populations can be affected both directly and indirectly by environmental change. Although alterations in bird numbers may indicate that an environmental change has occurred, the cause of change must still be determined. The response of individual bird species to human activity shows how these activities interact with each species requirements for food and shelter (Etoori & Abe, 1988). Waterfowl can be instrumental in developing and planning management of lake ecosystems when their populations are monitored effectively.

Conclusion

The Musambwa data of waterfowl counts is supported by Nature Uganda for the period of 1999 to 2005. It is interesting to note that some population decrease of some species occurred for some years, as it is reflected on the both graphs (Figures 1 & 2). The factors that led to this decline require further close analysis before the prediction method is credited with having accurately forecast the increase and decrease in waterfowl population on Musambwa Island. This method can also be applied anywhere to monitor the changes.

References

Byaruhanga, A, Kasoma, P, K, Pomeroy, D., 2001. *Important bird areas in Uganda*. Nature Uganda, East African Natural history society.

David B.P., 992. *Colonial fish eating birds as indicators of environmental health*. National wildlife research center, Canadian wildlife services, Ottawa, Ontario Canada.

DeAngela, D.L & Cushman, R.M., 1990. potential application of models in fore casting the effects of climate change in fisheries. *Transactions of the American fisheries society*, 119:224-239.

Etoori. DK, Abe. E., 1988. *The impact of human activity on bird distribution in a savanna*. Proceedings of the seventh Pan-Africa Ornithological congress Nairobi Kenya.

Lake Victoria case study, 2000. Available http/www.liv.ac.uk/rick/fisheries-web/vicassity.

LVFO, 2002. Available: http//www.inweh.unu.edu/lvfo.htm

Pomeroy E.D., 1992. counting birds: Technical handbook series. African Wildlife Foundation Nairobi Kenya. 8-12.

Toxic cyanobacteria and its toxins in standing waters of Kenya: implications for water resource use

Kiplagat Kotut[1], Lothar Krienitz[2] and Andreas Ballot[2]

[1]Biological Sciences Department, Kenyatta University, P.O. Box 43844 GPO 00100 Nairobi, Kenya
[2] Leibniz Institute of Freshwater Ecology and Inland Fisheries, Department of Limnology of Stratified Lakes, Alte Fischerhütte 2, D-16775 Stechlin- Neuglobsow, Germany
Email:kkotut63@yahoo.com

Abstract

Phytoplankton studies were carried out in selected Kenyan standing waters between 2001 and 2005. Cyanobacteria with known toxin producing potential were recorded in a number of waters. In the freshwaters studied, the most common toxin producing species were Microcystis and Anabaena while in the alkaline saline lakes, Anabaenopsis was common. Cyanobacteria blooms were recorded in a few lakes. A number of lakes and hot spring algal mats had detectable levels of microcystins and anatoxin-a. Cell bound microcystins (LR equivalents) concentration ranged from 1.6–19800 µg g^{-1} Dry Weight (DW) while anatoxin-a varied from below the limit of detection to 1260 µg g^{-1} DW. In alkaline-saline lakes, microcystins and anatoxin-a were also present in stomach contents and liver samples of dead flamingos. Monoculture strains of A. fusiformis from Lakes Sonachi and Bogoria had detectable levels of microcystins while anatoxin-a was present in strains isolated from Lakes Sonachi, Bogoria and Nakuru. The results confirm that cyanotoxins could have played a role in the mortality of flamingos in Lakes Bogoria and Nakuru. Two freshwater sites, Nyanza Gulf (L. Victoria) and Lake Baringo recorded cyanotoxin concentration exceeding WHO's upper limit of 1.0 µg L^{-1} for drinking water. Trends in toxin algae occurrence are considered in this paper. Additionally, implications of findings on water resource use, measures to be taken to reduce the risk of exposure, and eutrophication control steps to reduce cyanobacteria bloom formation are also considered.

Key words: algal blooms, cyanotoxins, flamingos

Introduction

Toxic algae problems in Kenya

The first officially confirmed case of toxic algal poisoning in Kenya is the widely reported death of thousands of marine wildlife along the Kenya Coast at the beginning of 2002 when huge numbers of fish, including manta rays, sharks and tuna were washed ashore (Tomlinson 2002). This led to a temporary ban on fishing along the Kenya coast. The ban, which lasted for two weeks, inflicted large losses on the indigenous fishing community of the Kenya coast. Episodes of wildlife dying under mysterious circumstances have previously been reported for the inland waters of Kenya, the most widely reported case being the successive episodes of massive flamingo die offs recorded at lakes Nakuru and Bogoria in 1993, 1997, 1997 and 2000 (Okoko 2000). Initial investigations in Lake Nakuru indicated that high metal pollution was the primary cause of the flamingo deaths (Kairu 1996; Nelson *et al.,* 1998). However, this explanation could not account for the flamingo die offs in Lake Bogoria, which is relatively free of heavy metal pollution (Wanjiru 2001). Later investigations also pointed at mycobacteria (Kock *et al.,* 1999). The possibility of algal toxins being responsible for the die offs was acknowledged during these investigations, although no confirmatory studies were carried out.

Elsewhere in Africa, occurrences of algal toxins has been documented in a few countries, the majority of the cases being reported from South Africa (Harding 1997, Harding *et al.,* 1995, Scott 1991, Van Ginkel 2003, Van Halderen *et al.,* 1995, Wicks & Thiel 1990) where the toxic effects on animals have been known for over 50 years. Recently, the presence of cyanotoxins in ponds and reservoirs has been reported in Morocco (Nasri *et al.,* 2004, Oudra *et al.,* 2002). The scanty records of cyanotoxin poisoning on the continent do not necessarily mean that the continent has been free of algal toxicity. According to Zilberg (1966), gastroenteritis among school children in Harare, Zimbabwe in the 1960s could have been caused by exposure to cyanobacterial toxins in the municipal drinking water supply. Hence the problem could be more widespread. The overall aim of the study was to document the biodiversity of phytoplankton communities in selected standing waters in Kenya and how their community structure and composition is related to some of the water quality problems experienced in the country. One specific aim of the study was the identification of water bodies with known toxin producing algae and toxin investigations in water bodies with blooms of toxin producing cyanobacteria.

Study methods

Phytoplankton analyses

During the preliminary stage of the study, more than 46 water bodies were investigated. These water bodies were selected from among the various categories of inland waters present in the country including the Rift Valley lakes, the Plateau lakes, Crater lakes, reservoirs and impoundments, small water bodies and sewage oxidation ponds (Figure

1). An output of the preliminary stage was the documentation of key waters with species of cyanobacteria known to have the potential for toxin production (Table 1).

Samples for determination of phytoplankton biomass and composition were collected on diverse dates between 2001 and 2005. Microscopic examination of phytoplankton samples from all the water bodies investigated was carried out using fresh and preserved samples. Enumeration of phytoplankton taxa in lake water samples was carried out on sedimentation chambers. Enumeration procedures are described in Ballot *et al.,* (2003), Ballot *et al.,* (2004 a,b) and Krienitz *et al.,* (2002, 2003).

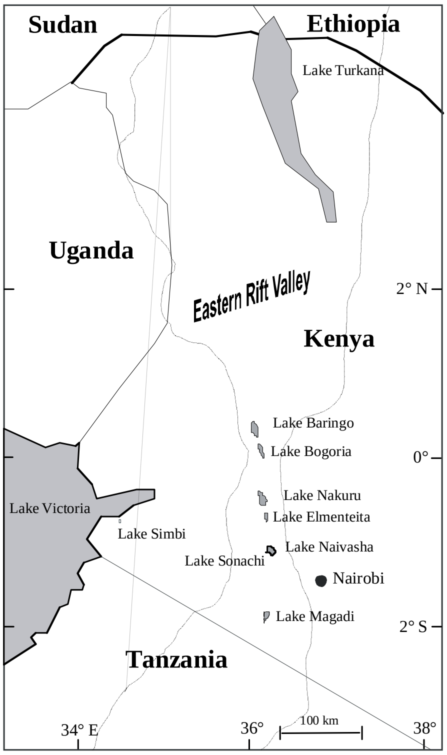

Figure 1. Map showing the location of some of the water bodies investigated.

Toxin analyses

Toxin analyses were carried in water bodies (Fig. 1) with blooms of known toxin producing cyanobacteria as well as on the cyanobacterial mats from the hot springs of Lake Bogoria. Toxin analyses were carried out in the School of Life Science, University of Dundee and at the Leibniz Institute of Freshwater Ecology and Inland Fisheries (working group Biogeochemical Regulation), Berlin. The details of the analytical procedures are provided in Ballot *et al.,* (2003), Ballot *et al.,* (2004 a,b) and Krienitz *et al.,* (2002, 2003). On diverse dates during the study period, dying and dead flamingos were observed around Lake Bogoria hot springs and in Lake Nakuru. Known weights of tissue samples from dead flamingos that included stomach contents, intestines, liver and fecal pellets were collected for toxin investigations. Isolated culture strains of *A. fusiformis* whose biomass dominated the phytoplankton of the saline lakes were also tested for cyanotoxins. To assess the potential risk of drinking lake water directly in Lakes Victoria and Baringo, an estimate of the toxin content per unit volume of lake water was carried out based on the cyanobacteria biovolume and the toxin concentration results.

Results

Distribution of cyanobacteria with toxin producing potential

Microcystis aeruginosa was the most common potential toxic cyanobacterium in freshwater lakes, e.g. L. Baringo and L. Naivasha. *Anabaena* spp. was common in samples from the Nyanza Gulf of the Lake Victoria. Other cyanobacteria with toxin producing potential in the freshwater lakes were members of the genera *Synechococcus* and *Chroococcus*. The dominant cyanobacterium of the saline-alkaline Rift Valley lakes was *Arthrospira fusiformis*, often accompanied by members of the genera *Anabaena* and *Anabaenopsis* (Table 2). The mats of the hot springs at Lake Bogoria were dominated by members of the genera *Phormidium*, *Oscillatoria* and *Synechococcus*.

Cyanotoxin findings

Seven water bodies were found to have cyanobacterial blooms. The seven waters, all natural lakes, can be grouped into two categories; freshwaters serving as important sources of drinking water and alkaline saline waters, most of which are the habitats of the Lesser Flamingoes (*Phoeniconaias minor* Geoffrey) and other avian species. The freshwaters studied were Lake Baringo and the Nyanza Gulf (Dunga Beach) of Lake Victoria, while the alkaline saline waters were Lakes Elmenteita, Sonachi, Nakuru, Bogoria and Simbi (Fig. 1). These lakes had typical surface scum that imparted upon the lakes various shades of blue green to yellow colors depending on the dominant cyanobacteria. An exception to this was Lake Baringo where lake color was mostly dominated by clay suspensoids. The cyanobacteria dominated algal mats on the Hot springs of Lake Bogoria were also included in the toxin investigations.

Phytoplankton biomass was typically high in all these water bodies, with a range from 1.54 to 282.00 mg L^{-1} and 6.7 to 777.1 mg L^{-1} in fresh water and saline lakes respectively (Table 3). Most biomass in these lakes was due to cyanobacteria and varied from 0.22 to 269.00 mg L^{-1} and 5.3 to 769.1 mg L^{-1} in fresh water and saline lakes respectively (Table

3). Among the seven lakes investigated for cyanotoxins, only Lake Elmenteita had a cyanotoxin concentration that was below the limit of detection. Microcystin concentration (LR equivalent) ranged from 39.2 to 19800 μg g^{-1} DW (dry weight) and from 1.6 to 4594 μg g^{-1} DW in the fresh water and saline lakes (including algal mats of Lake Bogoria Hot springs) respectively. Anatoxin-a concentration ranged from an undetectable level to 1260 μg g^{-1} DW and from 0.3 to 223 μg g^{-1} DW in the fresh water and saline lakes (also including algal mats on Lake Bogoria Hot springs) in that order. In all the lakes studied, dissolved cyanobacterial toxins were below the limit of detection (1 μg l^{-1}).

Table 1. Distribution of cyanobacterial species globally known for toxin production in Kenyan water bodies.

Water body, geographical position and altitude	Cyanobacteria	Abundance/comments
Lake Nakuru Pos. S 00°19.583'; E 36°05.325' Alt. 1782 m	*Anabaenopsis abijatae* *Anabaenopsis arnoldii* *Anabaena* sp. *Synechococcus* sp. *Arthrospira fusiformis** *Synechocystis* sp.	Dominance and subdominance changes involving *A. abijatae*, *A. arnoldii*, *Anabaena* sp., *Synechocystis* sp and *A. fusiformis* were noted
Lake Baringo Pos. N 00°36.786'; E 36°01.395' Alt. 998 m	*Microcystis aeruginosa*	*M. aeruginosa* is dominant for a better part of the year
Bogoria Hot springs	*Synechococcus bigranulatus* **Oscillatoria willei** *Phormidium terebriformis*	Mats dominated by each species occur at different occur parts of the hot spring rivulets
Lake Bogoria Pos. N 00°36.786'; E 36°01.395' Alt. 1013 m	*Arthrospira fusiformis** *Synechococcus* sp. *Synechocystis* sp. **Oscillatoria *sp.***	*A. fusiformis* is dominant throughout the year with the rest occurring in small numbers
Lake Elmenteita Pos. S 00°27.345'; E 36°15.333' Alt. 1795 m	*Anabaenopsis abijatae* *Anabaenopsis arnoldii* *Synechococcus* sp.	*A. abijatae* and *A. arnoldii* are occasionally dominant
Lake Naivasha Pos. S 00°48.915'; E 36°18.897' Alt. 1942 m	*Microcystis* sp.	Occasionally subdominant.
Lake Oloiden Pos. S 00°49.025'; E 36°15.834' Alt. 1942 m	*Chroococcus* sp. *Anabaenopsis* sp. *Arthrospira fusiformis**	Occasionally subdominant
Athi Basin Dam Pos. S 01°24.857'; E 36°55.687' Alt. 1556 n	*Microcystis* sp. *Anabaena* sp.	Both species occasionally subdominant
Lake Victoria (Nyanza Gulf) Pos. S 00°05.671'; E 34°42.362' Alt. 1143 m	*Anabaena flos-aquae*, *Anabaena discoidea* *Microcystis aeruginosa*	*A. flos-aquae* was dominant at the time of sample collection
Uhuru Park Pos. S 01°17.369'; E 36°48.995' Alt. 1688 m	*Microcystis* sp.	Occasionally dominant
Magadi Pos. S 02°00.168'; E 36°15.747' Alt. 627 m	*Arthrospira fusiformis** *Anabaena* sp.	
Sonachi Pos. S 00°47.333'; E 36°15.150' Alt. 884 m	**Arthrospira fusiformis** *Anabaenopsis arnoldii*	*A. fusiformis* is most dominant
Simbi Pos. S 00°22.297'; E 34°37.904' Alt. 1145	*Arthrospira fusiformis** *Anabaenopsis abijatae*	*A. fusiformis* is most dominant

*Cases where *A. fusiformis* is believed to be involved in toxin production

Toxin concentration in the stomach contents of dead flamingos averaged 0.196 µg microcystin LR equivalent $.g^{-1}$ of Fresh Weight (FW) and 4.349 µg anatoxin-a $.g^{-1}$ FW (Krienitz *et al.,* 2003). In liver samples, total microcystin ranged from 0.21 - 0.93 microcystin LR equivalent $.g^{-1}$ FW and from 1.76 – 5.82 µg anatoxin-a $.g^{-1}$ FW (Codd *et al.,* 2003 a). Microcystin concentrations of 2.2 and 15.02 µg $.g^{-1}$ DW were recorded from monoculture strains of *Arthrospira fusiformis* isolated from Lakes Sonachi and Bogoria respectively. Anatoxin-a concentrations of 0.3, 10.38 and 0.14 µg $.g^{-1}$ DW were measured from culture strains of *A. fusiformis* isolated from Lakes Sonachi, Bogoria and Nakuru in the same order. Toxins were not, however, detected in cultures of isolates of *A. fusiformis* from Lakes Simbi and Elmenteita.

Microcystin concentration of lake water at the Nyanza Gulf (L Victoria) was estimated at 1.065 µg L^{-1}. In Lake Baringo microcystin concentration (microcystin LR equivalent) ranged from 0.08 to 3.25 µg L^{-1} while anatoxin-a varied from 0.05-0.2 µg L^{-1}. Hence in both water bodies, the cyanotoxin concentration sometimes exceeded the upper limit of 1.0 µg L^{-1} for drinking water set by the World Health Organization (Hitzfeld *et al.,* 2000).

Table 2. Phytoplankton biomass and toxin concentration in Kenya's water bodies tested for cyanotoxins.

	Phytoplankton biomass (mg l^{-1})	Cyanobacterial biomass (mg l^{-1})	Microcystin (µg g^{-1}) DW (LR equi.)	Microcystin variants	Anatoxin-a (µg^{-1} DW)
Nyanza Gulf (a)	282.2	269.1	39.15-41.4	LR, RR, LF & LA	Undetectable
Baringo (b)	1.54-8.17	0.22-5.45	310-19800	LR, RR & YY	270-1260
Nakuru- Co (c)	6.7-104.3	5.3-96.4	129-4594	LR, RR, LF, LF & YR	5-223
Bogoria - lake (c)	62.2-777.1	61.1-769.1	16-227	LR, RR, LF, LF & YR	0.3-9
Bogoria - HS (d)	-	-	221-845	LR, RR, LF & YY	10-18
Elmenteita (c)	17.5-202.0	14.1-196.7	Undetectable	-	Undetectable
Sonachi (e)	312.8-3158.9	312.8-3158.4	1.6-12.2	RR	0.5-2.0
Simbi (e)	61.2-348.3	61.2-348.3	19.7-39.0	LR, RR, LA & YR	1.4

*References to the detailed results; a – Krienitz *et al.,* (2002), b – Ballot *et al.,* (2003), c – Ballot *et al.,* (2004 a), d – Krienitz *et al.,* (2003), e – Ballot *et al.,* (2004 b)

Discussion

Toxic cyanobacteria in alkaline saline lakes

The presence of cyanotoxins in some of Kenya's alkaline saline lakes confirms long held suspicions in the past that cyanotoxins could have played a major role in the mortality of flamingos in Lakes Bogoria and Nakuru. Observations made during the present study in support of the contribution of cyanotoxin poisoning to flamingo deaths include a high concentration of toxins in Lake Bogoria hot spring algal mats, presence of toxins in the stomach contents of dead flamingos (Krienitz *et al.,* 2003), lethal levels of anatoxin-a in liver samples of dead flamingoes and the behavioral pattern of the dying animals (staggering and opisthotonos) (Codd *et al.,* 2003 b). A high concentration of cellular cyanotoxins in the waters of Lakes Sonachi, Simbi (Ballot *et al.,* 2004 a), Nakuru and Bogoria (Ballot *et al.,* 2004 b) is additional evidence in support of cyanotoxin contribution to flamingo die offs. Hence bird mortalities may have possibly resulted from toxins acting alone or in synergy with other stress factors such as heavy metals (Kairu, 1996; Nelson *et al.,* 1998) and mycobacteria infection (Kock *et al.,* 1999).

Although blooms of known toxin producing species of *A. abijatae*, *A. arnoldii* and *Anabaena* sp. were present in Lake Elmenteita, they appear to be atoxygenic. However, the same species could have contributed to toxin production in Lakes Nakuru and Simbi. Toxin presence in Lakes Bogoria and Sonachi, both dominated by *A. fusiformes* suggests that toxic strains of *A. fusiformes* are present in Lakes Bogoria, Sonachi and possibly Nakuru. This finding was supported by the presence of toxins in culture strains of *A. fusiformes* from the three lakes (Ballot *et al.,* 2004 a,b). Although *A. fusiformes* has long been considered to be atoxygenic, recent observations suggest a possible existence of toxigenic strains (Gilroy *et al.,* 2000; Iwasa *et al.,,* 2002). The analysis of a *Spirulina* based commercial human dietary supplement by Gilroy *et al.,,* (2000) found a microcystin concentration of 0.15 to 2.12 µg $.g^{-1}$ DW. However, it was not clear whether the source was indeed *A. fusiformis* or perhaps some other cyanobacterial cells present in the supplement. Another incident was the liver injury sustained by a Japanese male after eating *Spirulina* (Iwasa *et al.,* 2002), which suggested a possible hepatotoxic effect.

Flamingo mortality linked to cyanobacteria toxin poisoning appears not to be confined to the saline lakes of the East African Rift Valley. Toxin related mortalities have been reported for the Greater Flamingos (*Phoenicopterus ruber*) in Spain (Alanso-Andicoberry *et al.,*, 2002) and for the Chilean flamingoes (*P. chilensis*) in Florida USA (Chittick *et al.,*, 2002). The high susceptibility of the flamingos possibly results from their feeding habits, which mostly targets the surface floating cyanobacteria (Codd *et al.,* 2003b).

Cyanotoxin presence in freshwater bodies

The occurrence of cyanotoxins at concentrations above the acceptable upper limit (1.0 µg microcystin LR equivalent L^{-1}) in water bodies used for various domestic chores presents new challenges to the management of freshwater resources in Kenya, a country already designated water scarce (with an annual freshwater availability of less than 1 000 m^3 per person, Ongwen, 1996). First, the presence of cyanotoxins means that the existing water quality assessment and monitoring strategies that employ microbial and physico-chemical criteria will no longer be adequate in the assessment of water quality, especially in water bodies showing evidence of progressive eutrophication. Second, although there are no confirmed records of human health problems that have been linked to toxic cyanobacteria in Kenya, the toxin levels recorded in some of the waters serving as sources of domestic water for some rural communities means that incidences of human health problems cannot be ruled out. Poor medical services and records in rural communities that rely on natural water bodies means that the health problems associated with consumption of contaminated water can easily go unnoticed. Third, presence of cyanotoxins in some of Kenya's freshwaters will further diminish the amount of clean water available to its citizens in a country were a significant proportion of the population already lacks access to clean water.

Although this study did not find significant concentrations of dissolved cyanotoxins, a separate study carried out by Mwaura *et al.,* (2004) recorded a dissolved microcystin concentration of up to 2.85 µg L^{-1} based on ELISA tests in a highland reservoir of Kenya during the dry season. Presence of cyanobacteria with the potential for toxin production in some of Kenya's freshwater bodies is another cause of concern for a number of reasons. First, this is a potential problem in that there is a possibility that an increase in nutrient input into these waters or water column stability changes can lead to bloom development by these cyanobacteria and the associated problems. Second, these waters could have been subjected to progressive eutrophication and the observations made are signs of impending bloom related problems unless the situation is promptly addressed.

Conclusions

- Cyanotoxin poisoning plays a major role in the episodes of flamingo die offs that have been recorded in the Rift Valley lakes. These die offs possibly results from toxins acting alone or in synergy with other stress factors that stem from human activities.

- The presence of cyanotoxins at concentrations above the acceptable upper limit (1.0 µg microcystin LR equivalent L^{-1}) has been demonstrated for some water bodies used for various domestic chores. This means that the water stressed communities in Kenya; especially those in the rural areas, as well as domestic and wild animals are presently exposed to cyanotoxin related problems.

- Presence of cyanobacteria known for toxin production in a number of water bodies further amplifies the already clear signs of a progressive deterioration of water quality in the country.

Recommendations

Reducing the risk of exposure

A long term solution in places where cyanobacterial toxins are prevalent is the provision of clean drinking water throughout the year. A continuous monitoring program is necessary to advice on areas that require the supply of drinking water. An immediate step is to inform affected communities on the dangers inherent to the use of toxin-laden waters, create awareness on early signs of bloom formation and provide practical ways of reducing the risk to exposure. Because cellular based toxin loads are considerably higher than those in solution (consistently below the limit of detection during the study), efforts that reduce the quantity of cyanobacterial cells can reduce the risk of toxin exposure to a large extent. Such measures would include advising communities living around toxin contaminated water bodies against directly drinking the untreated water, whether a surface bloom is visible or not, and preventing domestic animals from drinking such waters directly.

Eutrophication control

The first step in the prevention of eutrophication related water quality deterioration is to investigate the various sources of nutrients. Where effluents from sewage treatment plants find their way into the water body, effluent water quality standards should strictly be enforced and the treatment facilities maintained regularly. Because nutrient and pollution runoff from the land is a common source of eutrophication in the rural areas of the country, preventing these wastes from getting into the water can help reduce the frequency, toxicity and duration of harmful algal blooms. Special attention should be given to phosphorus, as an important nutrient for cyanobacterial dominance.

Research and monitoring

Some dedicated research focusing on HABs should be initiated. It is important to confirm the identity and distribution of toxin producing algae in Kenyan waters. The role of eutrophication and other environmental factors involved in the suppression or promotion of algal bloom development as well as toxin production needs to be determined. For example, the impact of changes in nitrogen phosphorus (N:P) ratios and concentration on toxin production, specific mortality by pathogens and grazing on HAB species. In the pursuit of natural control options, research should also focus on identifying the natural enemies of HAB species such as bacteria, viruses etc. and conditions under which they are most effective. Monitoring is important for the acquisition of the necessary data and information for making informed water resource management decisions. For each water body involved, ecological and water quality indicators of lake health must be carefully selected and a monitoring schedule established. Research efforts should focus on establishing useful indices of water quality change. For example, organisms sensitive to water quality deterioration that disappear when the quality declines, can be used as possible sentinels of deleterious changes.

Capacity building

Capacity building and institutional development are essential for successful implementation of the measures proposed above and should focus at water research institutions, local communities as well as those authorities responsible for the management these waters. It is also important to strengthen the capabilities of community groups to use and manage water resources prudently. To reduce cost limitation, low cost but effective data collection for monitoring involving citizen volunteers, schools etc. should be considered.

Acknowledgements

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References

Alonso-Andicoberry, C., Garcia-Villada, L., Lopez-Rodas, V. & Costas, E., 2002. Catastrophic mortality of flamingos in a Spanish national park caused by cyanobacteria. *Vet. Rec.* 151, 706-707.

Ballot, A., Krienitz, L. Kotut, K., *et al.,,* 2004a. Cyanobacteria and cyanobacterial toxins in three alkaline lakes of Kenya – Lakes Bogoria, Nakuru and Elmenteita. *J. Plankt*. *Res*. 26, 925-935.

Ballot, A., Pflugmacher, S., Wiegand, C., Kotut, K. & Krienitz, L., 2003. Cyanobacterial toxins in Lake Baringo, Kenya. *Limnologica* 33, 2-9.

Ballot, A., Pflugmacher, S., Wiegand, C., Kotut, K. & Krienitz, L., 2004b. Cyanobacteria and cyanobacterial toxins in the alkaline crater lakes Sonachi and Simbi. *Harmful algae* 4, 139-150.

Chittick, E., Puschner, B., Walsh, M. *et al.,*, 2002. Blue-green algae microcystin toxicosis in captive Chilean flamingos. *Proceedings of the American Association of Zoo Veterinarians*. Milwauke, USA, 115-116.

Codd, G.A., Metcalf, J.S., Morrison, *et al.,*, 2003a. Cyanobacterial toxins in the mass mortalities of flamingos at Kenyan lakes. *Phycologist*. Abstracts of the annual winter meeting, Galway, 2-5 January 2003.

Codd, G.A., Metcalf, J.S., Morrison, *et al.,*, 2003b. Susceptibility of flamingos to cyanobacterial toxins via feeding. *Vet. Rec.* 152, 722-723.

Gilroy, D.J., Kauffman, K.W., Hall, R.A., Huang, X. & Chu, F.S., 2000. Assessing potential health risks from microcystin toxins in blue green algae dietary supplements. Environ. Health Persp. 108: 435-439.

Harding, W.R., 1997. New regional emergence of cyanobacterial toxicosis. Harmful Algae News 16, 12-13.

Harding, W.R., Rowe, N., Wessels, J.C., Beattie, K.A., Codd, G.A., 1995. Death of a dog attributed to the cyanobacterial (blue-green algal) hepatotoxin nodularin in South Africa. *J. S. Afr. Vet. Ass.* 66, 256-259.

Hitzfeld, C.B., Höger, S.J., & Dietrich, D.R.,, 2000. Cyanobacterial toxins: removal during drinking water treatment, and human risk assessment. Environ. Health Persp. 108, 113-122.

Iwasa, M., Yamamoto, M., Tanaka, Y., Kaito, M., Adachi, Y., 2002. *Spirulina* associated hepatoxicity. *Am. J. Gastroenterol.* 97, 3212-3213.

Kairu, J.K., 1996. Heavy metals residues in birds of Lake Nakuru, Kenya. *Afr. J. Ecol.* 34, 397-400.

Kock, N.D., Kock, R.A., Wambua, J. Kamau, G.J. & Mohan, K., 1999. *Mycobacterium avuim*-related epizootic in free-ranging Lesser Flamingos in Kenya. *J. Wildlife Dis.* 35, 297-300.

Krienitz, L., Ballot, A., Wiegand, C., Kotut, K., Codd, G.A. & Pflugmacher, S., 2002. Cyanotoxin producing bloom of *Anabaena flos-aquae*, *Anabaena discoidea* and *Microcystis aeruginosa* (Cyanobacteria) in Nyanza Gulf of Lake Victoria, Kenya. *J. App. Bot.* 76, 179-183.

Krienitz, L., Ballot, A., Wiegand, C., Kotut, K., Codd, G.A. & Pflugmacher, S., 2003. Contribution of hot spring cyanobacteria to the mysterious deaths of lesser flamingos at Lake Bogoria, Kenya. *FEMS Microbiol. Ecol.* 43, 141-148.

Mwaura, F., Koyo, A.O. & Zech, B., 2004. Cyanobacterial blooms and the presence of cyanotoxins in small high altitude tropical headwater reservoirs in Kenya. *J. Water Health* 2, 49-57.

Nasri, A.B., Bouaicha, N., Fastner, J., 2004. First report of a microcystin-containing bloom of the cyanobacteria *Microcystis* spp. in Lake Oubeira, eastern Algeria. *Arch. Environ. Con. Tox.* 46, 197-202.

Nelson, Y.M. Thampy, R.J., Motelin. G.K., Raini, J.A., Disante, C.J., & Lion, L.W., 1998. Model for trace metal exposure in filter-feeding flamingos at alkaline Rift Valley lakes, Kenya. *Environm. Toxicol. Chem.* 20, 2302-2309.

Okoko, T., 2000. Lake Flamingos Threatened by Effluent. *Panafrican News Agency*, December 17, 2000, Nairobi, Kenya.

Ong'wen, O., 1996. NGO experience, intervention, and challenges in water strain, demand, and supply management in Africa. In: *Water Management in Africa and the Middle East: Challenges and Opportunities* (Eds E. Rached, E. Rathgeber, & D.B. Brooks) IDRC, Ottawa, Canada, 295 pp.

Oudra, B., Loudiki, M., Vasconcelos, *et al.,* (2002. Detection and quantification of microcystins from cyanobacteria strains isolated from reservoirs and ponds in Morocco. *Env. Toxicol.* 17, 32-39.

Scott, W.E., 1991. Occurrence and significance of toxic cyanobacteria in Southern Africa. *Water Sci. Technol.* 23, 175-180.

Tomlinson, C., 2002. Fish deaths confirmed to be caused by toxic algae, expected to clear naturally. *Associated Press*, February 08, 2002, Nairobi, Kenya.

Van Dolah, F.M., 2000. Marine algal toxins: Origins, health effects, and their increased occurrence. Environ. Health Persp. 108, 133-141.

Van Ginkel, C.E., 2003. A national survey of the incidence of cyanobacterial blooms and toxin production in major impoundments. Internal Report No. N/0000/00/DEQ/0503. Resource Quality Services, Department of Water Affairs and Forestry, Pretoria, v + 44 pp + 4 pp appendix.

Van Halderen, A., Harding, W.R., Wessels, J.C., *et al.,*, 1995. Cyanobacterial (Blue green algae poisoning of livestock in Western Cape Province of South Africa. *J. South Afr. Vet. Ass.* 66, 260-264.

Wanjiru, J., 2001. Kenya's Pink Flamingos Weighted Down by Heavy Metals. ENS, July 16, 2001. Available on URL: *http://www.worldlakes.org/News/Kenya*. Accessed August 2002.

Wicks, R.J. & Thiel, P.G., 1990. Environmental factors affecting the production of peptides in floating scums of cyanobacteriom *Microcystis aeruginosa* in a hypertrophic African reservoir. *Environ. Sci. Technol.* 24, 1413-1418.

Zilberg, B., 1966. Gastroenteritis in Salisbury European Children-A Five Year Study. *Centr. Afr. J. Med.* 12, 164-168.

Water quality trends and input loads to Lake Nakuru

Andrew Kulecho[1] and Victor Muhandiki[2]

[1]Nakuru Water and Sanitation Services Company Limited, P.O. Box 16314, Nakuru, Kenya
[2]Department of Environmental Systems Engineering, Ritsumeikan University, Kusatsu, Shiga, Japan Email: andrewkulecho@yahoo.com

Abstract

Water quality trends of Lake Nakuru, Kenya and its feeder streams were monitored in 2001-2004. Water samples were obtained from eleven stations in the lake and five points on inflowing streams. The items monitored were physico-chemical and biological components. The drainage basin was characterized by recurrence of dry spells throughout the period with water losses in the lake exceeding the amount received to cause a fall in water volume. The lake water salts content rose – conductivity and alkalinity reached 58 ms/cm and 5,200 mg/l respectively. PH was more or less constant (10.00 – 10.36). Daytime oxygen concentration in the mid-lake system rose (to greater than 20 mg/l) while remaining low in the inflow portions of the lake. On one occasion, the lake suddenly deoxygenated resulting to fish kills. Silt and organic loading were intensive in the north and south, contributing significantly to a decline in water transparency (secchi depth, 13 cm or less). Total phosphates varied significantly (4.39-9.128 mg/l). Of the nitrogen forms, only ammonia and total nitrogen were detected in appreciable amounts. The phytoplankton biomass was high and was mainly dominated by populations of Arthrospira fusiformis (mean, 158-2924 coils/ml). Various strains of Anabaenopsis spp. with Coccoids also occurred at low biomass. The only exception was in 2003 and 2004, when they developed into a dense bloom that was followed by flamingo mortalities. The northern portion had very low phytoplankton content while in the south, the phytoplankton community was one with relatively higher numbers of zooplankters and Anabaena spp. The yearly input load of the lake was 40,000 tons with Rivers Makalia and Njoro contributing significantly to the load. These results suggest eminent change of the lake ecosystem, which could arise due to human impacts from catchment degradation and urbanization.

Key words: Biomass, Load, Mortalities

Introduction

Past studies have highlighted the importance of Lake Nakuru as home of wildlife and an important tourist attraction. The lake is a hypereutrophic soda lake located at 0º 22' S 36º 05' E and altitude 1759 m.a.s.l. It is characterized by high evaporative processes which with its catchment geochemistry makes it an extreme ecosystem, unfavorable to most aquatic life. Only a few tolerant and adapted species attain high population density: (Jenkins 1929; 1932 & Veraschi 1981). The lake therefore supports a unique and highly productive community of algae, and few invertebrates that form the food basis of the lake food chain.

Principal input to its water budget are, rainfall, stream flow and springs which are major determinants of its salinity state. Effluent run off from the adjacent Nakuru town also end up in the lake. Various human activities are within the surrounding L. Nakuru basin which makes it highly vulnerable to pollution. For its long time conservation, a continuous monitoring system is vital. This is the objective of the current monitoring program.

Materials and methods

The locations of sampling points are shown in Figure 1. Water samples were obtained monthly. A total of 16 stations were monitored.

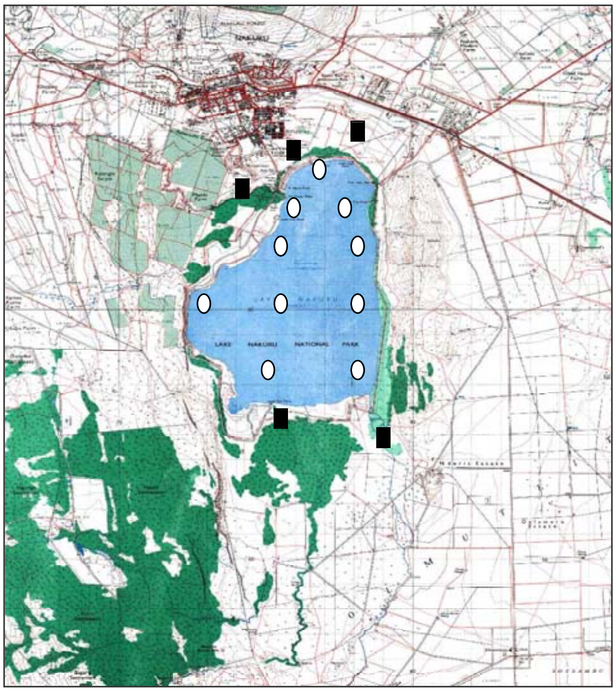

Figure 1. Sampling Locations.

Note: Dark squares = feeder streams sites; white circles = Lake sites

Water temperature, pH, conductivity, Salinity and Dissolved Oxygen were determined on site using a

WTW multiline meter. A secchi disc (40 cm diameter) with black and white quadrants was used for transparency measurements. Chlorophyll samples were filtered on site and transported to the laboratory in aluminium foils under cold storage. Stream flow discharges were measured using a Hiroi Current meter while lake levels were taken using standing gauges in the north of the lake. All laboratory parameters were analyzed at the Nakuru Water Quality Testing Laboratory using the Standard Methods 18th Edition procedures for Examination of Water and Waste Water. Stream flow discharges were measured using a Hiroi Current meter while lake levels were taken using standing gauges in the north of the lake.

Results

Lake levels

Variation in mean lake level during 2001-2004 is shown in Figure 2.

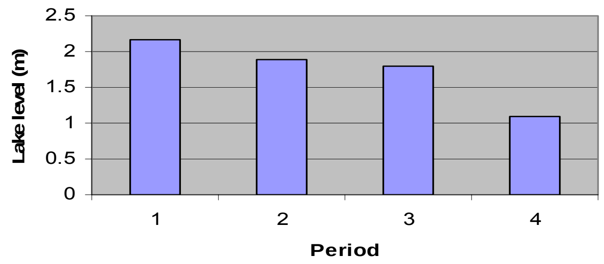

Figure 2. Lake levels during 2001 – 2004 (Note: x axis, 1-4 = 2001-2004)

The last major lake level rise was caused by the heavy 'Elnino' rains in 1998. After this episode, it declined steadily in a period that was characterized by low precipitation and low stream flow discharges. Major feeder streams (with the exception of the sewage drain), dried out lasting for several months. The lake level stabilized in 2004 after dropping by over 1 meter to start showing very small oscillations.

Light Conditions

The lake was characterized by remarkably high, long lasting phytoplankton content which inhibited underwater light for several months. Over the monitoring period, transparency was very low. In the very clear water phase, it hardly surpassed 40 cm. In the inflow sections, it was relatively lower (as low as 5 cm) with obvious turbid water. As figure 2 below implies, seasonal variations were quite distinct. Annual mean transparency increased steadily, showing much higher values during rainfall seasons. Occasionally, light shading and coloring of water by algal scum intensified underwater light inhibition resulting in very low transparencies. In the near shore areas, masses of scum transported by wind accumulated which together with the upwelling water (with its sediment content), had a remarkable effect on water transparency.

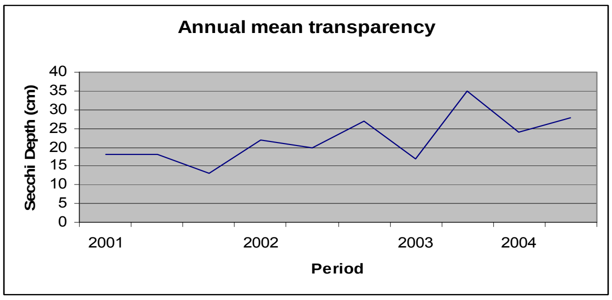

Figure 3. Mean Transparency in L.Nakuru during 2001-2004

Dissolved oxygen

High variance in the vertical profiles and daytime concentration were indicated throughout the period. Measured oxygen concentration showed steep vertical gradients during the daylight hours, euphotic zone oxygen rising to super saturation (10-22 mg/l), before depletion in the later hours. Spatial variation was also indicated in each sampling occasion. The Sewage and Makalia inlet areas showed extreme oxygen depletion with concentrations as low as 2 mg/l being recorded. These areas had remarkable numbers of low oxygen favoring organisms such as protozoa, and relatively low phytoplankton content. In August 2004, extreme oxygen depletion occurred suddenly, causing massive fish kills.

pH

The lake water pH was fairly constant, showing very small fluctuations. The annual mean value lied between 10.00 and 10.36.

Conductivity and Salinity

During 2001-2004, changes in conductivity and salinity tracked precipitation trends.

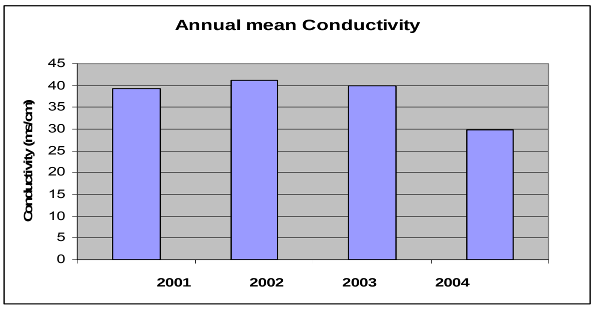

Figure 4. Mean conductivity in L. Nakuru during 2001-2004.

Droughts occurred during December-March every year, causing falls in water level with subsequent rise in conductivity and salinity. Yearly mean conductivity was 40 ms/cm in 2001-2003 but dropped to 30 ms/cm in 2004. In rainfall seasons, conductivity salinity values fell remarkably showing relatively lower values in inflow sections of the lake. During March 2002, the mid-lake system attained a level up to three times greater the yearly average value. Changes in spatial and temporal conductivity levels are known to exert major influences on the biota of the lake, since each aquatic species has individual tolerance level. Values obtained during this monitoring period were in the lower range of reported normal levels for L. Nakuru (range 9.5 – 165.5 ms/cm) though the maximum value for 2002 (58 ms/cm) might have exceeded the upper tolerance limit of 50 ms/cm for *Arthrospira fusiformis* (Vareschi 1982) for L. Nakuru.

Nutrients

Phosphate levels were high and quite variable. Total phosphate ranged 4.39-9.128 mg/l while soluble phosphate ranged 1.742-4.670 mg/l. However, no obvious spatial difference in phosphate levels was observed. Of the nitrogen forms, nitrite levels were very low, showing near constant levels (~6µ g/l) throughout the period. The annual mean concentration ranged 11-210 µg/l. Ammonia was the dominating form of inorganic nitrogen and showed sporadic elevations with algal decay. The annual mean concentration ranged 2.79-1.48 mg/l with maximal values (2.29 and 2.63 mg/l) in August and October 2002 respectively. Levels recorded in the north and south were also high indicating pollution. The mean concentration of nitrate ranged 0.04-0.45 mg/l, highest value (0.89mg/l) recorded in June 2001. However, no clear seasonal variation in nitrate concentrations was observed and the true trend could probably be revealed by continued monitoring. The total nitrogen content of the water was high and ranged 5.0-15.7 mg/l. Mid-lake areas showed relatively lower levels. The sewage area had very high total nitrogen levels suggesting high levels of organic load possibly from Nakuru town.

Table 1. Annual mean concentration of the nitrogen forms in L. Nakuru (Numbers in brackets indicate the range of mean concentration per year).

Year	NH3-N mg/l	NO2-N µg/l	NO3-N mg/l	TN mg/l
2001	0.69(0.4-1.17)	210(90-329)	0.45(0.37-0.87)	9.29(1.93-16.64)
2002	1.48(0.3-2.63)	11(5-8)	0.04(0.01-0.12)	15.65(74.70-25.64)
2003	0.28(0.04-0.81)	54(6-237)	0.05(0.02-0.11)	5(0.36-1.22)

Phytoplankton biomass, species composition and density.

Phytoplankton biomass was estimated by measurement of chlorophyll 'a'. The yearly mean chlorophyll 'a' ranged from 159- 619 mg/m³. Surface concentrations occasionally rose to levels as high as 980mg/ml. Chlorophyll concentration in the sewage area was however, much lower.

Table 2. Biological parameters in L. Nakuru during 2001-2004.

	2001	**2002**	**2003**	**2004**
Chlorophyll 'a', mean (mg/l)	159	241	575	619
Arthrospira Fusiformis, mean (coils/ml)	-	158	2924	1504
Anabaena arnoldii, mean (coils/l)	-	86	1187	518
Anabaena arbijartae, mean (coils/l)	-	108	383	540
Rotifers, mean (no./ml)	-	13	889	88
Ciliates	-	216	-	206

The phytoplankton community was dominated by cyanobacteria species *(Arthrospira fusiformis, Anabaena abijartae*, *Anabaena arnodii, Anabeana spp. and coccoid cyanobacteria)* which showed considerable temporal and spatial variation in density, composition and distribution throughout the period. *Arthrospira fusiformis* was the dominant cyanobactria species (1500 – 8700 coils/ml), accounting for more than 75% of the cyanobacteria biomass. But twice in the period (2003 and 2004), the phytoplankton composition shifted to a dense bloom dominated by *Anabaena Anrnoldii and Anabaenopsis spp.* In each occurrence, the bloom crushed to be followed by flamingo mortalities, underscoring the fact that a food source can also act as the course of toxicosis. The northern portion of the lake had very low phytoplankton content while in the sourth, the composition seemed to change to one dominated by *Anabaena Arnodii* with low biomass of *Anabaena abijartae* and coccoids.

Export load through influent streams

The export loads of the major feeder streams (Njoro, Makalia, Nderit, Baharini Springs and the Sewage drain) were monitored. The yearly, input load of the lake was 40,401.2t/yr. 78% of this was total dissolved solids while the remaining 12% comprised, total suspended solids, organic matter and nutrients. Over 69.5% of the load entered the lake through River Njoro, highest proportion being total dissolved solids. The annual suspended solids input load was 2,597t/yr, over 70% being delivered through River Makalia. Of the total nutrient load, 80-90% was delivered through rivers Njoro and Makalia, with the former carrying highest proportion. The total external organic matter input load was mainly through direct urban run-off, though sewage load had also important component.

Table 3: Mean discharge rates (m^3/day) for the feeder stream of L.Nakuru.

Stream	Mean flow	Minimum flow	Maximum flow
River Njoro	52116	0	268980
River Makalia	26065	0	174539
River Nderit	13703	0	53571
Baharin springs	1831	301	3924
Sewage drain	2184	173	5140

Table 4. Selected stream input loads (tons/yr) to L. Nakuru (2001-2002).

Stream	Suspended solids	Total Phosphate	COD	Total Nitrogen	Other inputs	Total load
Sewage drain	50	2.98	122	40.8	1164.22	**1380**
Baharin Springs	3	0.34	248	3.7	640.96	**896**
River Nderit	541	1.75	548	25.0	1107.25	**2223**
River Makalia	1852	10.54	168	56.6	7086.86	**9174**
River Njoro	200	20.88	712	176	26984.12	**28093**
Total load	**2646**	**36.49**	**3096**	**302**	**36982.42**	**40401**

Discussion

Human induced modifications in the L. Nakuru drainage basin are more or else responsible for the present unpredictable climate experienced in the region today that may affect the lake water quality. Evaporation losses were high, reducing the water volume with a remarkable shift in conductivity and salinity levels that might have influenced changes in phytoplankton composition observed over the period. Conductivity and salinity in 2002 probably exceeded the physiological tolerance limit for *Arthrospira fusiformis*.

The direct effect of siltation was especially marked in the inflow sections of the lake causing a decline in water transparency, especially during rain seasons. This influenced the dominance of the main algal species, *Arthrospira fusiformis* in favour of *Coccoids* and strains of *Anabaenopsis spp.* The Sewage area had a remarkable organic load content which inhibited phytoplankton production in favour of protozoa which were present in high numbers. High nutrient loading was indicated both in the north and south of the lake probably due to the direct effect of agriculture and urban run-off. Stream export load, data indicate River Makalia to be highest transporter of silt load reflecting extensive erosion occurring in its drainage area. Both River Njoro and Makalia that drain off agricultural areas were found to be highest nutrient transporters, reflecting the prevalent use of nitrogen and phosphorus-based fertilizers in their drainage areas. High organic load reach the lake through urban run-off, sewage drain and River Njoro.

Conclusion

Results of this monitoring exercise shows that conductivity states of the lake are linked to the climatic cycle, responding accordingly from low to high conductivity states. That the lake water is well buffered against changes, with PH only changing little. Water transparency results indicate that light conditions in the mid-lake and sheltered areas is regulated by the phytoplankton content while in the inflow zones, silt load has some significant role. The phosphate levels are high and quite variable implicating high reserves in sediment probably regulated by absorption de-sorption processes. Total nitrogen is also high reflecting a low nitrogen-phosphate ratio which indicates nitrogen to be in limited supply. Frequent elevations in ammonia levels also occur especially during bloom decay and near the river mouth. Marked influence of stream inputs occur in the north and south affecting the dominance of the main algal species. Shift in dominance to undesirable algal species occur in the entire lake, ending up with crashes that at times, spark off flamingo mortalities. The decaying algae create anoxic conditions which cause adverse effect on the introduced fish.

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References

Ballot *et al.,* 2004. *Cyanobacteria and Cyanobacteria toxins in the alkaline crater lakes Sonachi and Simbi, Kenya.*
Harper, D.M. *et al.,* 2003. *Aquatic Biodiversity and Saline Lakes*.
Lambert, 1992. *Predictability in lake ecosystem: the role of Biotic interactions*
S.Njuguna, S., 1985. *Nutrients and Phytoplankton in Lake Victoria, Kenya*.
Vareschi, E., 1982. *The Ecology of Lake Nakuru (Kenya) III. Abiotic factors and primary production.*

Deposition of phosphorus and nitrogen coming from precipitation into the lakes of North Poland

Wlodzimierz Marszelewski

Nicolaus Copernicus University in Torun, Institute of Geography, Department of Hydrology and Water Management, ul. Fredry 6, 87-100 Torun, Poland
Email: marszel@geo.uni.torun.pl

Abstract

This study presents the results of the investigations into the concentration of phosphorus and nitrogen in precipitation and its loads deposited directly on the lakes. The examinations comprised 137 lakes (>10 ha), which covered the area of 3840 km^2. The author assesses the threat phosphorus and nitrogen from wet deposition pose to the lakes on the grounds of the criteria of the permissible values of phosphorus and nitrogen loads released into the lakes. A particular attention was drawn to their considerable time and spatial diversity. The author found that the shallow lakes whose mean depth did not exceed 3.0-4.5 m were mostly susceptible to the negative influence of nitrogen and phosphorus coming from precipitation. Nitrogen was found to create the biggest threat to the lakes due to the amount of its permissible loads. In the case of the shallowest lakes (mean depth <1.5 m) the nitrogen load from precipitations often exceeded the value of a permissible load, particularly in the years 2000-2002.

Key words: deposition, lakes, nutrients.

Introduction

Phosphorus and nitrogen belong to the most important elements in precipitation. This results from the fact that these elements create a serious threat to aquatic and terrestrial ecosystems. They cause the increase in eutrophication and acidification of lake waters. Their share in the budget of biogenes prepared for each lake proved extremely different. In the case of the lakes heavily polluted with sewage it was insignificant and was lower than 0.5%. On the other hand, the lakes with natural features it sometimes exceeded 30%. At the end of the twentieth century, due to the fast increase of sewage purification and the decline of sewage discharge into many lakes, the role of phosphorus and nitrogen from wet deposition began to grow. On the other hand, owing to the restraint of pollution emission into the atmosphere, the concentration of these and other chemical substances in precipitation has been systematically decreasing. All this results in a significant changeability in the amount of phosphorus and nitrogen deposition and their share in the budget of lake biogenes. These issues have been presented on the grounds of the investigations conducted in various climatic zones (among others: Delumyea & Petel 1978; Sober & Bates 1979; Shaw *et al*. 1989, Driscoll *et al*. 1995; Bennett *et al*. 1998; Nõges *et al*. 1998; Bootsma *et al*. 1999; Morales *et al*. 2001; Tamatamah *et al*. 2005). The results of the above investigations indicate that the influence of phosphorus and nitrogen (and other substances) coming from wet deposition upon the lakes depends upon their geographic location and the related air pollution and the manner of catchment management.

This study attempts to present the assessment of the threat nitrogen and phosphorus from wet deposition pose to the lakes depending upon the lake's mean depth. The author calculated the portion of the real (observed) TN and TP loads with respect to the permissible loads for the lakes determined according to Vollenweider's criteria (1968).

Materials and methods

The study has been based upon the results of the investigations carried out within the State Environmental Monitoring conducted by the Inspection for Environmental Protection and Institute of Meteorology and Water Management in the years 2000-2003. Within this monitoring wet-only precipitation was collected by means of stationary collectors, which were only exposed during precipitations. Every day (if there was any precipitation), the volume of the collected precipitation was measured, and then the samples were frozen to the temperature below -20^{o}C. Chemical analyses were conducted in monthly cycles at the accredited laboratories. The results were initially examined at the Institute of Meteorology and Water Management, Branch in Wroclaw (Twarowski *et al*. 2002). Further research accounted for the data obtained from additional precipitation measurement stands, which were characteristic of the average field of the precipitation sum in Northern Poland. Then, the maps presenting spatial distribution of precipitation and the concentration of the substances found in precipitation, as well as the extent of their deposition were prepared.

The assessment of the threat phosphorus and nitrogen from wet deposition pose to the lakes was conducted according to the method which determined the permissible value of the annual loads of phosphorus and nitrogen released in the lakes, depending upon their mean depths (Vollenweider 1968). 137 lakes of the area bigger than 10 ha were selected for the examinations. The lakes are located in the north part of Poland and cover 3840 km^2 (Figure 1). Moreover, they belong to two geographic regions: the Kaszuby Lakeland and Bory Tucholskie.

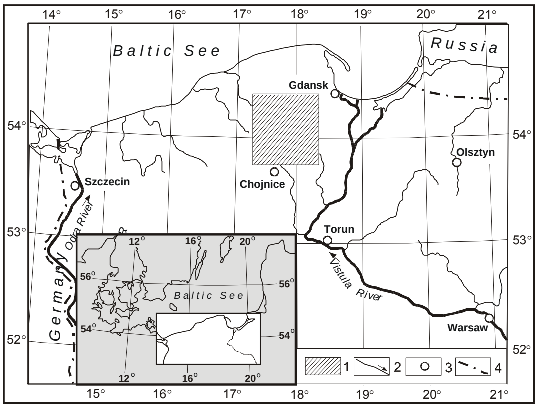


Figure 1. Location of the study area. 1-study area, 2-rivers, 3-towns, 4-country borders.

The rate of using the permissible load (ULp) was calculated for each lake. It is the quotient of phosphorus (or nitrogen) load from wet deposition and the permissible load (the final result is expressed in %). The ULp value equalled 100%, which meant that the released load of phosphorus (or nitrogen) corresponded to the permissible value. If ULp was <100% it meant that the load of phosphorus (or nitrogen) was lower than the permissible value, and if ULp > 100% then the load was greater than the permissible value. If ULp>200%, then there was an abuse of the critical load.

Results

The concentration of phosphorus in precipitation was characterised by considerable diversity. The extreme mean monthly values were recorded in 2000 and amounted from 0.16 mg P·dm^{-3} in January to 0.272 mg P·dm^{-3} in October (the data come from a monitoring station in the study area in Chojnice 53^{o}42'N, 17^{o}33'E). The time distribution revealed big fluctuations of mean values in particular months, with a distinct minimum in winter, i.e. in December, January and February (Fig.2). The concentration of nitrogen showed slightly smaller diversity. Its biggest fluctuations of the mean monthly values were recorded in 2002 (from 0.67 do 4.34 mg N·dm^{-3}). It must be noted that these fluctuations were several times smaller a year before, and ranged from 0.78 to 1.76 mg N·dm^{-3} (Figure 2).

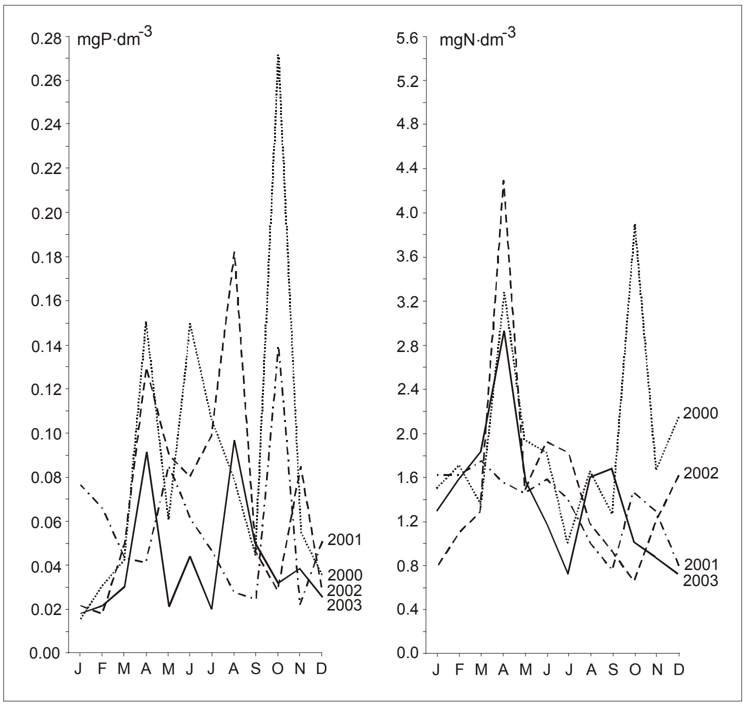


Figure 2. Diversity in the concentration of phosphorus and nitrogen in precipitation at the monitoring station in Chojnice in the years 2000-2003 (mean monthly values).Based on the data obtained within the State Environmental Monitoring from the Inspection for Environmental Protection and the Institute of Meteorology and Water Management.

The annual loads of phosphorus oscillated between 0.164 and 0.520 kg P·ha^{-1}, and of nitrogen between 5.72 and 10.22 kg N·ha^{-1}. Their extreme values were not always related to the sums of precipitations (Table 1). This resulted from significant short-term fluctuations of their concentration in precipitations according to the directions of advection (Twarowski *et al*. 2002). Other causes may have been connected to the local natural and anthropogenic conditions which get changed together with the seasons of the year.

Table 1. Annual loads of TP and TN from wet deposition and the sums of annual precipitations (P) in Chojnice in the years 2000-2003.

Year	TN ($kg \cdot ha^{-1}$)	TP ($kg \cdot ha^{-1}$)	P (mm)
2000	9.58	0.453	586.9
2001	10.22	0.372	810.8
2002	9.78	0.520	739.1
2003	5.72	0.164	465.3

The monthly loads of phosphorus and nitrogen from wet deposition showed considerable diversity. Their greatest values were recorded in the summer months, and the lowest in the winter months. This dependency is particularly noticeable in the case of phosphorus. In the analysed period a minor negative trend of phosphorus and nitrogen deposition was noted, particularly as a result of its low value in 2003 (Figure 3).

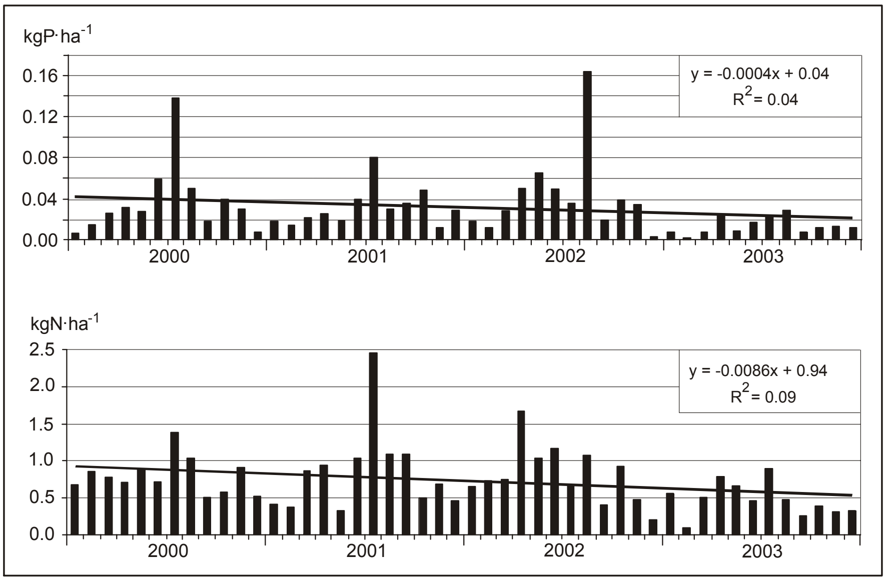


Figure 3. Monthly loads of phosphorus (P) and nitrogen (N) from wet deposition at the monitoring station in Chojnice in the years 2000-2003. Based on the data obtained within the State Environmental Monitoring from the Inspection for Environmental Protection and the Institute of Meteorology and Water Management.

Despite a smaller investigation area (3840 km^2), it was possible to notice greater spatial diversity in the annual values of phosphorus loads (from approx. 11% in 2002 to 68% in the following year) and significantly lower ones in the case of nitrogen (from approx. 4% in 2000 to 23% in 2003). Moreover, there were key differences in the mean annual loads for the entire study area, which in the case of phosphorus ranged from 0.027 to 0.053 $g \cdot m^{-2}$ (i.e. 95%), and from 0.70 to 1.19 $g \cdot m^{-2}$ (i.e. 70%) in the case of nitrogen. A complex spatial and quantitative system of the loads of phosphorus and nitrogen from wet deposition constituted a major cause for considerable diversity in their portion in the permissible load for the particular lakes (Table 2). This share, expressed in % as the rate of using the permissible load (ULp), amounted from 14.2 to 205.4% in the case of phosphorus and from 34.6 to 359.4% in the case of nitrogen (Figure 4).

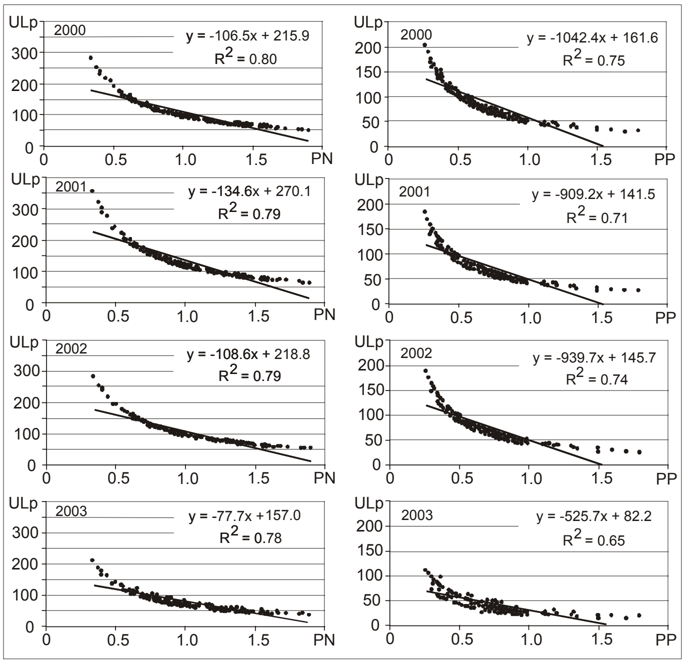


Figure 4. Dependencies between the rate of using the permissible load (ULp in %) and the permissible loads of phosphorus (PP) and nitrogen (PN) for the analysed lakes (permissible load = 1.0).

It is worth noting that the mean value of ULp of nitrogen for all 137 lakes was slightly greater than 100% in the years 2000-2002 (Table 2). This meant that the annual permissible load of nitrogen was (on average) abused only because of nitrogen supplied together with precipitation. The share of phosphorus load from wet deposition in the permissible load was considerably lower and equalled from 44.9% in 2003 to 87.6% in 2000.

Table 2. Ranges (R) and mean values (M) of mean annual loads of phosphorus and nitrogen coming from precipitation into the lakes (Lo) and the rate of using the permissible load (ULp) in the years 2000-2003 (ULp 100% = permissible load).

Year	Phosphorus				Nitrogen			
	Lo ($g \cdot m^{-2}$)		ULp (%)		Lo ($g \cdot m^{-2}$)		ULp (%)	
	R	M	R	M	R	M	R	M
2000	0.050-0.057	0.053	29.3-205.4	87.6	0.95-0.99	0.97	51.9-285.6	105.0
2001	0.042-0.048	0.046	26.3-184.6	76.4	1.15-1.23	1.19	61.2-359.4	129.0
2002	0.044-0.049	0.047	26.4-188.5	79.0	0.94-1.01	0.96	49.4-282.9	104.3
2003	0.021-0.036	0.027	14.2-112.7	44.9	0.65-0.80	0.70	34.6-209.7	75.6

The mean depths, which amounted from 0.9 to 15.5 m, were another cause of the considerable diversity in the share of phosphorus and nitrogen loads from wet deposition in the permissible load for particular lakes. In the years 2000-2002 the permissible load of phosphorus was exceeded in almost all the lakes with the mean depth lower than approx. 3.5 m. In 2003 due to the substantial decline of phosphorus concentration in precipitation, the permissible load was exceeded only in the two shallowest and smallest lakes with the mean depth 0.8 m and 0.9 m. As for nitrogen in the years 2000-2002, its permissible load was exceeded in the definite majority of the lakes whose mean depth was lower than 4.5 m, including over 2.5 times lower than in the shallowest lake, Płocice. A year later, due to the substantially lower nutrients deposition, the abuse of the permissible load occurred only in several shallow lakes with the depth of up to 3.2 m. The data concerning the load of nitrogen from wet deposition in the differentiated depth classes of the lakes are presented in Table 3.

Table 3. Comparison of nitrogen loads from wet deposition to the loads permissible for the lakes (mean values). Symbols: N-number of the lakes, U_N-number of the lakes with the abused permissible loads, ULp-the rate of using the permissible load in % (permissible load = 100%).

Depth class (m)	N	2000		2001		2002		2003	
		U_N	ULp	U_N	ULp	U_N	ULp	U_N	ULp
<1.5	7	7	234.4	7	291.3	7	233.1	7	168.6
1.5–3.0	22	22	146.0	22	178.9	22	144.7	12	104.1
3.1–5.0	46	39	109.7	46	132.0	38	108.3	2	79.0
5.1–7.5	26	0	84.2	15	103.4	26	83.9	0	60.9
7.6–10.0	20	0	67.9	0	82.4	0	68.6	0	48.1
10.1–15.0	13	0	60.2	0	73.7	0	60.5	0	43.6
>15.0	3	0	52.6	0	63.6	0	51.1	0	38.8

The threat of phosphorus from wet deposition was noticeably lower. In the years 2000-2002 the permissible load was exceeded only in those lakes whose mean depth was <3 m. In 2003 the permissible load was not abused in any lakes, and in the case of the deepest lakes (mean depth >10 m) it barely exceeded 20% (Table 4).

Table 4: Comparison of phosphorus loads from wet deposition to the loads permissible for the lakes. Symbols: N-number of the lakes, U_N-number of the lakes with the abused permissible loads, ULp- the rate of using the permissible load in % - mean value (permissible load = 100%).

Depth class (m)	N	2000		2001		2002		2003	
		U_N	ULp	U_N	ULp	U_N	ULp	U_N	ULp
<1.5	7	7	175. 7	7	156.3	7	161.4	2	92.3
1.5–3.0	22	22	133.6	22	116.1	22	120.3	0	67.1
3.1–5.0	46	10	91.9	3	80.1	5	83.3	0	47.2
5.1–7.5	26	0	70.0	0	64.3	0	62.9	0	36.5
7.6–10.0	20	0	57.7	0	51.2	0	52.0	0	28.2
10.1–15.0	13	0	40.6	0	36.2	0	36.5	0	21.7
>15.0	3	0	40.5	0	34.6	0	35.3	0	21.1

Discussion

The presented results indicate that the concentration of phosphorus and nitrogen in precipitation is highly diverse both in particular months and years. Hence, their loads are also different, even over a small area. They perform, however, a big role in supplying the lakes (and also other ecosystems) with biogenic compounds. In reality, the role is even bigger and this study discusses only the scale of their direct deposition into the lakes. It does not mention their deposition into the lakes' catchments.

Shallow lakes, whose mean depths did not go over 3.0-4.5 m were found to be mostly exposed to the negative influence of phosphorus and nitrogen from precipitation. A similar conclusion was presented with respect to the biggest lakes in Poland (>100 ha) examined in the years 1999-2001 (e.g. Blachuta J. & Twarowski R., 2003, unpublished data). The above assumptions are crucial to the discussed lakes as over 40% of them have smaller mean depths. This issue can be illustrated correspondingly in the case of the lakes located in the neighbouring lakelands. If the permissible loads are considered, nitrogen presents a bigger threat to the lakes. In the case of the shallowest lakes (mean depth <1.5 m), the load of nitrogen from precipitation even abused the value of the critical load more frequently, particularly in the years 2000-2002.

However, in recent years in some parts of Europe the deposition of phosphorus, nitrogen and other substances from precipitation has been declining as a result of restraining the emission of pollution into the atmosphere (among others: Skjelkvåle *et al*. 2001; Sopauskiene *et al*. 2001; Rekolainen *et al*.

2005). If this process turns to be continuous further restraint of the concentration of nutrients in lake waters may be expected. Moreover, polluted sewage discharge into the lakes has been stopped. This should limit, and eventually decline the degree of lake eutrophication, which is the biggest limnological problem not only in this part of Europe.

References

Bennett E.M., Reed-Andersen T., Houser J.N., Gabriel J.R. & Carpenter S.R., 1999. A Phosphorus Budget for the Lake Mendota Watershed. *Ecosystems* 2, 69-75.

Bootsma H.A., Mwita J., Mwichande B., Hecky R.E., Kihedu J. & Mwambungu J., 1999. The Atmospheric Deposition of Nutrients on Lake Malawi/Nyasa. In: *Water Quality Report. Lake Malawi/Nyasa Biodiversity Conservation Project* (eds. H.A.

Bootsma & R.E. Hecky. pp. 24-50. Southern African Development Community, Global Environmental Facility.

Delumyea R.G. & Petel R.L., 1978. Wet and dry deposition of phosphorus into Lake Huron. *Water, Air, and Soil Pollut.* 10, 187-198.

Driscoll C.T., Poster K.M., Kretser W. & Raynal D.J., 1995. Long-term trends in the chemistry of precipitation and lake water in the Adirondack Region of New York, USA. *Water, Air, and Soil Pollut.* 85, 583-588.

Morales J.A., Albornoz A., Socorro E. & Morillo A., 2001. An estimation of the nitrogen and phosphorus loading by wet deposition over Lake Maracaibo, Venezuela. *Water, Air, and Soil Pollut.* 128, 207-221.

Nõges P., Järvet A., Tuvikene L. & Nõges T., 1988. The budgets of nitrogen and phosphorus in shallow eutrophic Lake Võrtsjärv (Estonia). *Hydrobiol.* 363, 219-227.

Rekolainen S., Mitikka S., Vuorenmaa J. & Johansson M., 2005. Rapid decline of dissolved nitrogen in Finnish lakes. *Journal of Hydrology* 304, 94-102.

Shaw R.D., Trimbee A.M., Minty A., Fricker H. & Prepas E.E., 1989. Atmospheric deposition of phosphorus and nitrogen in Central Alberta with emphasis on narrow Lake. *Water, Air, and Soil Pollut.* 43, 119-134.

Skjelkvåle B., Tørseth K., Aas W. & Andersen T., 2001. Decrease in acid deposition – recovery in norwegian waters. *Water, Air, and Soil Pollut.* 130, 1433-1438.

Sober R.F. & Bates M.H.: 1979. The atmospheric contribution of phosphorus to an aquatic system. *Water, Air, and Soil Pollut.* 11, 63-69.

Sopauskiene D., Jasineviciene D. & Stapcinskaite S., 2001. The effect of changes in european anthropogenic emissions on the concentrations of sulphur and nitrogen components in air and precipitation in Lithuania. *Water, Air, and Soil Pollut.* 130, 517-522.

Tamatamah R.A., Hecky R.E. & Duthie H.C., 2005. The atmospheric deposition of phosphorus in Lake Victoria (East Africa). *Biogeochem.* 73, 325-344.

Twarowski R., Blachuta J., Gendolla T., Liana E. & Sienkiewicz R., 2002. Spatial distribution for the deposition of pollutants from precipitation over Poland according to the directions of advection. *In.* J. Burchard ed. *Stan i antropogeniczne zmiany jakosci wod w Polsce*, pp. 33-40, Wydawnictwo Uniwersytetu Lodzkiego, Lodz.(in Polish, summ. in English)

Vollenweider R.A., 1968. *The scientific basis of lake and stream eutrophication with particular reference to phosphorus and nitrogen as eutrophication factors*. Technical Report DAS/C81/68, OECD, Paris.

Effects of land use changes on bird composition along River Njoro: A watershed of Lake Nakuru

Faith Milkah Ngugi[1], W. A. Shivoga[1], Mucai Muchiri[2] and Scott Miller[3]

[1]Egerton University, Environmental Science Deparment, P.O. Box 536, Njoro, Kenya
[2]Moi University, Fisheries Department, .P.O. Box 3900, Eldoret, Kenya
[3]University of Wyoming, Rangeland Ecology and Watershed Management, Department 3354,000 E. University Avenue, Laramie, WY 82071, USA
Email: fngugi2001@yahoo.com

Abstract

The rift valley lakes and their associated watersheds are home for millions of resident and migratory waterfowl. However, they have been recently challenged by land use changes. Lake Nakuru supports diverse biological resources of global, regional and national importance. It is saline and River Njoro is its main source of freshwater. Deforestation, cultivation and urbanization in vital watershed areas have altered the hydrological regime of River Njoro. There is need to preserve and restore aquatic, riparian and terrestrial biological diversity in the watershed to be able to restore and maintain ecological health of Lake Nakuru. The impact of these recent land use changes on birds in the River Njoro Watershed is of concern to conservationists. This study aimed at giving basic information on the status of bird community in the River Njoro watershed. The upper river course was surveyed. The different land uses along the upper river course were assessed and classified according to the quality of the available habitat. A total of seven sampling sites were studied and observations made. Bird species were identified, counted and recorded. The data was compared among the different land use and correlated to the quality of the habitat. The data revealed that species are distributed largely on the basis of microhabitats. Subsequently, sites with higher number of microhabitats recorded higher species composition. Overall, the upper River Njoro Watershed has a high diversity of birds (>0.90 Simpson indices). Even then, there was a significant difference between sites along the river depending on riparian land use ($p<0.05$). Harmonious integration of land uses in the watershed is the underline for high diversity in the watershed.

Key words: Birds, Landuse, River Njoro, watershed

Introduction

In east Africa, the rift valley lakes and their associated watersheds are recognized as key resources in the maintenance of global biodiversity. They are home to unique aquatic life and serve as breeding and/or staging grounds for millions of resident and migratory waterfowl. The Rift Valley lakes and their inflowing rivers however, have been recently challenged to a high degree by land use changes that threaten their ecological integrity.

Lakes are not isolated entities, but are linked and associated to both natural and human induced processes and activities taking place in their catchments (Kenya Wildlife Society, 2002). The catchments of Lake Nakuru is a unique ecosystem containing a variety of habitats that include a shallow strongly alkaline lake, the largest Euphorbia forest stand in East Africa, a wildlife rich savannah and highland moist forest. The lake supports diverse biological resources that are of global, regional and national importance. It is a centerpiece of a world-renowned national park famous for its spectacular bird life, particularly the Lesser Flamingos (*Phoeniconaias minor*), Greater Flamingos (*Phoenocopterus ruber*) and Great Pelicans (*Pelecanus onocrotalus roseus*). It forms a major ground for feeding, staging and displaying ground for the region's millions of lesser and greater flamingos together with over 400 species of other birds -often referred to as "the greatest ornithological spectacle on earth" (Mohamed & Earnes, 1987). It is protected as a National Park, designated as a Ramsar Site and an Important Bird Area (IBA).

River Njoro is the main source of fresh water supply for the hyper saline Lake Nakuru. Its watershed lies within the latitudes $0^o15'$S and $0^o25'$S and between longitudes $35^o50'$E and $35^o05'$E. River Njoro is about 50km long with a catchment's area of about 280km^2. It originates in the Eastern Mau Escarpment, with two main tributaries; flows passed forested and highly human settlement areas and intensively cultivated land and then empties into Lake Nakuru. The watershed River Njoro drains is highly disturbed and its land use cover is in rapid transition (Baldyga *et al.*, 2004). This transition has resulted in conversion of previously natural and plantation forest land to small-scale agricultural, grazing and urban uses. For example, between 1986 and 2003, there has been an increase of 26% in land area used for small-scale agriculture and pasture (Baldyga *et al.*, 2004). This transition has also led to lose of 10% and 9% in indigenous and plantation forests respectively.

Degradation of watersheds generally diminish their capacity to provide critical ecosystem functions, including the cycling and chemical transformation of nutrients, purification of water, attenuation of floods, maintenance of stream flows and stream temperatures, recharging of underground water, and establishment and maintenance of habitat for fish and wildlife (Kauffman *et al* 1997). The impact of these recent land use changes in and lose of forest habitats on birds in the River Njoro Watershed has not been investigated but is of concern to local conservationists and environmentalists. Recent research in the Lake Naivasha Basin indicate that bird species composition is at least related to the composition of plant species and the closeness of riparian zones to human settlements (Everard *et al*., 2002).The study on effects of land use changes on bird composition along river Njoro was undertaken as part of a wider research program aimed at testing if birds can be used as a community of organisms for assessing impacts in watersheds. It would be interesting to know the pattern of change in bird species composition, abundance and diversity in relation to land use change and habitat disturbance in River Njoro Watershed. The specific objectives of the study were to determining birds species composition and the relationship between diversity, abundance of the birds, and habitat structure in the upper River Njoro Watershed

Materials and methods

A total of seven sites that represent different riparian land uses on the upper reaches of River Njoro watershed were sampled. The site were Logoman, Ruguma, Sigaon, Nessuit, Sigotik, Forest camp and Confluence. Logoman is at high altitude and has a patch of indigenous forest and plantation forest which are quite disturbed, Ruguma has a grassland and cultivated land, Sigaon is mainly a bamboo-forest-grassland mosaic and cultivated land. Nessuit, Sigotik and Confluence are basicly surrounded by cultivated land. Forest camp has patches of plantation and indigenous riparian forests that are guarded, on the west of the forest patch is grass strip followed by a long strech of cultivated land which is far from human settlement.

To compare sites, a modified Qualitative Habitat Suitability Index (QHSI) based on availability of potential bird micro-habitats (i.e. grassland, indigenous forest, plantation forest, regenerating forest, and cultivated land) along the riparian corridors of first order streams in River Njoro Watershed was developed. To arrive at a site QHSI, a score was given depending on number of microhabitats at a site as shown by Table 1. At each site a transect across the riparian strip covering both sides of the river was established each for a particular habitat type. The researcher walked along this transect and identified and counted bird individuals belonging to every species encountered.

Table 1. showing the site scores for QHSI.

SITES	**MICROHABITATS**					
	Grassland	Indigenous Forest	Plantation Forest	Regenerating Forest	Cultivated Land	**SITE SCORE**
Logoman	✓	✓	✓	✓		**4**
Ruguma	✓				✓	**2**
Sigaon	✓	✓			✓	**3**
Sigotik					✓	**1**
Nessuit		✓			✓	**2**
Forest camp	✓	✓	✓		✓	**4**
Confluence					✓	**1**

Data was summarised into a species composition list for a site. Simpson diversity index (Brower *et al.*, 1990) was then used to estimate diversity of birds at a site. To compare sites, all individuals encountered were summed as a percentage of the total population of birds encountered per site. Non parametric Kruskal-Wallis Analysis of Variance was used to test for differences in diversity and abundance between sites. A linear regression analysis was performed to test the hypothesis that bird diversity and abundance are dependent on habitat structure.

Results and discussion

Bird species composition

A total of 533 birds distributed among 66 species were encountered in the upper river watershed over the study period. A closer look at the data revealed that species are distributed largely on the basis of microhabitats. Subsequently, sites with higher number of microhabitats recorded higher species composition.

The birds that were present in the entire seven sites sampled were from the families- Muscicapidea, Flingillidae, Pycnonotidae, and Nectarinidae. Forest camp site had the highest number of bird species with over 65% of all the bird species sampled present. 13 species out of 66 were exclusively found at Forest camp site this include *Batis molitor, Chrysococcyx klaas, Corvus capensis, Dendropicos goertae rhodeogaster, Estrilda quartinia kilimensis, Euplectes capensis, Merops pusillus cyanostictus, Myrmecocichla aethiops*, *Ploceus pelzelni*, *Pogoniulus bilineatus, Serinus sulphuratus, Terpsiphone viridis, Uraeginthus bengalus.*

At Logoman 4 species including *Buphagus erythrorhynchus* which is associated with livestock, a highland Swift *Apus niansae*, and a high altitude Canary *Serinus canicollis flavivertex*, were found exclusively. *Motacilla flava* which is a paleartic migrant was among the 4 species that were only encountered at Sigotik. A norcturnal species (*Caprimulgus poliocephalus)* was also recorded only at Nessuit. Ruguma, Sigaon and Confluence sites had the least number of species recorded although Sigaon had two species that occurred exclusively there i.e. the White Starred Robin *Pogonocichla stellata intense* and the Black-backed Puffback *Dryoscopus cubla.* Sigaon, the uppermost point on Little shuru stream cannot compare with Logoman since its natural vegetation is bamboo-forest-grassland mosaic which is known to be impoverished avifauna (Bennun & Njoroge, 1999).

Community structure

Out of 533 birds that were encountered in this study, 24% were recorded at Logoman (Figure 1). Nearly a similar percentage was recorded at forest camp. The rest were divided almost equally among the remaining sites. The trend on the two streams is similar. The sites on Enjoro stream are: 1-logoman, 2-Ruguma, and 3-Sigotik while on Little Shuru are 1-Sigaon, 2- Nessuit, 3-Forest camp and Site 4 is the Confluence site

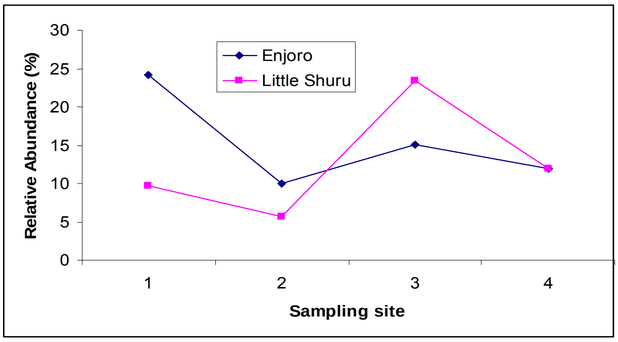


Figure 1. distribution of bird individuals encountered in the upper River Njoro.

Overall, the upper River Njoro Watershed has a high diversity of birds as exemplified by Simpson indices (all >0.90). Even then, there was a significant difference between sites along the river depending on riparian land use ($p<0.05$). Along Enjoro, there was a general but insignificant decline in bird species diversity along the river going downstream. Along the Little Shuru tributary, the upper most sites (Sigaon and Nessuit) had a low diversity. This however rose to the highest at Forest Camp (Figure 2) before declining to its lowest at the Confluence site.

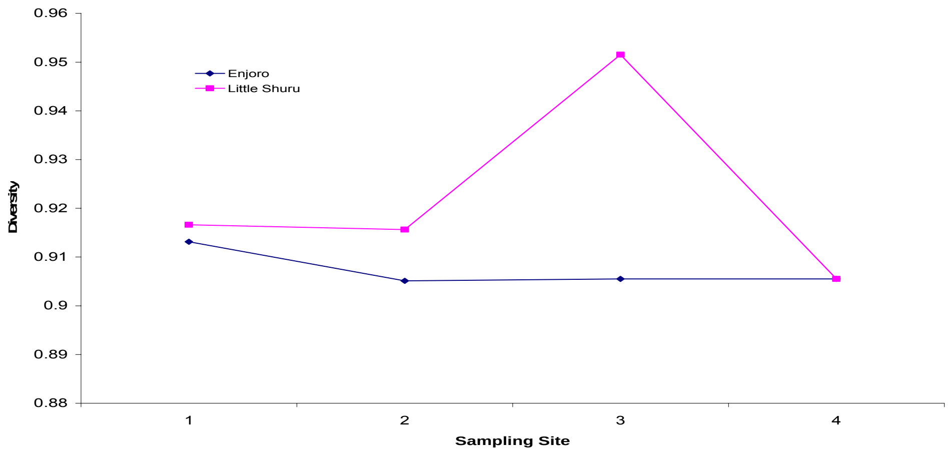
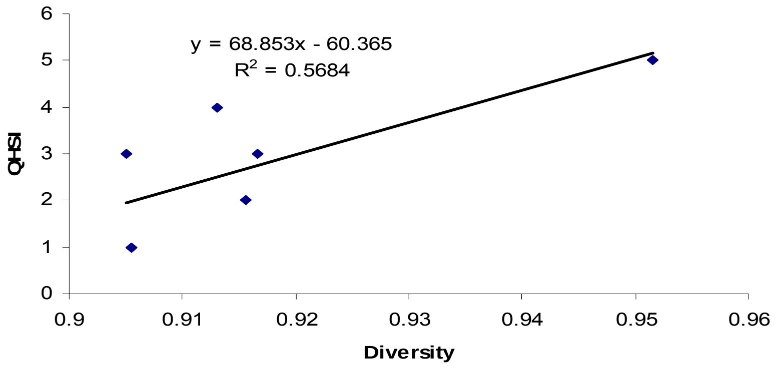


Figure 2. diversity of birds in the various sites of upper River Njoro watershed.

Relationships

Regression analysis showed that diversity and relative abundance of birds in the River Njoro Watershed depend significantly on habitat structure (Figure 3). This shows that the more the bird microhabitats, the more the number of birds likely to be encountered at a site.



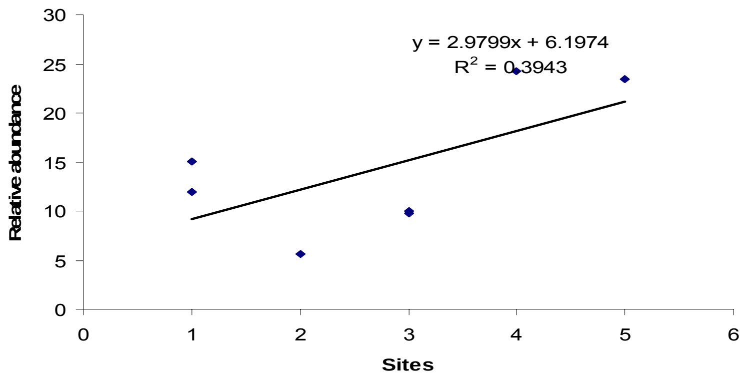


Figure 3. regression curves showing relationships between bird species diversity and relative abundance with a measure of habitat structure.

Conclusion

This study agrees with the general ecological concept that a higher diversity of habitats leads to higher diversity of species (Brower *et al* 1990). The mere existence of a forest does not necessarily mean there will be more birds. Harmonious integration of land uses might be the underlying secret for high bird diversity in the watershed. There is need for a more elaborate study to establish if actually the species encountered exclusively at sites are characteristic of land use changes or some yet to be measured factor.

Acknowledgements

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References

Baldyga T.J., Miller S.N., Shivoga W.A. & Gichaba C.M., 2004. Land cover change detection in the river Njoro watershed: a landscape in transition. Proceedings of the 57th annual meeting of the society for range management, January 24th-30th, 2004. *Salty lake city*, UT

Bennun L. & Njoroge P., 1999. *Important Bird Areas in Kenya*. Nairobi: Nature

Bennun L. & Waiyaki E.M., 1992a. Using birds to monitor environmental changes in the Mau Forests *Research Reports of the Center for Biodiversity, National Museums of Kenya: Ornithology **2**.*

Brower J.E., Zar J.H., & Ende C.N., 1990. *Field and Laboratory Methods for General Ecology.* 3rd ed. WCB Publishers

Everard M., Kuria A., Macharia M., Vale, J.A., & Harper, D.M. (2002) Aspects of the biodiversity of the rivers in the Lake Naivasha catchment. *Hydrobiologia* **488**: 43-55

Kauffman J.B., Beschta, R.L., Otting, N. & Lytjen, D., 1997. An ecological perspective of riparian and stream restoration in Western United States. *Fisheries* **22**(5): 12-24

Kenya Wildlife Society, 2002. Lake Nakuru integrated ecosystem management plan 2002-2012. Kenya Wildlife Society and Government of Kenya, Nairobi
Kenya.

Mohamed A.,& Eames J., 1987. *Insight Guides: Kenya*. APA Productions

Zimmerman D.A., Turner, D.A. & Pearson D. J., 1996. *Birds of Kenya and Northern Tanzania.* Halfway House: Russel Friendman Books.

Toxic cyanobacteria occurrence in Lake Victoria-Uganda

W.O. Okello[1], J.S. Balirwa[1], F.W.B. Bugenyi[2] and R. Kurmayer[3]

[1]Fisheries Resources Research Instititute, PO Box 343, Jinja, Uganda
[2]Department of Zoology, Makerere University, PO Box 7062, Kampala, Uganda
[3]Austrian Academy of Sciences, Institute for Limnology, Mondseestrasse 9, A-5310, Austria
Email: wiokello@yahoo.com

Abstract

While cyanotoxins studies are on-going in temperate regions, little is known about its occurrence in Lake Victoria reported to be eutrophic with increased cyanobacterial blooms. An investigation carried out in 2004 characterised cyanobacterial species composition in relation to the environmental conditions and cyanotoxin production in Lake Victoria (Murchison Bay and Napoleon Gulf) and the satellite water-body of Lake Nabugabo. Species determination was done microscopically and 41 strains were isolated using two different media for bioassay and the determination of cyanotoxins. Chemical composition of microcystins and other unidentified peptide like compounds were determined by high performance liquid chromatography. Toxicity of strains and field samples was determined using the *Thamnocephalus platyurus* bioassay. Nine major cyanobacteria species: *Anabaena, Aphanocapsa, Merismopedia, Microcystis, Oscillatoria, Planktolyngbya tallingi, Planktolyngbya limnetica, Pseudanabaena and Spirulina* were recorded. Cyanobacteria were dominant in Lake Victoria but Desmidiaceae dominated Lake Nabugabo samples. Two cultures comprising of *Anabaena* and *Pseudanabaena* contained small amounts of unknown variants of microcystin. Correspondingly, two field sample extracts contained microcystin (Lake Victoria: Murchison Bay and Napoleon Gulf). All cultures and field samples contained unidentified peptide-like and cyanopeptide compounds. There was no correlation between the occurrence of microcystin and the observed toxicity in *Thamnocephalus* bioassay neither with cultures nor with the field samples. Lake Nabugabo showed lower toxicity even in concentrated plankton net samples. Although cyanobacteria tend to proliferate under eutrophic conditions and nine cyanobacteria species dominated the phytoplankton communities in Lake Victoria, the observed levels of microcystin production were lower than those reported in the literature for temperate lakes. The mismatch between microcystin occurrence and the toxicity observed suggests production of possibly unidentified toxic compounds in Lake Victoria and its catchment area. *Thamnocephalus* bioassay is recommended as a quick and easy-to-use bio-monitoring tool even in water bodies containing unknown cyanobacterial toxins.

Key words: bioassay, cyanobacteria, microcystin, Lake Victoria, Uganda

Introduction

Cyanobacteria toxins are amongst the most potentially hazardous natural substances in surface waters used by humans. Cyanobacteria toxins are a diverse group of natural toxins, both from the chemical and the toxicological aspects and occur as, cyclic peptides, alkaloids and lipopolysaccharides (Chorus and Bartram 1999). The most frequently found toxins in algal blooms from freshwaters are the cyclic peptide toxins of the microcystin and nodularin family. Cyanotoxins now occur with unnatural frequency and concentration. Sivonen and Jones (1999) estimated that the frequency of occurrence of toxicity in cyanobacterial bloom was on average 59% world over. The toxins pose a major challenge to the production of safe drinking water from surface sources containing cyanobacteria.

Microcystins have been characterised from planktonic *Anabaena, Microcystis, Oscillatoria (Planktothrix), Nostoc*, and *Anabaenopsis* species while Nodularin only from *Nodularia spumigena* (Chorus and Bartram 1999; Haider *et al*. 2003). Alkaloids that include anatoxin-a, anatoxin-a (S) and saxitoxins have been reported from North America, Europe and Australia where they have caused animal poisonings. Anatoxin-a has been found in *Anabaena, Oscillatoria* and *Aphanizomenon*, homoanatoxin-a from *Oscillatoria*, anatoxin-a (S) from *Anabaena* and saxitoxins from *Aphanizomenon*, *Anabaena*, *Lyngbya* and *Cylindrospermopsis* (Chorus and Bartram 1999). Lipopolysaccharides (LPS) were first isolated by Weise *et al*. (1970) from the cyanobacterium *Anacystis nidulans* since then, numerous reports of endotoxins in cyanobacteria have followed. LPS are an integral part of the cell wall of all Gram-negative bacteria, including cyanobacteria, and can elicit irritant and allergic response in human and animal tissue that come in contact with the compounds. Cyanobacterial LPS however are less potent than that from pathogenic gram-negative bacteria like *Salmonella*. More studies though are still needed to evaluate the chemical structures and health risks of cyanobacterial LPS (Chorus and Bartram 1999).

While cyanopeptides are currently being studied in Europe, (Chorus 2004) virtually nothing is known about cyanopeptide production in freshwater of tropical countries with few papers published (Sekadende *et al* 2005 and Krienitz *et al* 2002) on Lake Victoria. This is not only because the species distribution differs between temperate and tropical regions, but also cyanopeptide production has been shown to be strain-specific that cannot be differentiated under the microscope.

Most of the Ugandan water bodies (20% of the land area) (Fig. 1) are experiencing increasing pressure from rapid population growth, increased industrial activities, environmental degradation, soil erosion, drainage of wetlands and pollution (DWD 2004). These pressures have led to eutrophication resulting into increased cyanobacterial blooms especially in Lake Victoria (Kling *et al*. 2001; Lungaiya *et al*. 2000; Mugidde 1993; Ochumba and Kibaara 1989). Some of the cyanobacteria found have been reported to be toxic (Sivonen and Jones 1999). This research carried out from April to September 2004 characterised cyanobacteria species composition and toxicity in Lake Victoria-Uganda and the satellite Lake Nabugabo in the same catchment area but along a trophic gradient.

Materials and methods

Samples were taken in the month of May and June 2004 from Lake Victoria sites: Ggolo, Murchison Bay (N00°16.911' E032°38.398'), Napoleon Gulf, (N00° 24.411' E033°13.132'). To signify the gradient in trophy, Lake Nabugabo (N00°21.283' E031°52.789') was also sampled (Figure 1).

Nutrients were analysed according to Wetzel and Likens (2002) while phytoplankton species composition and abundance were determined by lugol counting (Utermöhl 1958). Cyanobacteria were isolated and purified following the Rippka (1988) method before harvest onto pre-weighed glass fibre filters of type C and oven dried at 50°C for three days. The harvested dried cells on filters were later extracted for analyses using HPLC (Fastner *et al*. 1998; Lawton & Edwards 2001). Part of the extract was used in the *Thamnocephalus platyurus* bioassay experiment (Kurmayer & Jüttner 1999) based on the reduction in survival rate expressed as percentage mortality. For external validation of these results on microcystin occurrence, four filters from May integrated water samples were analysed in Oslo, Norway using liquid chromatography-mass spectrometry (LC-MS) which has a lower detection limit of 27ng L^{-1} MC-LR in water in comparison to HPLC-DAD (Spoof *et al*. 2003).

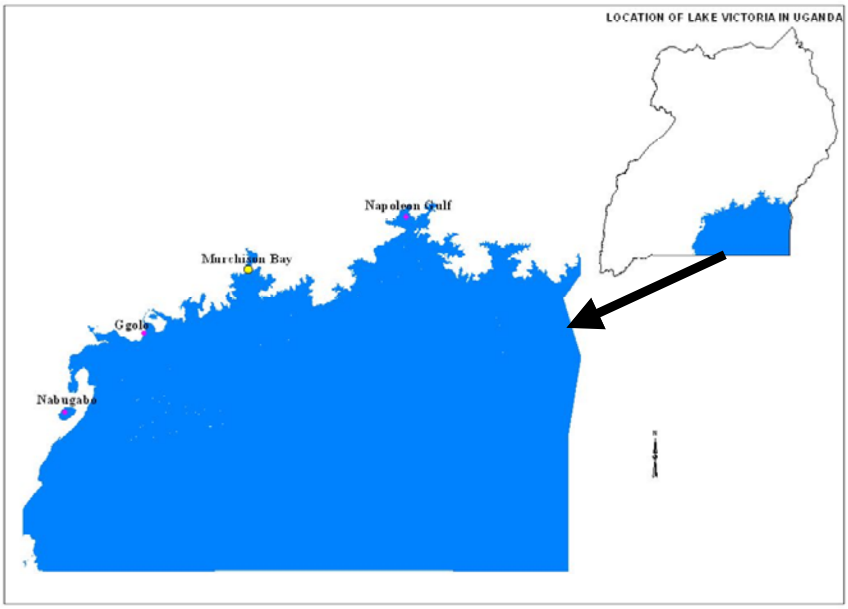

Figure 1. Sampling sites for both environmental parameters and cyanobacteria in Lake Victoria (Ggolo, Murchison Bay & Napoleon Gulf) and Lake Nabugabo.

Results

There was a gradient in trophy among the sampling sites, ranging from the mesotrophic (Lake Nabugabo) to eutrophic conditions (Murchison Bay) (Figure 2). High total phosphorus (TP) values were recorded from Lake Victoria site: Murchison Bay in comparison to Lake Nabugabo. NO_3-N followed the same trend also.

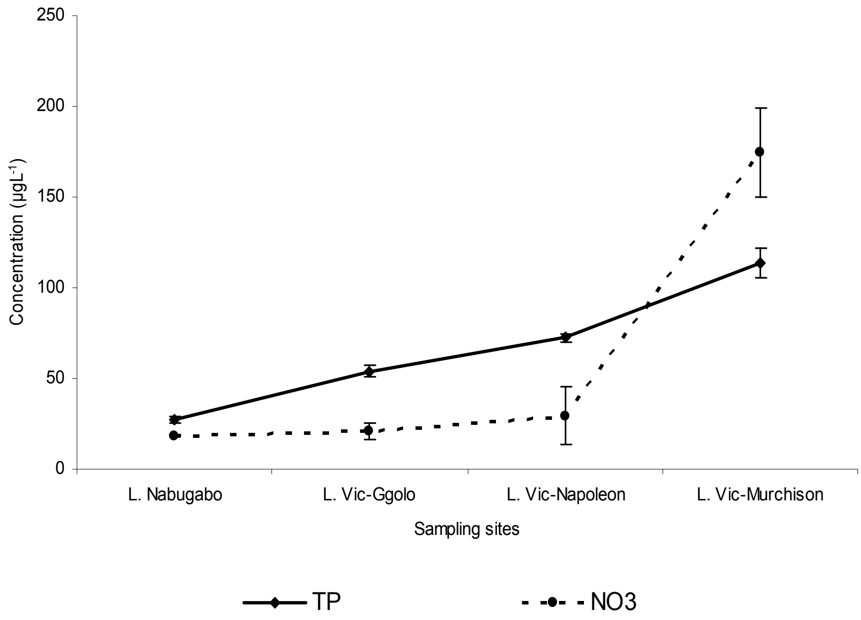

Figure 2. Variation in mean total phosphorus (TP) and nitrate (NO_3) from Lake Nabugabo, Lake Victoria (Ggolo, Napoleon gulf and Murchison bay) in 2004.

Four cyanobacteria species: *Anabaena, Aphanocapsa, Microcystis* and *Planktolyngbya* contributed greatly to the biovolume. In between the two months, there were small changes in total biovolume. *Microcystis* biovolume dropped from almost all the sampling sites while *Aphanocapsa* increased in the three sites except in Napoleon Gulf (Table 1). A total of 62 filters were extracted of which, 41 were from the cultures. Only two of these cultured extracts showed presence of microcystin. The microcystin producers were *Anabaena* and *Pseudanabaena* (Table 2). The rest of the cultures had other peptide like compounds present. There was no single strain without any peptide like compound. Among the 21 field samples extracted, only two net samples showed the presence of microcystin (Table 3). Microcystin was not detected in integrated samples. In Murchison Bay with a dominance of *Anabaena* in May low microcystin concentrations was recorded. No microcystins was detected in June when *Anabaena* biovolume was highest. Both the cultured and field extracts were highly toxic to the *Thamnocephalus* larvae (Tables 2 and 3). 100% mortality implied that all the larvae could not survive in the extract for 24 hours under incubation.

Microcystin (MC-LR) was detected only from L. Victoria Murchison Bay site and Napoleon Gulf May samples. These were in harmony with the finding from Oslo, Norway. Consequently, the result obtained by LC-MS in another laboratory on the same samples was in close agreement to the results obtained during this thesis. We could then say microcystin concentrations expressed per biovolume and per liter in Lake Victoria-Uganda could be very low.

Table 1. Cyanobacterial biovolume (mm^3L^{-1}) from Lake Nabugabo and Lake Victoria sites: Ggolo, Napoleon Gulf and Murchison Bay during the months of May and June 2004.

	Anabaena		***Aphanocapsa***		***Microcystis***		***Planktolyngbya***	
Lake	May	June	May	June	May	June	May	June
Nabugabo	1.13	1.09	1.60	1.41	1.44	0.72	0.94	0.79
Victoria-Ggolo	1.53	2.82	6.52	16.61	0.72	1.22	0.96	1.65
Victoria-Napoleon	6.83	15.07	27.96	26.36	4.48	3.07	0.16	0.68
Victoria-Murchison	8.52	89.94	8.31	31.15	1.24	7.57	0.10	0.33

Table 2. Cultures of cyanobacteria showing presence of microcystin (MC-LR) expressed in equivalents microgram (µg) per mg of dry weight (DW) and their level of toxicity.

Date of culture	**Lake**	**Species**	**Dry weight (mg)**	**MC-LR equivalents µg/mg DW**	**Toxicity**
18-May-04	Nabugabo	*Pseudanabaena*	6.2	0.34	100%
18-May-04	Murchison Bay	*Anabaena*	6.0	0.25	100%

Table 3. Field extract showing presence of microcystin and their level of toxicity.

Date of sampling	Lake	Sample type	Dry weight (mg)	MC-LR equivalents µg/mg DW	Toxicity
08-May-04	Victoria-Murchison Bay	Net	20.5	0.03	100%
09-May-04	Victoria-Napoleon Gulf	Net	15.7	0.20	100%

Discussion

The low mean concentration of total phosphorus, nitrate coupled with a phytoplankton community mainly dominated by Desmidiaceae in Lake Nabugabo probably account for the low cyanobacteria biovolume. In comparison to Lake Victoria especially in Murchison Bay, Lake Nabugabo may be considered as presently being less impacted by human activities with low nutrient loading. Murchison Bay and Napoleon Gulf exhibited conditions that highly promote cyanobacteria proliferation (Baldia *et al*. 2003) in addition to other factors like population growth, increased industrial activities and drainage of wetlands. Cyanobacteria clearly were favoured by eutrophication with the two sites in Lake Victoria showing high nutrient concentrations. Consequently, if Lake Nabugabo were to become similarly eutrophic, the phytoplankton composition is predicted to be dominated by cyanobacteria.

The two cultured extracts made up a minor part of the total isolate collection and showed presence of microcystin at two sites out of the four sites sampled. These extracts all came from May sampling. In this study, *Pseudanabaena* was found to produce microcystin that was also toxic thus eliminating the original thinking of not producing microcystins. This finding is in agreement with Oudra *et al.* (2002) who reported two *Pseudanabaena* strains that produced microcystin even though the identified microcystins still need validation using advanced mass spectrometric methods described by Kurmayer *et al*. (2004). The toxicity analyses of both culture and field extracts indicated that microcystins were not solely responsible for the observed toxic effects to *Thamnocephalus* in Lake Victoria.

In the field only two net samples were shown to have low concentrations of microcystin. During the set up of the field sampling programme, the net sampling in addition to integrated lake sampling was used as a tool to obtain lake samples representative of the lake and to search for microcystin producers of low abundance. In most samples no microcystin could be found, not even in concentrated samples. On the other hand the occurrence of microcystin in Lake Victoria is likely as indicated by its general worldwide distribution (Chorus and Bartram 1999) and the isolation of a few strains with high microcystin contents during this study. Therefore although the worldwide distribution of microcystin producers has been confirmed, microcystin producers might not be of quantitative importance Uganda lakes.

Conclusion

Environmental parameters were of profound importance in the distribution of algal species within the four sampling sites. Cyanobacteria were the dominant algal class in Lake Victoria-Uganda except Lake Nabugabo which was dominated by Desmidiaceae. Nine different species were identified and *Microcystis* and *Anabaena* occurred at all sites. A few unialgal strains that were sampled in May from Lake Victoria and Lake Nabugabo had microcystin and were toxic to the *Thamnocephalus* larvae. Consequently the occurrence of microcystin producers in Lake Victoria-Uganda has been documented.

Much as Murchison Bay and Napoleon Gulf had toxic microcystin in their May net samples, the abundance and significance of microcystin producers in Lake Victoria-Uganda are however, considered to be low.

Apart from the toxic microcystin, there were high numbers of peptide compounds detected in both pure strains of cyanobacteria and in the field samples. The majority of strains as well as the majority of field samples was found to be highly toxic, but the toxicity level could not be related to either peptide like compound numbers or the total area of peptide compounds per unit of dry weight. The toxicity found in this study is rather related to the presence of single so far unidentified peptide like compounds than to the relatively few occurrences of microcystins.

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References

Chorus I. (2004) Toxic and bioactive peptides in cyanobacteria Available from URL: http://www.pepcy.de/index.htm. Accessed 2004, October, 02.

Chorus I. and Bartram J. (1999) *Toxic cyanobacteria in water. A guide to their public health consequences, monitoring and management.* WHO, E & FN Spon, London: 113-153.

DWD (2003) Directorate of Water Development, Uganda. Water resources management. Available from URL: http://www.dwd.co.ug/waterManagement-background.php Accessed 2004, April 18.

Fastner J., Flieger I. and Neumann U. (1998) Optimised extraction of microcystins from field samples- a comparison of different solvents and procedures. *Wat. Res.* 32: 3177-3181.

Haider S., Naithani V., Viswanathan P. N. and Kakkar P. (2003) Cyanobacterial toxins: a growing environmental concern. *Chemosphere* 52 (1): 1-21

Hecky R. E. (1993) The eutrophication of Lake Victoria. – *Verb. Internat. Verein. Limnol*. 25: 39-48.

Kling H. J., Mugidde R. and Hecky R. E. (2001) Recent changes in phytoplankton community of Lake Victoria in response to eutrophication. In: *The Great Lakes of the World (GLOW): Food web, health and integrity, Ecovision World Monograph series* (eds Munawar M. and Hecky R. E.) pp. 47-65. Backhuys publishers, Leiden, The Netherlands.

Krienitz L., Ballot A., Wiegand C., Kotut K., Codd G. A. and Pflugmacher S. (2002) Cyanotoxin-producing Bloom of *Anabaena flos-aquae*, *Anabaena discoidea* and Microcystis aeruginosa (Cyanobacteria) in Nyanza Gulf of Lake Victoria, Kenya. *Journal of Applied Botany* 76: 179-183

Kurmayer R., Christiansen G., Fastner G. and Börner T. (2004) Abundance of active and inactive microcystin genotypes in populations of the toxic cyanobacterium Planktothrix spp., *Environmental Microbiology* 6: 831- 841.

Kurmayer R. and Jüttner F. (1999) Strategies for the co-existence of zooplankton with the toxic cyanobacterium *Planktothrix rubescens* in Lake Zurich, *J. Plankt. Res*. 21: 659-683.

Lawton L. A. and Edwards C. (2001) Purification of microcystins, *Journal of Chromatography A*, 912: 191-209.

Lungaiya H. B. O., M´harzi A., Tackx M., Gichuki J. and Symoens J. (2000) Phytoplankton community structure and environment in the Kenya waters of Lake Victoria. *–Freshwater Biology*, 43: 529-543.

Mugidde R. (1993) Changes in phytoplankton primary productivity and biomass in Lake Victoria (Uganda). *Ver. Internat. Verein. Limnol*, 25: 846-849.

Ochumba P. B. O. and Kibaara D. I. (1989) Observations on blue-green algal blooms in the open waters of Lake Victoria, Kenya. *Afr. J.Ecol*. 27: 23-24.

Oudra B., Loudiki M., Sbiyyaa B., Martins R., Vasconcelos V., and Namikoshi N. (2001) Isolation, characterisation and quantification of microcystins (heptapeptides hepatotoxins) in *Microcystis aeruginosa* dominated bloom of Lalla Takerkoust lake-reservoir (Morocco), *Toxicon* 39: 1375-1381.

Oudra B., Loudiki M., Vasconcelos V., Sabour B., Sbiyyaa B., Oufdou Kh. and Mezrioui N. (2002) Detection and Quantification of Microcystins from Cyanobacteria Strains Isolated from Reservoirs and Ponds in Morocco, *Environ Toxicol* 17: 32-39.

Rippka R. (1988) Isolation and purification of cyanobacteria, *Meth. Enzymol*. 167: 3-27.

Sekadende B. C., Lyimo T. J. and Kurmayer R. (2005) Microcystin production by cyanobacteria in the Mwanza Gulf (Lake Victoria, Tanzania). *Hydrobiologia* 543: 299-304.

Sivonen K. and Jones G. (1999) Cyanobacterial toxins Chorus I and Bartram J (eds.) *Toxic cyanobacteria in water*. A guide to their public health consequences, monitoring and management. WHO, E & FN Spon, London 41-112.

Spoof L., Vesterkvist P., Lindholm T. and Meriluoto J. (2003) Screening for cyanobacterial hepatotoxins, microcystins and nodularin in environmental water samples by reversed-phase liquid chromatography-electrospray ionisation mass spectrometry, *Journal of chromatography A*, 1020 105-119.

Utermöhl (1958) Zur Vervollkommnung der quantitativen Phytoplankton-Methodik. Mitt. Int. Verein. Limnol. 9: 1-38.

Wetzel R. G. and Likens G. E. (2000) Limnological analysis. 3rd Edition. Springer Verlag. New York.

Levels of heavy metal pollution in water and sediments in Simiyu wetland of lake Victoria basin (Tanzania)

L. Henry[1] and P. Semili[2]

[1]Department of Laboratory Technology, Dar es Salaam Institute of Technology. P.O. Box 2958, Dar es Salaam, Tanzania
[2]Lake Victoria Environmental Management Project (LVEMP), P.O. Box 211, Mwanza, Tanzania
Email:pmwinyimvua@yahoo.co.uk

Abstract

Human activities play a major role in introduction of various anthropogenic pollutants into surface waters, resulting into elevated metal levels and lowered water quality. At present some metals such as iron (Fe), copper (Cu), zink (Zn), manganese (Mn), molybdenum (Mo) are believed to be essential for animal life. Furthermore, heavy metals such as mercury, lead, cadmium, and arsenic are extremely useful in industrial and agricultural technology. However, these heavy metals are highly toxic to human life, and have been found to be associated with a variety of diseases in man. For example, lead affects blood cells resulting to low haemoglobin levels and hence causes anaemia and also it inhibits enzymes. Cadmium disrupts the normal functioning of plant enzyme through inhibition, and in human beings it accumulates in ovaries, kidneys and liver hence result to kidney and liver damage. Methylmercury (MeHg) is a potent neurotoxic chemical, and it can biomagnify through food webs, and can reach concentrations 10 6 times higher in top predator fish than in either their ambi-ent environment or diet. In order to understand the levels of heavy metal pollution in River Simiyu wetland, concentrations of total zinc (Zn), cadmium (Cd), manganese (Mn), chromium (Cr), copper (Cu) and lead (Pb) were determined in both water and sediments in series of the sites of along the river and in Lake Victoria. The study was carried out in May and December 2003. Generally higher levels of the elements measured were found in sediments than in water samples. However, the range of concentrations of the six heavy metals recorded in the water column is of great concern to the human health. The results also show insignificant variation of the levels of heavy metals along the sites regardless of their spatial locations from the entire lake. Likewise, the comparison between wet and dry seasons indicates little variation except for manganese and zinc which were more abundant in the dry season; the two elements were well marked in the sediments.

Key words: Heavy metals, Lake Victoria, Sediments

Introduction

The continuous discharge of agricultural, industrial and domestic wastes into water bodies has largely contributed to ecological degradation of wetlands in the lake Victoria Basin. Such pollutants, particularly the heavy metals, can endanger the public health by being incorporated into the food chains. Heavy metals are not biologically degraded like most of the organic pollutants. Increased urbanization and industrialization in various parts of Lake Victoria Basin cause increased level of heavy/trace metals in the environment. In the environment, heavy metals tend to accumulate, particularly in sediments in association with organic and inorganic matter, and involves adsorption, complex formation and chemical combination. some of the trace metals are quite toxic to humans while some are essential for human health. However, the human body only requires the essential metals at a certain levels beyond which they may cause adverse effect. The trace metals mostly linked to human poisoning are lead, mercury, arsenic and cadmium (Linda et al 2003). Others are copper, zinc and chromium, which are actually essential to the body in small amounts only.

Heavy metals in the environment may be associated with air emission from coal burning, metal smelters, industrial facilities, waste incinerators, building paints, etc (Fostner, V. and Wittman, G.T.W. 1979). Heavy metals can also enter the environment through natural processes such as naturally occurring geological deposits (underlying rocks) dissolving into ground water and potentially resulting in unsafe levels of these heavy metals in water. Cycling of these metals is based on the sharing of the metals among organisms, air, water and plants and back to organisms again.

Traces of heavy metals may exist in fresh water bodies as simple hydrated ions and inorganic and organic complexes of low molecular weight. They may also be sorbed onto organic and inorganic particulate species. Protection and management of aquatic system requires knowledge of the distribution and fate of the toxic metals in both water column and sediments. The physico-chemical status of the aquatic ecosystems has been widely studied word wide because of its importance in the prediction of transport, bioavailability and the fate of the metals.

Knowledge about the chemical speciation of heavy metals in natural water is important in integrating their biological and geochemical cycling in aquatic systems. Due to their toxicity and their potential bioaccumulation in biological systems, heavy/trace metals constitute to health risk. Toxicity depends primarily on the physicochemical state of the metal, not the total concentration of free (hydrated) metal ions which is regarded as the most toxic form. Apparently, there are very few studies that have been conducted to demonstrate the impact of agriculture and non-point pollution on the Lake Victoria wetland.

The aim of this study was to investigate levels and distribution of heavy/trace metals along the Simiyu wetland in Mwanza Region (Tanzania). Such information is essential to understanding of the status of heavy/trace metal pollution and the potential human risk of the water and fish used for human. Trace metals captured in the compartments (water and sediments) can be used as a tracer for air mass transportation and revealing the history and intensity of regional pollution. The study may try to find out some sources of the metals so that adequate policy issues can be put in place.

Methods

The study was conducted along River Simiyu in Mwanza Region (Tanzania) in May and December 2003. The wetland lies along the agricultural area and residential houses. There are no improved sewerages and some burial sites can be seen. People along Simiyu are mainly farmers, and few fishermen and cattle keepers. Lack of proper sewerage systems, burying and burning of organic matter are common phenomena in the wetland. The above facts together with the presence of underlying rock may contribute highly to the levels of heavy/trace metals in the wetlands. Eight sampling sites were located by GPS along the River Simiyu, from up-stream to down stream about 400 metres into the lake. The sites include Maligisu, Ididi, Bugandando, Lumeji, Bridge, River mouth, River mouth (+100m and 300m). The 100 and 300 metres were measured into the lake from the river mouth.

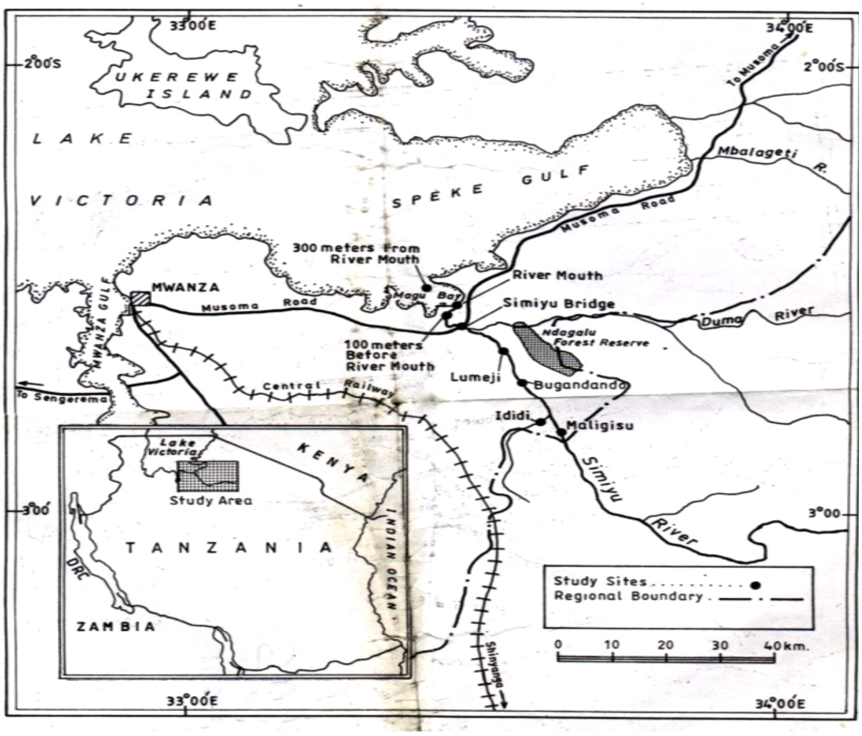

Figure 1. Map showing River Simiyu and sampling stations.

Water samples were collected from the eight stations, 5cm below the surface. Nitric acid was added into the samples to reduce the pH to 2.0. Three replicates were dissolved in a mixture of aqua regia (3;1HNO_3;HCl). The samples were analysed in Atomic Absorption Spectrophotometer (AAS) by direct aspiration into acetylene-air flame. The analyses were done in a Perkin Elmer Model 2380 AAS.

The river bottom sediments were collected using Heckman grab sampler. To avoid contamination the inner part of the fresh sediments was scooped and transferred into a pre-cleaned polythene bags and frozen. Composite samples were taken from 4-5 samples at each sampling station. In the laboratory the sediments were dried at 105^0C, crushed to a powdered from and passed through a 63um aperture sieve (stainless steel mesh material, 410 BS). A portion of 2 gram powdered sample was digested using 10ml aqua regia (1:3HNO_3;HCl) for 30mn at 60^0C using an aluminium heating block. The sample solutions were diluted to 25mls in volumetric flasks and aspirated into the 2380 model AAS.

Results and discussion

Trace metal distribution among sites

The concentration levels and distribution of trace metals in both water and sediments are presented in Tables 1–4. Relatively higher levels of the elements were found in sediments during both dry and wet seasons (Table 1 and 2). Unlike the organic pollutants, trace metals are not usually eliminated from the aquatic system by natural processes like bacterial action. Toxic metals such as Cadmium (Cd) and Copper (Cu) tend to accumulate in the bottom sediments where they may be released by various processes of mobilization (Foster and Wittman, 1979) thereby increasing their bioavailability. The study showed relatively higher levels of Magnesium (Mn) and Zinc (Zn) than the other studied elements. However, no significant variation of the levels in regard to the respective distances of each sampling site from the lake. This may suggest higher contribution of the underlying rocks (Nyazanian-Kavirondo rock) to the availability of the detected elements on the area. Presence of the metals in the study area can partly be attributed to the dissolution of rocks, soil and minerals into the water through natural processes (Bolt and Bruggnwert, 1972; Oteko, 1989).

Table 1. Concentrations of trace metals in water and sediment samples in dry season (in ppm or mg/L).

Analyte	Matrix	Sampling station								
		1	2	3	4	5	6	7	8	Mean
Cd	*Sediment*	1.04	4.00	4.04	5.08	4.87	3.79	3.17	0.70	3.33
	Water	0.02	0.04	0.05	0.07	0.07	0.09	0.10	0.09	0.07
Cu	*Sediment*	1.08	12.09	1.48	3.01	7.12	11.38	6.79	13.43	7.05
	Water	0.07	0.07	0.05	0.05	0.07	0.05	0.05	0.07	0.06
Pb	*Sediment*	0.21	<0.01	<0.01	<0.01	<0.01	1.94	<0.01	2.28	<0.01
	Water	0.01	0.02	0.02	0.02	0.05	0.04	0.05	0.04	0.03
Cr	*Sediment*	2.66	4.53	5.30	1.32	<0.01	14.69	5.61	20.87	7.85
	Water	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Zn	*Sediment*	194.70	184.45	12.94	14.22	1.563	80.83	130.00	137.06	94.47
	Water	2.91	0.19	0.07	0.20	0.15	0.06	0.05	0.02	0.46
Mn	*Sediment*	99.13	146.34	64.16	74.24	71.78	497.67	375.58	758.07	260.87
	Water	0.19	0.16	0.32	0.34	0.33	0.27	0.26	0.18	0.26

Table 2. Concentrations of trace metals in water and sediment samples in wet season (in ppm or mg/L).

Analyte	Matrix	Sampling station								
		1	2	3	4	5	6	7	8	Mean
Cd	*Sediment*	1.01	0.07	0.05	0.06	0.22	0.22	0.44	0.50	0.20
	Water	0.01	0.01	0.02	0.02	0.02	0.02	0.02	0.02	0.02
Cu	*Sediment*	6.80	6.50	4.00	5.70	14.40	14.50	66.40	88.70	25.87
	Water	0.02	0.01	0.02	0.02	0.01	0.01	0.02	0.06	0.02
Pb	*Sediment*	5.70	5.70	3.40	2.40	9.90	6.90	23.60	22.40	10.00
	Water	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Cr	*Sediment*	n.a	n.a	n.a	n.a	n.a	n.a	n.a	n.a	n.a
	Water	n.a	n.a	n.a	n.a	n.a	n.a	n.a	n.a	n.a
Zn	*Sediment*	14.60	11.30	8.80	9.30	43.40	27.90	73.20	87.00	34.44
	Water	0.03	0.02	0.04	0.03	0.05	0.04	0.03	0.04	0.04
Mn	*Sediment*	242.45	137.27	129.62	105.87	312.81	246.50	569.87	735.94	310.04
	Water	1.50	0.31	1.21	1.39	1.90	1.98	1.72	1.48	1.44

Previous study in Mwanza south and north recorded also the prevalence of Mn and Zn in some study areas and associated to industrial complexes and railway oil deposits (Kondoro and Mikidadi, 1998). The upper part of Simiyu wetland is found near a railway line in which the possibility of such contamination is also likely. Presence of higher concentrations of Zn recorded in the previous studies are linked to ground wood pulp production, metal plating and steel works ((Kondoro and Mikidadi, 1998). The argument can still hold for the current study since a number of garages involving steel works are common in the nearby areas around Simiyu. This has resulted to a number of scrap metal deposits near the wetland which by different processes can easily reach the river water/sediments.

Seasonal variation trace elements in water and sediments

Simiyu wetland experiences an abundant water flow through out the year hence maintaining equilibrium between the rocks underneath and the water column through the processes such as dissolution, precipitation and absorption. However, less availability of some elements (Cd, Cu, Pb and Cr) in water samples, especially during wet season may suggest higher dilution due to the added water volume or that some of the elements are easily been carried out by the water. On the other hand, the higher availability of some elements may also depend on its complexes with other compounds in the crust. Some may form heavier complexes which can easily settle down to the bottom rather than been taken away by the moving water. Mn and Zn were detected at higher concentrations in both seasons. However, in dry seasons recorded higher levels, then in wet (Tables 3 and 4). The reasons are not yet known by authors.

Table 3: Seasonal variations of trace metal levels in water (in ppm or mg/L).

Analyte	Season	Sampling station								
		1	2	3	4	5	6	7	8	Mean
Cd	*Dry*	0.02	0.04	0.05	0.07	0.07	0.09	0.10	0.09	0.07
	Wet	0.01	0.01	0.02	0.02	0.02	0.02	0.02	0.02	0.02
Cu	*Dry*	0.07	0.07	0.05	0.05	0.07	0.05	0.05	0.07	0.06
	Wet	0.02	0.01	0.02	0.02	0.01	0.01	0.02	0.06	0.02
Pb	*Dry*	0.01	0.02	0.02	0.02	0.05	0.04	0.05	0.04	0.03
	Wet	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Cr	*Dry*	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Wet	n.a	n.a	n.a	n.a	n.a	n.a	n.a	n.a	n.a
Zn	*Dry*	2.91	0.19	0.07	0.20	0.15	0.06	0.05	0.02	0.46
	Wet	0.03	0.02	0.04	0.03	0.05	0.04	0.03	0.04	0.04
Mn	*Dry*	0.19	0.16	0.32	0.34	0.33	0.27	0.26	0.18	0.25
	Wet	242.45	137.27	129.62	105.87	312.81	246.50	569.87	735.94	310.04

Table 4. seasonal variations of trace metal levels in sediments (in ppm or mg/ L).

Analyte	Season	Sampling station								
		1	2	3	4	5	6	7	8	Mean
Cd	*Dry*	1.04	4.00	4.04	5.08	4.87	3.79	3.17	0.70	3.34
	Wet	0.01	0.07	0.05	0.06	0.22	0.22	0.44	0.50	0.20
Cu	*Dry*	1.08	12.09	1.48	3.01	7.12	11.38	6.76	13.43	7.05
	Wet	6.80	6.50	4.00	5.70	14.40	14.50	66.40	88.70	25.87
Pb	*Dry*	0.21	<0.01	<0.01	<0.01	<0.01	1.94	<0.01	2.28	<0.01
	Wet	5.70	5.70	3.40	2.40	9.90	6.90	23.60	22.40	10.00
Cr	*Dry*	2.66	4.53	5.30	1.32	<0.01	14.69	5.61	20.87	7.85
	Wet	n.a	n.a	n.a	n.a	n.a	n.a	n.a	n.a	n.a
Zn	*Dry*	194.70	184.45	12.94	14.22	1.563	80.83	130.00	137.06	94.47
	Wet	14.60	11.30	8.80	9.30	43.40	27.90	73.20	87.00	34.43
Mn	*Dry*	99.13	146.34	64.16	74.24	71.78	497.67	375.58	758.07	260.87
	Wet	1.50	0.31	1.21	1.39	1.90	1.98	1.72	1.48	1.44

Water quality

Heavy/trace elements are important chemical pollutants in natural waters. The heavy metal pollution of aquatic ecosystems is often most obviously reflected in high metal levels in sediments, microphysics and benthic animals, than in elevated concentrations in water (Linnik and Zubenko, 2000). On the other hand, small quantities of the trace elements are permitted by in some standards water requirements as shown in the Table 5 below.

Table 5. permissible elemental concentrations in drinking water in mg /L.

Element	USPHS- 1962	WHO - 1963	WHO-1993	EU -1961
Cr	0.05	0.05	0.03	0.05
Mn	0.05	0.10	-	0.10
Cr	1.00	1.50	0.05	3.00
Zn	5.00	15.0	-	5.00
Pb	0.05	0.05	0.01	0.10

Source: United States Public Health services (USPHS), World Health Organization (WHO) and European union (UN) as reported by Holden (1970) and Anon (1993).

The levels of the elements detected in the Simiyu wetland are far above those recommended by USPHS, WHO and EU (Table 5), thus the water is not safe for human consumption. Although a good number of the residents resort to well waters for drinking there is high possibility of having even higher levels of the elements in well especially those residing from the parent rock. Simiyu water is used for irrigation in small horticultural plots, drinking for cattle, washing clothes and domestic utensils, bathing and to a lesser extent for human consumption (drinking and cooking).

Conclusion

The study has shown that the levels of the elements detected in the Simiyu wetland are above the permissible levels in drinking waters (USPHS, WHO and EU) as shown in Table 5. Therefore, there is an urgent need to protect the Simiyu River and its wetland from further pollution. A study investigating levels of heavy metal pollution in wells used as source of domestic water is highly recommended. The levels of turbidity and faecal coliforms (FC) in most samples exceeded the Criteria of Surface Water for Public Water Supplies found in the literature (Hammer, 1975; WHO, 1984; WRC, 1998). Electrical conductivity and chloride concentrations were within the Criteria of Surface Water for Public Water Supplies. The study and published information also revealed that agriculture and urban runoff, and municipal raw sewage, domestic and industrial wastes are the major sources of pollution in Mwanza gulf. It was further noted that agricultural activities and destruction of wetlands aggravated the situation. Other human activities like fish processing, washing cargo handling, and local boat making which carried out either at the lake shores or near the lake are also significantly cause pollution in the gulf. This implies that without intervention measures the cost of water treatment would increase. It is also implies that the health of people using water from Lake Victoria prior treatment would be threatened.

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References

Bolt, G. H. and Bruggenwert, M.G.M (1972) Soil Basic Elements, Elsevier Science Publishers, Amsterdam.

Fostner, V. and Wittman, G.T.W. 1979 Metal Pollution in the Aquatic Environment, Berlin Hedelburg, New York.

Holden W.S. (1970) Water Treatment and Examination, J & A Churchill Publishers; pp.326.

Kondoro, J.W.A and Mikidadi, M.H. Heavy metal pollution in Lake Victoria, Tanz. J. Sci. 1998, 24; 9 – 21.

Linda M. Campbell 1, * , Robert E. Hecky 1 , Joseph Nyaundi 2 , Rose Muggide 3 , and D. George Dixon (2003) . Distribution and Food-web Transfer of Mercury in

Linnik, P.M. and Zubenko, I.B. Role of bottom sediments in the secondary pollution of aquatic environment by heavy metal compounds: Lakes & Reservoirs, Research and Management, 2000, 5; 11-21.

Napoleon and Winam Gulfs, Lake Victoria, East Africa. *J. Great Lakes Res. 29 (Supplement 2):267–282. Internat. Assoc. Great Lakes Res*.

Oteko, D. (1989) Trace metal pollution along the Kenyan Coast in Khan M.R. & Gijzen (eds), Proceeding of a Symposium on Environmental Pollution and its management in East Africa, University of Dar es Salaam.

Water quality status in the Kenyan waters of Lake Victoria

Sitoki L., Albert Getabu, W. Osumo

Kenya Marine and Fisheries Research Institute, P. O. Box 1881, Kisumu Kenya
Email:sitoki@hotmail.com

Abstract

This paper reports on the prevailing physical and chemical conditions as well as phytoplankton species composition in Lake Victoria, Kenya. Three categories namely; (1) stations with poor water quality; (2) stations with water of intermediate quality and (3) stations with high water quality were distinguished. The stations differed mainly in conductivity, soluble reactive silica, total hardness and total alkalinity. Nutrient values ranged between 45.8 ± 9.1 μg PO_4-P l^{-1}, 1.0 ± 0.6 μg NO_3-N l^{-1} and 9.8 ± 1.4 mg SiO_2-Si l^{-1}. Regarding the phytoplankton species composition and numerical abundance, a total of 52 algal genera were identified. Cyanophytes were the most dominant contributing over 50% of the total biomass, Bacillariophyceae (25%) Chlorophyceae (21%) were equality important, while other algal groups were largely insignificant. Classification of water quality of this sort will hopefully assist decision makers in defining restoration / conservation strategies, and will provide an important tool for assessing water quality status of L. Victoria.

Key words: water quality, Algae, nutrients, conservation

Introduction

Lake Victoria is the largest freshwater body in Africa which is rich in natural resources such as fisheries and others. Population around it is over 30 million with 3m depending directly on fisheries. Its also a foreign exchange earner to the riparian countries. Despite this importance, water quality issues remain a major concern as the riparian population continues to grow leading to degradation of the ecosystem and water quality. A lethal cocktail of toxic waste and nutrients derived from industry, agriculture and new housing is causing serious damage to the lake ecosystem. Particularly serious is the process of eutrophication, which is a form of water pollution, leading to concerns of the status of water quality in the Lake.

In the past decade, eutrophication of the Lake Victoria has resulted in increased phytoplankton biomass (by a factor of 7), increased primary production (by a factor of 2), a dramatic decrease in dissolved silicon concentrations (by a factor of 10) and a significant decrease in ratio of N and P available to the phytoplankton since 1960 when nitrogen fixing bacteria were a minor element in the in the lakes phytoplankton (Kling et al., 2001). There have been profound changes in the fish community in the last three decades that may be associated with deteriorating water quality. However, the more recent rapid reduction, especially Haplochromines coincided with a cascade of changes that begun with the sudden increase in stocks of the increased predator, Nile perch (*Lates niloticus*). Nile perch (*Lates niloticus*) and Nile Tilapia (*Oreochromis niloticus*) introduced in the late 1950s and early 1960s underwent dramatic population increases in the late 1970s and early 1980s having a detrimental effect on the diverse, endemic populations of the Haplochromines (Ogutu-Ohwayo, 1990; Goldschmidt and Witte, 1992; Kaufman, 1989; Kaufman, 1992). The effects of hypoxia and intense predation probably interacted with competition between native and introduced cichlids, further aggravating the decline of some native species. This study examined the prevailing environmental factors and the distribution of phytoplankton as well as estimating primary production levels with the overall aim of assessing the water quality status of Lake Victoria, Kenya.

Material and methods

Samples for water and phytoplankton analysis were taken from 1998 to 2005 in the main Lake (Figure 1). Light penetration was estimated with a 20-cm diameter, black and white Secchi disk. Turbidity was measured with a Hach Turbidimeter 2100P. *Insitu* measurements were made for temperature, dissolved oxygen, conductivity, pH automatically using a Hydrolab Surveyor II connected to an SVR 2- DU display and a 401 - CA circulator assembly.Water samples were collected with a 3-litre Van Dorn sampler. Portion of the water samples (50 mls) were analyzed for total alkalinity and total hardness by titration with 0.02 N HCl to final pH of 4.5 using methyl indicator and with 0.02 N EDTA respectively as outlined in GEMS (1992). Spectrophotometric methods were employed to determine the soluble reactive phosphorous (PO_4-P) and nitrate nitrogen (NO_3-N), Total P and Total N as outlined by Mackereth et al., (1978) and soluble reactive silica (SiO_2-Si) according to APHA (1985).

For phytoplankton analysis, 250 ml of the water was placed in polyethylene bottle and fixed immediately with Lugol's iodine solution. After 48 hours decantation, the lower layer (20-25 ml) containing the sedimented algae was put in a glass vial and stored in a dark cool box. The known volume of the concentrated sample was used for the identification and counting of the phytoplankton under an inverted microscope. Phytoplankton species were identified using the methods of Huber –Pestalozzi (1968) as well as some publications on East African lakes (Cocquyt et al., 1993). Phytoplankton densities (individuals ml^{-1}) were estimated by counting all the individuals whether these organisms were single cells, colonies or filaments. Primary production levels were estimated by collecting water samples from predetermined depths using a Van-don sampler. A rubber tubing was used to conduct the water into BOD bottles for incubation from the same depth the water was collected. In most stations, 4 depths were sampled: surface, 1, 2 and 3 metres. The standard light and dark BOD bottles were used for the experiment. Incubation was done for 2 hours. Titrimetric methods were used to determine the oxygen levels as outlined in standard methods (APHA 1985). Sampled stations were grouped into five different ecological zones for comparison purposes (Table 1).

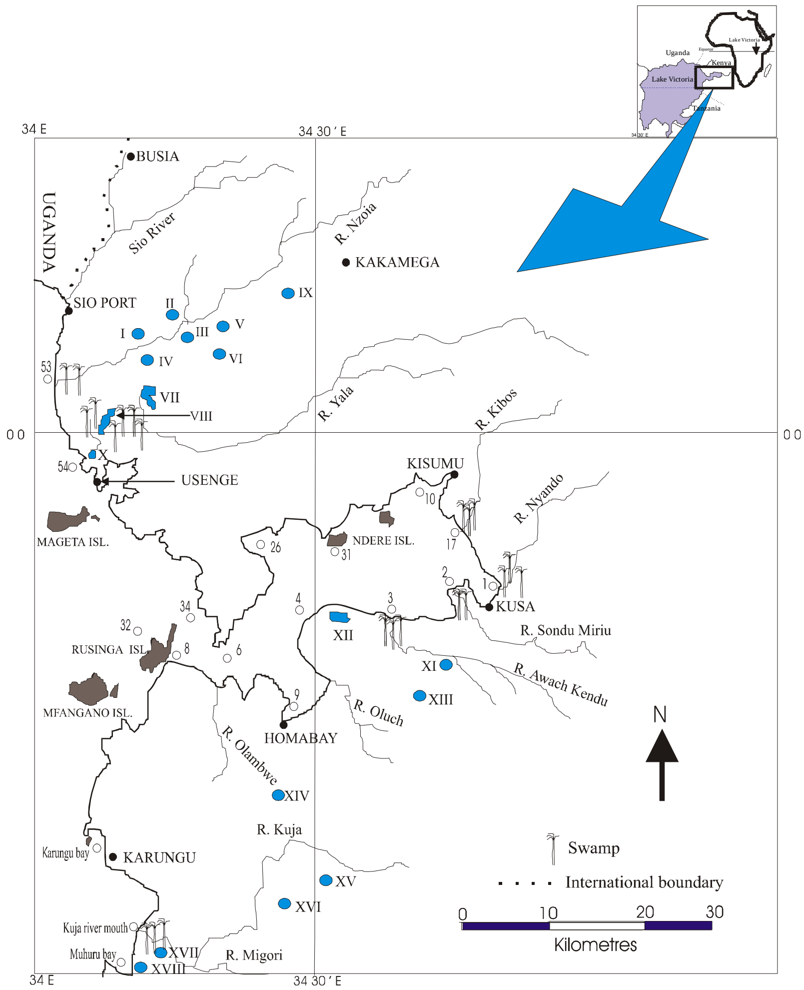

Figure 1. Map of Lake Victoria, Kenya sector, showing sampling stations.

Table 1. Stratification of the sampling stations of Lake Victoria.

Zone	Condition	Stations
I	River influence	Kuja, Nzoia, Sio, Oluch
II	Urban influence	Homa Bay, Dunga, Sori
III	Mid-Gulf	Gingra, Gudwa, Ochieng-Oneko, Maboko, Ndere, Kopiata, Mirunda
IV	Offshore	Utajo, Bridge Island, Naya
V	Less disturbed bay	Asembo, Luanda-Gembe, Miti Mbili

Results

A total of 23 sampling stations in the main Lake were sampled at least twice a year. Several water quality parameters were subjected to various analytical tools and using hierarchical clustering the stations were grouped into 4 clusters as shown in Figure 2. The clusters were clearly separated from each other on the basis of offshore with water of good (Matara, Kuja, Muhuru & Sori), intermediate with water of moderate quality (Mbita, St 6 and Maboko, inshore with poor water rest of the stations. The fourth cluster of station 1 & 2 are of extremely poor quality (polluted heavily) are river mouths whose source drain rich agricultural lands as well as industrial and urban areas.

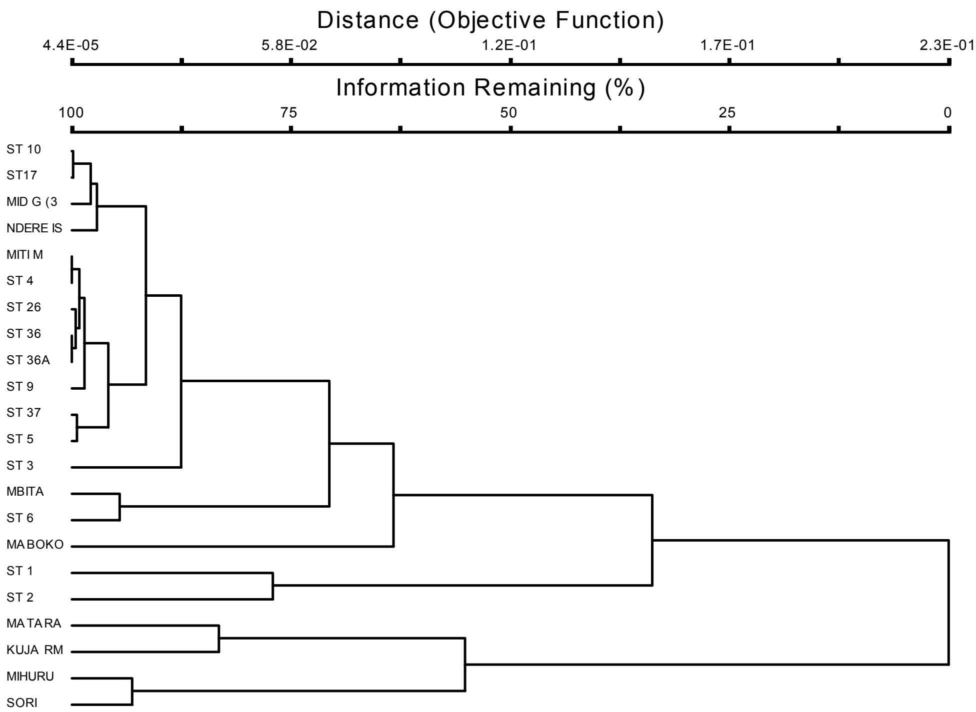

Figure 2: Four main clusters in a dendrogram form for Lake Victoria based on physical and chemical parameters. The 1st cluster Matara, Kuja, Muhuru and Sori, Cluster 2 stations 1 and 2, Cluster 3 has stations Mbita, St 6, and Maboko while the rest represent inshore stations.

Table 2: Mean ± SE of surface physico-chemical parameters and nutrients in L. Victoria, 2000- 2001, September 2003 and July 2004.

Parameter	**Nov 2000**	**n**	**Sept 2003**	**n**	**July 2004**	**n**
Temperature (°C)	26.8 (0.2)	23	25.98 (0.16)	30	25.4 (0.12)	20
DO (mgL^{-1})	7.3 (0.2)	23	8.18 (0.35)	30	7.3 (0.12)	20
Chloro-*a* (μgL^{-1})		23	15.8 (4.4)	30	31.6 (12.6)	17
Phosphate (μgL^{-1})	45 (9.1)	22	34.9 (24)	30	14.9 (5.2)	17
Nitrate (μgL^{-1})	1 (0.6)	22	14.9 (2.7)	30	7.9 (1.0)	17
Turbidity (NTU)			28.17 (3.47)	28	23 (5.07)	7
Transparency (M)	0.9 (0.1)	22			0.6 (0.07)	9

There was little variation in temperature between and within stations. This suggests that there was no thermal stratification in the sampled sites. Dissolved oxygen concentrations were relatively high in surface waters (7.3 ± 0.12), and in bottom waters (5.28 ± 0.33), rarely going below 3 mgL^{-1}. Mean variation in some physical and chemical parameters and nutrients from the Lake are displayed in Table 2. Transparency increases inshore to offshore with very thin productive water column (Fig 4). The ratio of total nitrogen to total phosphorus is generally low both inshore and offshore (Fig 2). Algal biomass was 23 μgL^{-1}inshore and 15 μgL^{-1} offshore with cell densities of over 2000 cells mL^{-1} (Fig 5). Cyanophytes dominate algal community of Lake Victoria.

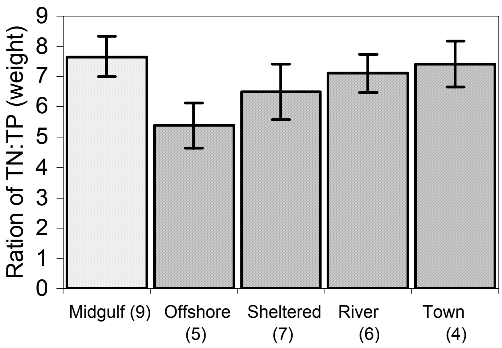

Figure 3. TN:TP ratios in various ecological zones in of L Victoria.

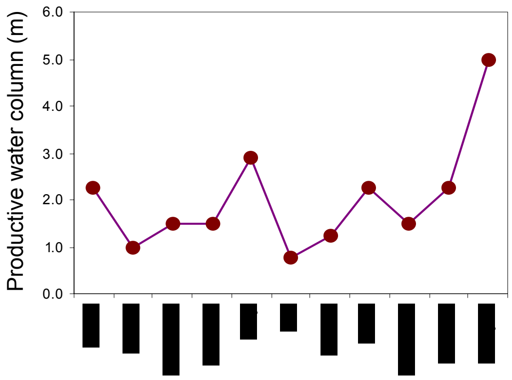

Figure 4. Estimated euphotic depth (m) in various zones in the of L Victoria.

There were 81 species of algae identified belonging to 52 genera. The species identified were grouped into a higher taxon (family) and then clustered. It was clear that Cyanophytes were the most dominant taxa in most of the stations. In areas with heavy algal blooms like river mouths, there were correspondingly high productivity values particularly in Rivers Oluch and Nzoia. Lake Victoria's average gross primary production stood 800 mg C M^{-3} h^{-1} while the net production was 533 mg C M^{-3} h^{-1} (Fig 6). Variations between gross and net primary production were significant however variations within the different ecological zones were minimal with exception of river zones.

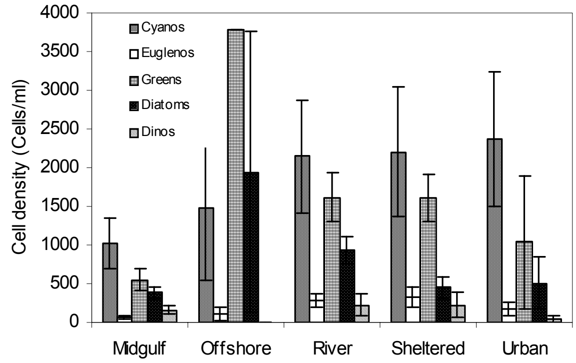

Figure 5. Algal density (cells/ml) in various ecological zones in L Victoria.

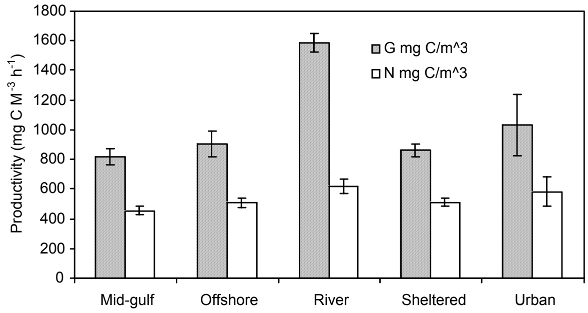

Figure 6. Primary production rates in Lake Victoria, March, April, June and August 2005 in various zones of Lake Victoria in five different ecological zones.

Discussion

Over the years since the first limnological analysis of the main Lake water by Worthington (1930), the lake has continued to receive pollutants that have changed its quality (Lungaiya et al. 2001). The present survey findings suggest that the environmental problems around lake Victoria have continued to increase both in intensity and diversity. These changes in the Lake's ecosystem can directly be attributed to anthropogenic activities, management problems and to a lesser extent natural causes. Natural such as Lake level fluctuations due to drought and Elnino phenomenon. In particular, inundation of farmlands that occur during such periods as Elnino may contribute large loads of sediments and nutrients to the lake whereas decrease in water levels can modify nearshore habitats and change breeding area available to many fish species (Lungaiya et al. 2001).

Secchi depth readings from the lake are comparable to those from Lung'aiya, (1998), Melack (1979) and Ochumba and Kibaara (1989) but lower than those recorded by Worthington (1930). This decrease in transparency (and increase in turbidity) may be attributed to increased erosion from the catchment, turbulence and resuspension of sediments, as well as river influence and increase in algal biomass. Values for Conductivity in the main Lake compared well with those in Hecky and Bugenyi, (1992) and Lung'aiya, (1998). These increasing values may suggest increasing levels of inputs of ions (in form of fertilisers) and dissolved solids mainly from terrestrial sources.

The analysis of phytoplankton show similarities with early works reported in literature (Talling, 1965, 1987; Lung'aiya et al., 2000). These include *Niztschia acicularis, Pediastrum duplex, P. simplex, ceratium sp., Aphanocapsa sp. Microcystis flos-aquae, Anabaena flos-aquae, Anabaenopsis tanganyikae* and *Rhizosolenia sp.* However, some differences do exist in the agal cell numbers and chlorophyll-*a* concentrations, which appear to be increasing. Average algal numbers of 2,000 per millilitre is much higher than reported by Talling (1965) and Lung'aiya et al., (2000). Lake Victoria's phytoplankton community is dominated by Cyanobacteria which is known to be as a result of eutrophication, where noxious algae grow fast to predominate the other groups. High nutrient enrichments especially in the inshore areas has been reported by Lung'aiya *et al.* (2001) and Ochumba (1989) as the principal cause of the persistent algal blooms that have become more frequent. The net result being depletion of oxygen in the lower underlying waters, thereby reducing habitats for living organisms especially fish. Heavy algal blooms cause oxygen supersaturation in upper water layers in addition to high pH fluctuations due to photosynthesis and respiration, such huge fluctuation often are detrimental to fish.

Primary production experiments show that most of the ecological zones on average have about the same levels of productivity. In shallow areas especially river mouths, algal densities are highest with equally high productivities. High phosphate loads has resulted in a significant reduction in TN:TP ratio thus favouring the blue-greens. Blue-green are known to produce toxins that may pose a heath risk. In addition, high algal densities cause light attenuation thus, algae becomes self - limiting resulting in a reduction of the productive water column. Increase of water transparency from the littoral zones to the pelagic waters results in the increment of the depth of productive water column consequently increasing fish habitat.

Primary production increases in the same fashion due to a decrease in light attenuation thus, the amount of the water column that is productive increases from Kisumu bay – Bridge transect. In offshore areas, primary production is somewhat lower but is compensated by a deep productive water column. Altogether, the factors investigated in this study depict that the mid-gulf and offshore stations have a relatively good quality water hence support high diversity of the algal flora. This is important for sight feeding fish besides the benefits of increased niche for fisheries production and general good water quality. However, inshore areas show a general reduction in water quality, which is a big cause for concern.

Conclusions

- Declining water quality may be responsible for trophic changes, loss of habitats and biodiversity, steps to reduce nutrient loads especially P should be undertaken

- Reductions in nutrient loads and concentrations are crucial to maintaining good water quality.

References

American Public Health Association (APHA), 1985. Standard Methods for The Examination of Water and Wastewater, 16th Edition Port City Press Baltimore MD.

Bootsma, H. A., Hecky, R. E., 1993. Conservation of African Great Lakes: A limnological perspective. Conserv. Biolo. 7 (3) 644-656.

Cocquyt, C., Vyverman, W., and Compere, P., 1993. A checklist of the algal Flora of the East African Great Lakes (Malawi, Tanganyika and Victoria). National botanical Garden of Belgium, Meise.

Global Environmental monitoring system Gems, 1992. WATER operational guide 3rd Edition

Goldshimdt, T., Witte, F., 1992. Explosive speciation and adaptive radiation of the Haplochromine cichlids from lake Victoria: an illustration of the scientific value of a lost speies flock. Mitt. Internat. Verein. Limnol. 23:101-107.

Hecky, R.E., Bugenyi, F.W.B., 1992. Hydrobiology and chemistry of the African Great Lakes and water quality issues: problems and solutions. *Verh. Int. Verein. Limnol.* 23: 45-54.

Huber-Pestalozzi, G., 1968. Cryptophyceae, Chloromonadophyseae and Dinophyseae. Das phytoplankton des Susswars, 3 Teil (ed.G. Huber –Pestalozzi), 2.Aufl.,pp.1-IX + 1-322.Schweizerbartsche-Verlagsbuch-handlung, Stuttgart.

Kaufman, L. S., 1989. Challenges to fish faunal conservation programmes as illustrated by captive biology of the Lake Victoria cichlids spp. 105-120. Fifth world Congress on breeding endangered species in captivity.

Kaufman, L.S., 1992. Catatstrophic change in species rich freshwater ecosystems:the lessons of Lake Victoria. Bioscience (II) 42: 846-858.

Kling, H. J., Mugidde, R., Hecky, R. E., 2001. Recent changes in the phytoplankton community of Lake Victoria in response to eutrophication. The Great Lakes of the World (GLOW): foodweb, health and integrity, pp 47-65. Edited by M. Munawar and RE Hecky. Ecovision World Monograph Series. 2001 BackhuysPublishers, Leiden The Netherlands.

Lung'aiya H.B.O. 1998. Relevance of limnological studies to biodiversity conservation and fisheries management in Lake Victoria. Paper presented at the National Workshop of the Lake Victoria Environmental Management Project (LVEMP) on biodiversity and social economics, 28th-31st July 1998, Rongo, Kenya.

Lung'aiya, H.B.O., M'Harzi, A., Tackx, M., Gichuki, J., Symoens, J.J., 2000. Phytoplankton community structure and environment in the Kenyan waters of Lake Victoria. *Freshwat. Biol*. 43: 529-543.

Lung'aiya, H.B.O., Sitoki, L., Kenyanya, M., 2001. The nutrient enrichment of LakeVictoria (Kenyan waters) Hydrobiologia 458: 75-82.

Mackereth, F. J. H., Heron, J., Talling, J.F., 1978. Water analysis. Some revised methods for limnologists. Freshwater Biological Association 1978. Scientific Publication No. 36.

Melack, J.M., 1979. Photosynthetic rates in four African fresh waters. *Freshwat. Biol*. 9: 555-580.

Ochumba, P.B.O., Kibaara, D., 1989. Observations on the blue-green algal blooms in the open waters of Lake Victoria, Kenya. *Afri. J. Ecol*. 27: 23-34.

Ogutu Ohwayo, R., 1990. The decline of the native fish species of Lake Victoria and Kyoga (East aFrica) and the impact of the introduced species especially the Nile perch, *Lates niloticus* and the Nile tilapia *Oreochromis niloticus*. *Environ. Biolo. Fish* 27: 81-96

Talling, J.F., 1965. The photosynthentic activity of phytoplankton in East African lakes. *Int. Rev. ges. Hydrobiol*. 501: 1-32.

Talling, J.F., 1987. The phytoplankton of Lake Victoria (East Africa). *Ergebn. Limnol*. 25: 229-256.

Worthington, E.B., 1930. Observations on the temperature, hydrogen-ion concemtration and other physical conditions of the Lakes Victoria and Albert Nyanza. *Int. Rev. ges. Hydrobiol. Hydrogr*. 24: 328-357.

Role of wetlands in maintaining stability of tropical lakes: A case study of River Nyando delta in Lake Victoria basin, Kenya

Taita Terer[1], Nathan N. Gichuki[2] & George G. Ndiritu[1&3]

[1]National Museums of Kenya, Center for Biodiversity (Wetlands & Marine Resources Programme), P.O. Box 40658-00100, Nairobi, Kenya
[2]University of Nairobi, Department of Zoology, P.O. Box 30197, Nairobi, Kenya.
[3]University of Arkansas, USA
Email: taaita@yahoo.com /cbd@museums.or.ke

Abstract

Freshwater lakes in the tropics tend to have extensive areas of marginal wetlands, which occur in the form of bays, estuaries, delta, sandy or muddy beaches and dense marginal vegetation. The lacustrine wetlands associated with Lake Victoria basin play an important role in purifying water and stabilizing its supply to the lake. They also constitute a highly productive littoral zone with numerous ecological niches. The recycling of nutrients and regeneration of biological resources also takes place in these marginal habitats. However, the ecological significance of these wetlands in improving water quality and maintaining the stability of the lake ecosystem and fisheries resources is currently not well understood and hence less appreciated by natural resource users and managers. For that reason, marginal wetlands are drained for agriculture, misused as dumping areas for waste materials and the resources therein overexploited. This paper describes the pollution buffering capacity of a tropical wetland (Nyando delta) and the ecological implications of this function in maintaining biological diversity and fisheries in the tropical Lake Victoria in Kenya. The sampling of physical-chemical parameters was carried out a long the Nyando river in four main sections: the upstream, the middle section, section just before entering the swamp and in Nyakach bay immediately after the swamp. Multivariate methods of data analysis were used to explore water physical-chemical characteristics. Comparisons of water characteristics were made before and after passing through the Nyando swamp. The results indicated significant levels of pollution in the affluent water especially its high nutrients load (nitrates and phosphates). However significant differences were found in water entering and leaving the swamp. The buffering capacity of this wetland was relatively high but was likely to change due to human interventions especially reclamation of its parts for cultivation and pasture.

Introduction

A number of studies have shown that presence of wetlands in the lower parts of rivers retain and reduce the amount of pollutants entering drainage basins (Pan & Stevenson 1996, Maher *et al.,* 1999). Moreover wetlands especially those occurring at the shoreline play a vital role as breeding ground for fish (Balirwa, 1995) hence help to sustain the fisheries in Lake Victoria in addition to supporting other biodiversity. Globally, wetlands are known for their biodiversity (Hammer & Bastian 1989). Functionally, wetlands have a sponge effect and are known for their filtration capacity (Kansiime, 2004). The buffering capacities of vast and diverse types of wetlands around Lake Victoria have not been fully documented (M'mayi, 1997). However, it is known that human activities in the catchments hundred of miles away continue to directly influence the lake ecology . This is made possible by the unidirectional and linear form of lotic systems that insures an intimate relationship between rivers and their drainage basins. Describing riverine landscape and its relation to biodiversity patterns and disturbances, Ward (1998) observed that the river channels are part of an extensive interconnected series of biotopes and environmental gradients, which together with, their respective biotic communities constitute lotic ecosystems. The effects of perturbation and alterations in the catchments can always be transmitted to the river channel and further downstream. Normally rivers exist in a dynamic equilibrium with landscape with changes in the whole river system happening only slowly (Jeffries & Mills, 1990). The current study was guided by the hypothesis that River Nyando's deltaic swamps modifies and filters pollutants from water entering Lake Victoria. This preliminary study noted that the ecological integrity of the Nyando delta is threatened yet its biodiversity is poorly documented. The delta is an important refuge for threatened fish species assemblages in Lake Victoria especially *Haplochromis spp*. Current exploitation of wetland vegetation (papyrus) and fishery resources in the delta is unsustainable and does have serious negative effects on wetland biodiversity, ecological functions and the local people's livelihood.

The overall goal of the study was to establish the significance of Nyando delta to the preservation of aquatic biodiversity and improvement of water quality in Winam Gulf of Lake Victoria, Kenya.

Study area

The Nyando river system where the study was conducted (Fig 1) lies in western part of eastern Rift Valley and flows into Lake Victoria through the Winam gulf. Lake Victoria (68,890 km^2) is situated across the equator at an altitude of 1134 m in East Africa and is shared by three countries namely Kenya (6%), Uganda (47%) and Tanzania (47%). It stretches 412 km from north to south between latitudes $0^0 30'$N and $3^0 12'$S and 355 km from west to east between longitudes $31^0 37'$ and $34^0 53'$E (Hughes & Hughes 1992). The lake has maximum and average depth of 79 m and 40 m respectively (Hughes & Hughes 1992, McClanahan & Young 1996). The lake shoreline estimated to be 3460 km is irregular and characterized by many headlands, inlets and islands that reflect the relief of the landscape existing prior to the formation of the lake, 750 years ago (Welcomme 1972). On the Kenyan side, it is drained by six major rivers which discharge an average of 7.29 billion m^3 and many minor rivers (Hughes & Hughes 1992). Detailed hydrological and physical characteristics of Lake Victoria and Winam gulf have been summarized by Burgis *& Symoens* (1987), Mavuti & Litterick (1991) and Crul (1995). The climate of the catchments area is of equatorial type (Walter *et al.,* 1960) with two main rainy seasons, the long rains occurring from March to May and the short rains from November to December.

River Nyando comprises of the river itself, the delta (76 km^2) and Nyakach bay in the Winam gulf. River Nyando has a total length of 153 km and a catchment area of 3450 km^2 (MOWD, 1992). The upper parts of the river are characterized by agricultural activities primarily of maize, sugar and tea plantations. The lower parts include the extensive Kano plains (90,000 ha) and the delta where rice growing and livestock rearing are the dominant human activities. The Kano plains experiences heavy floods during wet seasons and usually destroys crops, animals and property and have been recognised as one of the major handicaps to poverty eradication in Kisumu district (G.o.K, 1999). The floods are also associated with water borne diseases such as cholera, typhoid, malaria and bilharzias. Perhaps the escalating floods in the region can be attributed to degradation of wetlands.The delta is primarily vegetated by sedges, T*ypha spp, Cyperus papyrus*, hippo grass and water lilies from the landside to the lakeside. River Nyando receives domestic sewage from Ahero town, industrial wastes from sugar factories e.g. Muhoroni and Chemelil and agricultural pollutants from rice schemes, sugar and maize plantations. Overgrazing along the river has also been recorded (M'mayi *et al.,*1997).

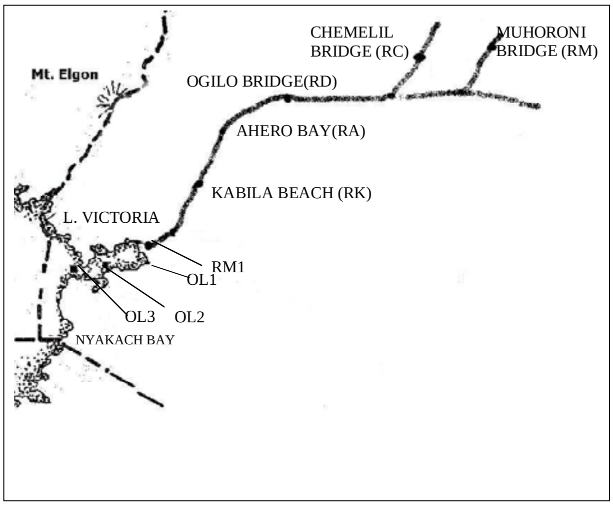

Figure 1. A sketch map-showing location of River Nyando and its delta. Modified from M'mayi *et al*., 1997.

Table 1. Description of study sites along the Nyando River with respect to major human activities .

Site name	Acronyms	Major human activities
Muhoroni bridge	RM	Sugar cane plantation dominated the vegetation on the upper parts of Nyando river valley. Close to the river, the principal vegetation consisted of open patches of star grass, with scattered Eucalyptus, Ficus and Croton trees. Cultivated land close to the river had maize, bananas and nappier grass for livestock. There was little soil erosion and the water was clear. The river flowed on a gravel bed and the water was covered with floating grass on the edges. The stones under water harboured numerous aquatic invertebrates.
Chemelil Bridge	RC	Trees growing close to the river included mature Acacia seyal, Ficus, young Eucalyptus, Lantana bushes and Sodom apple. Land close to riverbank was cultivated and planted maize and sugar cane. There were signs of soil erosion with deep gullies leading to the river edge. Sisal bushes prevent soil erosion on the upper parts of the riverbank. The water was brown characteristic of effluents discharged from Chemilil Sugar factory.
Ogilo bridge	RO	This site was used by livestock for drinking water. Sixty six cattle were counted over duration of 30 minutes. The river water was highly turbid. The eastern bank was steep with *Acacia seyal* trees, *Albizia gumifera* trees and Ficus trees. A few eucalyptus trees were planted at the water edge. Riparian forest that existed previously with abundant lianas had been destroyed. Only scattered native trees were left, charcoal burning was going on at the site during the field visit. Maize and nappier grass was planted at the riverbank in small plots of cultivated land. The river contained dried logs of trees and carried soil and abundant leaf litter. People were using the site to fetch water for domestic use and wash their clothes.
Ahero town	RA	This was an urban and residential area. Observation site was open with grass and scattered shrubs on either side of the river. Most trees especially the eucalyptus were planted. Local people utilized water for domestic use despite the fact that it was turbid. The site was also used as a watering point for livestock (56 animals) counted at the time of sampling. The river had solid waste especially plastic bags, plastic containers and paper originating from the town commercial activities at Ahero. Local people were also washing vehicles next to river and oil would be seen floating on the water.

Kabila beach	RK	This site was used as a fish-landing beach by the local fishermen. Inlets from the land to Nyakach bay were invaded and clogged with water hyacinth invasion. Drifting floating papyrus reeds and water hyacinth closed the passages and made it difficult for the fishermen to access Lake Victoria. During the study there were 19 fishing boats at the beach but according to the local fishermen the site used to have up to 45 boats before 1999 when fish catches were good. At the edges of the fringing marshes, there were 3 fishponds that were constructed by women group to rear tilapias. However, snakes, marsh mongoose and piscivorous birds hindered the establishment of the fish. They ate fish fries before they matured. There were plans to build chicken mesh wire enclosures around and over the ponds to keep away animal pests
Nyando delta	RM2	The river channel was accessed from the river mouth by using a small boat. Patches of pistia and water lilies were frequently encountered in the river channel. Leninia occurred at the water edges, where a dense fringe of hippo grass and dense (mature) papyrus stands occurred. The water lilies and water hyacinth had beautiful pink flowers.
Nyando delta	RM1	Isolated patches of water hyacinth, papyrus reeds and hippo grass fringed the river mouth. The latter was the dominant vegetation. Small patches of floating water lilies occurred at the edges of the extensive river mouth. Most of the water surface was open and the water was relatively clean, except for suspended swamp debris.
Nyakach bay	OL3	This is a semi-circular bay into which river Nyando drains. The water was clear and had small fish. The entrance to the bay was shallower than the bay itself. Dead tree trunks, implying a rise in Lake water level, were observed on the southeast near Kusa beach.
Nyakach bay	OL2	
Nyakack bay	OL1	

Materials and methods

Sampling strategy

The study was carried out during the dry month of July 2002. Sampling sites were selected along the Nyando river, the delta and Nyakach bay. Six sampling sites were selected along the river based on accessibility and land use practices. Two sampling sites were selected on the delta and three sites were selected in the lake depending on the distance from the river mouth. The idea was to determine the influence of the river on the lake. Sampling in the open lake was carried out using a research boat hired from KMFRI. To reach the river mouth from the bay, a small engine boat was used to access the shallow waters where the research boat could not reach. Small fishing canoes were used to navigate along the river channels in the delta. Locations of sampling sites were taken using Global positioning System (GPS) to enhance future monitoring.

Water physical-chemical characteristics were determined in situ using a Hydrolab (Survey 40). Parameters taken included pH, temperature (TEMP), turbidity (TURB), conductivity (COND), water depth and total dissolved solids (TDS). Water flow was determined using a flow meter (GO 11821). Biological oxygen demand (BOD) was determined by incubating water for four days using 500 ml BOD bottles and dissolved oxygen meters used to determine dissolved oxygen O_2 (Hanna H191410).

Assessment of biodiversity was also undertaken especially birds, aquatic macro-invertebrates, aquatic plants and fish. Birds were assessed by direct observation and spot count method (Verner1985) using pairs of binoculars. Approximately 30 minutes per site was spent on the counts. All records were made from one spot for all birds seen and metred within a radius of approximately 200 meters. These birds were recorded by walking over a stretch of 1 km and over a perpendicular distance of 100 m. Aquatic macro-invertebrates were sampled using pond nets and placed in white trays for sorting out and identification (Southwood 1978). They were also sampled from the soft mud by using a core sampler as well as scraping the bottom side of stones particularly in the upper parts of the river. Samples of aquatic plants were collected from the water surface and edges. They were subsequently preserved for further analysis in the herbarium department at National Museums of Kenya (NMK). Common fish from the lake were recorded from fish landing bays. Suitable biological specimens collected were carefully packed and taken to NMK for curation and preservation.

Data analysis

Multivariate, parametric and non-parametric methods were used to analyse the water physico-chemical data. An ordination computer programme, Principal Component Analysis (PCA) was used to classify river, delta and bay sites with respect to environmental parameters. Groups given were subjectively split and one-way Analysis of Variance (ANOVA) performed to test whether they were significantly different. Spearman rank correlation (Statistica software version 4.0) was used to calculate correlations between different environmental variables.

Results

Water physico-chemical characteristics

Measured water physico-chemical characteristics are presented in table 2. The data was used to classify sites using PCA and, the resulting groups or clusters given compared for differences in each environmental variable using one-way ANOVA. Correlation analysis was also performed using the environmental data.

Table 2 Water physico-chemical characteristics in Nyando River in July 2002. The sites are ordered according to decreasing altitude from top to bottom of the table.

Sample Site	Site code	Depth (m)	Water Temp. ^0C	TDS (mg^{-1})	COND μscm^{-1}	PH DO (mg^{-1})	TSS (mg^{-1})	Turb NH4 (mg^{-1})	NO_3 (mg^{-1})	Dsc. m^3s^{-1}	Alti. (m)
Muhoroni	RM	0.10	22.07	21.34	333.12	8.585.65	12.00	35.160.28	6.01	2.37	1520
Chemelil	RC	0.10	22.55	28.89	451.40	8.081.94	34.00	74.270.87	26.73	2.42	1216
Ogilo	RO	0.30	24.98	23.52	367.48	8.414.90	33.00	32.120.16	5.88	18.08	1181
Ahero	RA	0.26	25.69	22.90	357.45	8.454.82	102.5	95.780.34	7.70	35.19	1160
Kabila	RK	0.10	25.54	23.00	311.75	8.097.16	112.0	100.00.77	12.46	40.8	1150
Awasi	AS	0.20	28.96	15.00	238.30	7.833.76	547.0	300.01.09	42.83	*	1146
River mouth 2	RM2	0.10	25.63	21.18	330.50	7.872.51	20.00	68.000.17	5.75	*	1136
River mouth 1	RM1	0.30	26.74	19.42	304.20	8.084.79	27.00	62.000.32	2.73	*	1135
Nyakach Bay	OL3	0.30	25.00	12.50	195.70	9.185.56	90.00	20.200.36	38.47	*	1134
Nyakach Bay	OL2s	0.30	27.30	12.64	197.30	9.246.66	10.00	12.000.08	2.65	*	1134
Nyakach Bay	OL1s	0.30	24.50	11.92	186.80	8.905.56	12.00	18.600.16	0.20	*	1134

* Asterisk signifies sites whose water flow rate was not measured.

Association between physicochemical parameters

Results of the correlations analysis of the physico-chemical parameters are shown in table 3 and depicted by PCA diagram (Figure 2). Arrows represent environmental variables and the longer the arrow the higher the variation. Arrows close together signify high association between water parameters whilst stations clustered together have similar water physico-chemical characteristics. Dissolved oxygen was negatively but not significantly associated with turbidity and pH. Pollution indicating parameters (i.e. TSS, TDS, NH4, NO_3 and turbidity) were positively correlated (at p=0.01) with each other. Further figure 2 shows that, stations situated in the bay were associated with high O_2 and temperature, while river stations were associated with total suspended solids. The stations in the delta were characterised by high NH_4 and NO_3 (see also table 3 for correlation analysis)

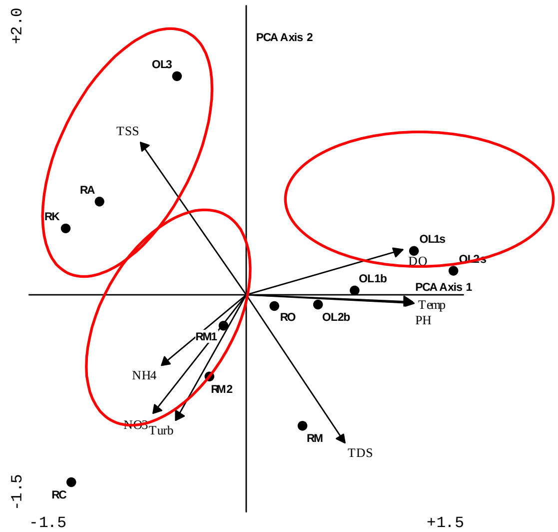

Figure 2. A PCA biplot of physical characteristics of water drawn from rivers and Winam gulf. Axis has a total eigen value of 1.0 indicating that the axis was important for ordination of sites with respect to physical chemical characteristics of water. The value explained by the first and second axes were 53 % and 24 %, respectively.

Table 3. Spearman rank correlation matrix letters between water physico-chemical characteristics from Nyando river, delta and the Nyakach bay a, b, c, represent significant correlation levels at (a= 0.05, b = 0.01, c = 0.001).

	TEMP	TDS	pH	O_2	TSS	TURB	NH_4
TDS	0.01	1					
pH	-0.47^a	-0.57^a	1				
O2	-0.06	-0.12	0.55^a	1			
TSS	0.52^a	0.49^a	-0.56^a	-0.18	1		
TURB	0.43^a	0.62^a	-0.81^c	-0.40^a	0.77^c	1	
NH_4	0.34	0.44^a	-0.61^a	-0.22	0.84^c	0.73^c	1
NO_3	0.26	0.77^c	-0.65^b	-0.16	0.76^c	0.79^c	0.81^c

Comparison of water physico-chemical characteristics from river and lake sites

The results of one-way ANOVA performed on physical chemical properties of water between the river sites and lake sites showed significant differences in TDS and pH (Table 4). The mean TSS, TURB, NH4 and NO3 values were far much higher in the lake than in the river. The swamp absorbed most of the pollutants so that river water had low levels.Variances approached but did not reach significant levels.

Table 4. A one-way ANOVA and mean values of physical characteristics of water among river and lake sites. (*= Significant).

Parameters	Lakes	Rivers	Both	df	F	P
TDS	12.4±0.42	21.9±3.9	18.2±5.7	11*	28.19*	0.000*
PH	8.2±0.28	9.02±0.18	8.5±0.49	11	36.35	0.000*
TEMP	25.27±2.2	24.55±0.6	25±1.8	11	0.496	0.496
DO	4.4±1.7	5.4±0.84	4.81±1.5	11	1.406	0.26
TSS	110.9±180.2	27.6±34.9	78.9±145.4	11	1.01	0.336
TURB	95.9±86.04	23.3±9.7	68±75.5	11	3.42	0.091
NH_4	0.5±0.35	0.176±0.11	0.38±0.32	11	3.798	0.077
NO_3	13.8±13.9	2±1.48	9.23±12.2	11	3.421	0.091

Biological resources

The survey showed that Nyando River is rich in aquatic macro invertebrates. The most common included mayfly nymphs (ephemeroptera), dragonfly nymphs (odonata), water beetles (coleoptera), and mosquito larvae (diptera). The aquatic invertebrates were abundant in upstream stations but were relatively scarce in lower (delta) stations (fig. 3). Detailed study of aquatic macro invertebrates is urgently needed.

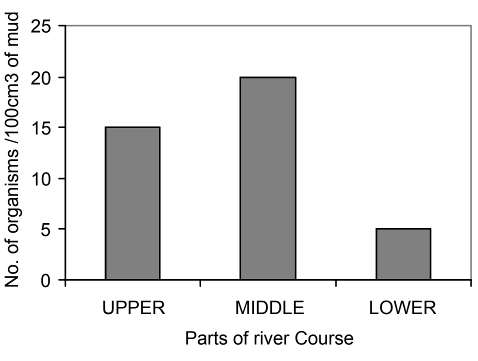

Figure 3. Abundance of invertebrates in different parts of the Nyando river course.

The delta is densely vegetated by macrophytes, which show zonation from the landside to lakeside. Some of the common ones are sedges and grass on landside. The tall, typha and cyperus sedges (*Typha latifolia*, *Cyperus papyrus*) dominated the floating macrophytes while hippo grass (*Echinocloa pyrimidalis*) and water lily (*Nymphaea caerulea*) were common on the lakeside. Lake waters were also vegetated by *Eichhornia crassipes* and *Pistia stratiotes.*

The Nyando river system is also endowed birds species. It had diverse habitats for birds, ranging from open water, margins of river channel and water pools, dense emergent macrophytes to open grasslands with scattered bushes and isolated forest patches. A total of 38 both aquatic and terrestrial bird species were recorded in eight sites located within the Nyando river system. Cattle egret and Grey crowned cranes were found in Kano flood plains. Common crake, African jacana and Black crakes were found in the floating aquatic vegetation. Hadada Ibis, Hammerkop and Grey Heron were found in shallow wetlands with grass or short sedges as well as in the numerous ponds, which were rich in fish. Farmland habitats and forest fragments on the upper parts of the river attracted relatively small birds especially granivores, insectivores and piscivores. The Nyando river mouth and Nyakach bay hosted King fishers, gray gulls, White Winged Black Terns and Cormorants. Generally species diversity and bird biomass increased downstream, from Muhoroni to Nyando delta and river mouth.

Discussion and recommendations

The results of this study indicated that water quality between the river and lake was significantly different only in total dissolved solids and pH. Correlations analysis showed that water pollution indicating variables (i.e. TDS, TSS, turbidity, NH_4 and NO_3) showed significant positive correlations among themselves. They were not associated with O_2 but were negatively associated with pH. Interestingly, oxygen was positively associated with pH. Multivariate analysis (PCA) showed that some open lake and river mouth sites were not clustered in either lake or river groups. They showed transition tendencies, which need to be confirmed in the next survey. Grouping of river mouth water from lake's waters need to be investigated further in order to confidently establish the buffering capacity of Nyando river delta.

Nyando River at Chemelil showed low O_2 concentration and the watercolour was brown. This was attributed to the effect of effluents from Chemelil and Muhoroni sugar factories. High-suspended solids down stream corresponded with high number of livestock grazing along the river and human waste from urban centers. Improper land use practices were predicted to be responsible for the large amount of riverbed sediment. The sediments are re-suspended as river flows and deposited in Nyakach Bay.

Nyando river system was rich in biodiversity. The vast and extensive deltaic swamps supported unique wetland vegetation, which showed zonation with respect to soil moisture and water characteristics. The vegetation provided habitat for different bird species and dictated their distribution in the floodplain and delta. This aspect was poorly investigated and the next study should shed some light into the trophic structure of the bird and fish communities.

Although data from the current study was insufficient to confidently establish the buffering capacity of the Nyando wetlands, this important role has been demonstrated in detailed studies in the Ugandan side of Lake Victoria. Studies to assess the filtration and buffering capacity of natural and artificial wetlands in wastewater treatment in Uganda showed that wetlands remove nutrients, faecal coliforms and other pollutants (Kizito, 1986, Kansiime *et al.,* 1994, Kansiime & Nalubega 1999, Kipkemboi *et al.,* 2002, Azza *et al.,* 2000, Kansiime, 2001, Kymabadde *et al.,* 2004). It has been suggested that almost all particulate matter accumulates in the shore zone provided the lake possess a proper supralittoral and littoral zone with vegetation of macrophytes. Lewis *et al.,* (1984) estimated that 20-25% of nitrogen input and 30-90% of phosphorous input is carried by particulate matter. Permanent burial of sedimented materials represent an ultimate sink for the intercepted pollutants like phosphorous (Howard-Williams & Gaudet, 1985). Adsorption of Phosphorous onto detritus takes place and has been reported to be main mechanism for its removal from the swamp water (Kelderman, 2004). For the intercepted nitrogen, denitrification is dominant process and occurs mostly at nutrient rich, oxygen limited swamp bottom (va Dam, 2004; Kansiime *et al.,* 2004). A denitrification rate as high as 2727 kg/ha*y has been measured in a reed swamp adjacent to a lake (Jørgensen *et al.,* 1988). Additional losses from the swamp water occur through adsorption of $NH4^+$ onto particulate matter and immobilisation by microorganisms (Kansiime, 2004). In this respect, aquatic macrophytes play a vital role in wetland physical, chemical and biological processes (Wetzel, 1993). The attached microorganisms help in the transformation of nutrients, metallic ions and other compounds entering a wetland (Hammer & Bastian, 1989). Sorrounding aerobic pockets are anaerobic zones where denitrification and pollutants transformation occur (Faulkner & Richardson, 1989). Also in the anaerobic zone, CO_2 and humic substances are produced which are largerly responsible for the acidic character of the swamp water. Wetlands acts as nutrient reservoirs of a number of pathways such as uptake and accumulations in plant tissues and sediment accumulation. The nutrient accumulated in plants can be removed by grazing and harvesting (Jørgensen, 1990; Kansiime 2004). Satouchi (1989) and Kurata & Vira (1986) have estimated the harvest of *Phragmites communis* from a lagoon adjacent to Lake Biwa will imply a removal of 36.6g nitrogen and 4.3g phosphorous per m^3.

It is important to underscore the role that macrophytes play in regard to the buffering capacity of wetlands. They act as both nutrients and sediment traps (Gaudet, 1974; Sloey *et al.,* 1978, Gilliam et al., 1988, William 1990, Boar *et al.,* 1999). They also trap heavy metals, biocides and other toxins in the water entering the lake from the catchment (Kadlec & Kadlec, 1979, Fritz & Helle, 1978; Maltby, 1986). With these important functions freshwater wetlands have been shown to act as wastewater treatment sites (Kiefer, 1968; Mitsch, 1976; Spangler *et al.,* 1976). It is recognised and accepted that the most management unit for conservation of fresh water taxa is river catchment (Darwall *et al.,* 2005). In order not to overload and overstretch the functioning capacity of wetlands downstream towards the lake, remedial measures should be undertaken at the catchment level. This paper while recognising the need to equally protect wetlands downstream recommends that proper land use practices and industrial activities should be controlled at the catchment sites. River Nyando river system which leads to Lake Victoria is one of the many sources of influence to the lake ecosystem. This influence has been shifted to the negative side because of wanton destruction of wetlands and uncontrolled human activities including industrial waste into the rivers. This phenomenon is evident in Lake Victoria with proliferation of water hyacinth and eutrophication due to high inputs of nutrients from the catchment areas. There is need therefore to take precautionary measures to reduce pollution discharges into the river systems draining into Lake Victoria. The way forward in this front is to have proper management of catchment areas where the pollutants are emanating from, strict control of industrial and sewage wastes, keeping wetlands intact especially swamps to help buffer most of the unavoidable pollution right from the catchment areas to the lake. It is hoped that with completion and implementation of Kenya Wetland Policy the fight to save lake Victoria ecosystem (on Kenyan side) and other wetlands countrywide will be strengthened.

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References

Azza N.G.T., Kansiime F., Nalubega M. & Denny P., 2000. Differential permeability of papyrus and *Miscanthidium* root mats in Nakivubo swamp, Uganda. *Aquatic Botany* 67: 169-178.

Balirwa J.S., 1995. The lake Victoria environment: its fisheries and wetlands: a review. *Wetlands Ecology and Management* 3:209-224.

Boar R.R., Harper D.M. & Adams C.S., 1999. Biomass allocation in *Cyperus papyrus* in a tropical wetland, Lake Naivasha, Kenya. *Biotropica* 31(3): 411-421.

Burgis MJ & Symoens J.J., eds., 1987. *African wetlands and shallow water bodies.* Council for Scientific and industrial Research, South Africa.

Crul R.C.M., 1995. *Limnology and hydrology of Lake Victoria*. UNESCO/IHP-IV Project M-5.1. UNESCO, Paris.

Darwall W., Smith K., Lowe T., & Vié J.-C., 2005. The status and Distribution of Freshwater Biodiversity in Eastern Africa. IUCN SSC Freshwater Biodiversity Assessment Programme. IUCN, Gland, Switzerland and Cambridge, UK. Vii+ 36pp.

Faulkner S.P. & Richardson C.J., 1989. Physical and chemical characteristics of fresh water wetland soils. In. Hammer D.A. ed. *Constructed wetlands for wastewater treatment*. Lewis publisher, Inc. Michigan, pp.41-72.

Fritz W. R. & Helle S. C., 1978. Cyperus wetland as natural treatment method for secondary effluents. In. Drew M. A., ed. *Enviromental quality through wetland utilization*. Symp. Coord. Council. Restoration of Kissi. Mmee. R. Valley & Taylor Creek Nubin. Slough Basin, Tallahasa, Florida.

Gaudet J.J., 1974. The normal role of vegetation in water. *In.* D.S. Mitchell, ed. *Aquatic Vegetation and its Use and Control*. UNESCO, Paris: 24-37.

Gilliam J.W., Chescheir G.M., Skaggs R.W. & Broadhead R.G.,1988. Effects of pumped agricultural drainage water on wetland water quality. In. Hook D.D. et al.,(eds.), The Ecology and Management of Wetlands, vol. 1. Croom Helm, London, pp. 275- 283.

G.o.K (Government of Kenya), 1999. *Nyando District Development Plan 1999-2001*. Nairobi, Kenya.

Hammer D.A. & Bastian R.K., 1989. Wetlands ecosystems: natural water purifiers? *In.* Hammer D.A., ed. *Constructed wetlands for wastewater treatment: Municipal, industrial and agricultura*l. Lewis publisher, Chelsea, pp.5-19.

Howard-Williams C. & Gaudet J.J., 1985. The structure and functioning of African swamps. In. Denny P., ed. *The Ecology and Management of African Wetland Vegetation*. Dr W. Junk Publishers. Dordretch, the Netherlands, pp. 153–175.

Hughes R.H. & Hughes J.S., 1992. *A Directory of African Wetlands*. IUCN, Gland, Switzerland and Cambridge, UK xxxiv + 820 pp.

Jeffries M. & Mills D.,1990. *Freshwater ecology: principles and application*. Belhaven Press, London and New York.

Jørgensen S.E, Hoffmann C.C. & Mitch W.J., 1988. Modelling Nutrient Retention by a Reedswamp and Wet Meadow in Denmark. *In.* Mitsch W.J., Straskraba M. & Jørgensen, eds. *Wetland Modelling*. Elsevier, Amsterdam.

Jørgensen S.E., 1990. Erosion and Fitration. *In.* Jørgensen S.E & Löffler H., eds. *Guidelines of Lake Management vol.3*. Lake shore management. International Lake Environment Committee and UNEP, pp.13-20.

Kadlec R.H. & Kadlec J.A., 1979. Wetlands and water quality. *In.* P.E. Greeson, Clark J.R. & Clark J.E., eds. *Wetland Functions and Values*: The State of Our Understanding. Minneapolis, MN: Water Resources Association Technical Publication, pp.436-456.

Kansiime F. & Nalubega M. 1999. *Wastewater Treatment by a Natural Wetalnd: the Nakivubo Swamp, Uganda. Processes and implications.* PhD. Thesis, A.A. Balkema Publishers, Rotterdam, The Netherlands.

Kansiime F. & Nalubega M., Bugenyi F.W.B. & Tukahirwa E.M., 1994. The role of Nakivubo swamp in maintaining the water quality of Inner Murchison Bay-Lake Victoria. *Journal of Tropical Hydrobiology and Fisheries* 5:79-87.

Kansiime F. & van Bruggen J.J.A., 2001. Distribution and retention of faecal coliforms in the Nakivubo wetland in Kampala Uganda. *Water Science and Technology* 44: 11-12.
Kansiime F., 2004. Functioning and dynamics of wetland vegetation of Lake Victoria. *In*. *Ecotools project, Tools for wetland ecosystem resource management in Eastern Africa.* EC RTD INCO-DEV Programme, pp.5-11.
Kelderman P., 2004. Plug-flow modelling of phosphorous retention in papyrus wetlands. *In. Ecotools project, Tools for wetland ecosystem resource management in Eastern Africa.* EC RTD INCO-DEV Programme, pp.43-52.
Kiefer W., 1968. *Biological waste water treatment with plants*. Umschau No. 7, 210. Trans. From German for the Great Lakes Basin Comm. And N. E. Wisconsin Regional Plan. Comm.
Kipkemboi J., Kansiime F. & Denny P., 2002. The response of *Cyperus papyrus* (L) and *Miscanthidium violaceum* (K. Schum) Robys to eutrophication in natural wetlands of Lake Victoria, Uganda. *African Journal of Aquatic Sciences* 27: 11-20.
Kizito Y.S., 1986. T*he evaluation of pollution levels in Nakivubo channel, Kampala-Uganda*. M.Sc. Thesis, Makerere University, Uganda.
Kurata A. & Satouchi M., 1989. *Function of a lagoon in Nutrient Removal in Lake Biwa*, Japan. In. Mitsch W.J. & Jorgensen S.E. (eds.), Ecological Engineering. John Wiley & Sons, New York.
Kyambadde J., Kansiime F., Gumaelius L. & Dalhammar G., 2004. A comparitive study of *Cyperus papyrus* and *Miscanthidium violaceum* –based constructed wetlands for wastewater treament in a tropical climate. *Water Research* 38:475-485.
Lewis W.M., Saunders J.F., Crumpacker D.W. & Brendecke C.M., 1984. *Eutrophication and land use*. Springer-Varlag, New York, Berlin, Heidelberg, Tokyo.
M'mayi P., Katua S. & Mbarire S.K. eds., 1997. *Buffering capacity of Lake Victoria wetlands and the sustainable utilization and management of wetlands and wetland products pilot project*. Proceedings of the induction workshop of the wetland management component of Lake Victoria Environment management programme. Busia, Villa Hotel, Nairobi, Kenya.
MacClanahan T.R. & Young T.P.,1996. *East African ecosystems and their conservation*. New York/Oxford, Oxford University Press, 452 pp.
Maher W., Batley G.E. & Lawrence I.,1999. Assessing the health of sediment ecosystems: use of chemical measurements. *Freshwater Biology*, 41:361-272.
Maltby E., 1986. *Waterlogged wealth. International Institute for Environmental and Development*. London and Washington D.C. Russel press. Nottingham, U.K.
Mavuti K.M. & Litterick M.R., 1991. Composition, distribution and ecological role of zooplankton community in Lake Victoria, Kenya waters. *Verhandlungen der Internationale Vereinigung für Theoretische und Angewandte Limnologie* 25: 846-849.
Mitsch W.J., 1976. Ecosystem modeling of water hyacinth management in Lake Alice. *Florida. Ecological Modelling,* 2: 69-89.
MOWD (Ministry of Water Development), 1992. *The study on the National water Master Plan. Sectoral Report (N): Environmental Conservation.* Government of Kenya and Japan International Cooperation Agency.
Pan Y. & Stevenson J.J.,1996. Gradient analysis of diatom assemblages in Western Kentucky Wetlands. *Journal of Phycology* 32:222-232.
Sloey W.E., Spangler, F.L. & Fetter, C.W.Jr., 1978. Management of freshwater wetlands for nutrient assimilation. *In.* Good R.E., Whigham D.F. & Simpson R.L., eds. *Freshwater Wetlands*. Academic Press Inc., New York, pp. 321-340.
Southwood T.R.E., 1978. *Ecological methods*. Chapman and Hall, London.
Spangler F.L., Sloey W.E. & Fetter C.W.Jr., 1976. *Waste-water treatment by natural and artificial marshes*. EPA-600/2-76-207.
van Dam A.A., 2004. Nitrogen retention modelling for papyrus wetlands. *In. Ecotools project, Tools for wetland ecosystem resource management in Eastern Africa*. EC RTD INCO-DEV Programme, pp.27-42.
Verner J., 1985. Assessment of bird counting techniques. *In.* Johnson R. F., ed. C*urrent ornithology*. Plenum Press, New York and London, Vol. 2.
Ward J.V., 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. *Biol cons* 83(3): 269-278.
Welcomme R.L., 1972. An evaluation of acadja method of fishing as practised in the coastal lagoons of Dahomey (West Africa). *Journal of Fish Biology* 4: 39-55.
Wetzel R.G., 1993. Contructed wetlands: Scientific foundations are crucial. *In.* Moshiri, G.A., ed. *Constructed wetlands for wetlands for water quality improvement.* Lewis publishers, Boca raton, pp.3-7.
Williams M., 1990. Understanding Wetlands. *In.* Williams M., ed. *Wetlands: A threatened Landscape*. Basil Blackwell Ltd, U.K, pp.1-41.

Can an introduced, non-indigenous species save the fisheries of Lakes Baringo and Naivasha, Kenya?

Britton J. Robert[1], Joseph Ng'eno[2], Joyce Lugonzo[3] & David M. Harper[4]

[1]National Fisheries Laboratory, Environment Agency, Bromholme Lane, Brampton, Huntingdon, Cambridgeshire, PE28 4NE, England
[2] Lake Baringo District Fisheries Department, PO Box 46, Kampii Ya Samaki, via Nakuru, Kenya
[3] Lake Naivasha District Fisheries Department, Lake Road South, Naivasha, Kenya
[4] University of Leicester, University Road, Leicester, England
Email: robert.britton@environment-agency.gov.uk

Abstract

Following a peak in declared catches in 1999 and 2000, the performance of the commercial fisheries of Lakes Naivasha and Baringo declined rapidly in 2001, a result of over-exploitation. Periods of fishery closure followed, but were unable to restore catches to former levels. Factors responsible were believed to relate to low lake levels and environmental degradation preventing the full regeneration of stocks. The species now dominating catches are introduced and non-indigenous, and have wide environmental tolerances. The lungfish Protopterus aethiopicus has been present in commercial catches in Baringo since 1984 and comprised 62% of catches in 2004. The carp Cyprinus carpio has only been present in Naivasha since 1998, yet in May 2005 comprised >90% of the catch. It is argued that as lungfish have low resilience to exploitation, the Baringo fishery requires careful regulation of effort to prevent further decline. However, the life history traits of carp ensure populations are highly resilient to exploitation, so the species should continue to provide increasing catch returns in Naivasha, but this is likely to have a large ecological cost.

Key words: Exploitation, *Cyprinus carpio*, *Protopterus aethiopicus*

Introduction

Lakes Naivasha and Baringo are both shallow, freshwater lakes in the Eastern Rift Valley of Kenya and have Ramsar status (Ramsar, 1995, 2002). The resources of the lakes have high social and economic importance to the region. Lake Naivasha is important for flower production (export), geothermal electricity generation, tourism and conservation (Harper *et al*. 1990). Lake Baringo is important for agriculture, with water diverted from the lake for irrigation (Hickley *et al*. 2004a). A feature of both lakes are fluctuating water levels, resulting from periods of low rainfall, with this accentuated in recent years by the increased agricultural use of water (Harper & Mavuti 2004). The turbidity of both lakes has also increased, with the high turbidity of Baringo a key ecological feature resulting from over grazing in the catchment. Both lakes support fish populations that are the basis of commercial fisheries. The Naivasha fishery was originally based on species introduced from 1925 (Muchiri & Hickley 1991), whereas the Baringo fishery had been sustained by endemic species (De Vos *et al*. 1998). The exploitation of the lakes' resources has to be balanced against the maintenance of a sustainable fish population. However, large fluctuations in catch returns suggest that a suitable balance has not always been achieved (Hickley & Harper 2002; Hickley *et al*. 2004a), with periods of fishery closure used to mitigate this.

The introduction of alien fish species and their subsequent establishment has been a feature of both lakes. The most recent introductions were the marbled lungfish *Protopterus aethiopicus* (Heckel, 1851) into Baringo in 1975 and the carp *Cyprinus carpio* L. into Naivasha in 1998. *P*. *aethiopicus* is endemic to Africa, having a large native range in East and Central Africa (Mlewa & Green 2004). The species first appeared in Baringo catches in 1984 (De Vos *et al*. 1998) and in conjunction with the endemic catfish *Clarias gariepinus* (Burchell, 1822), form a long line fishery. *C*. *carpio* was accidentally introduced into Lake Naivasha following the escape of some fingerlings from a flooded fish farm on the principal river that feeds the lake, the Malewa (Hickley *et al*. 2004b). This species, non-native to the African continent, was first recorded in the lake in 2001 and has appeared in the commercial fishery returns since 2002 (Hickley *et al*. 2004b). Given the importance of the commercial fisheries, it is crucial to understand how these introductions have affected subsequent catch returns and how the trends in catch statistics can be used to better manage the fishery. Therefore, the aims of this paper are to assess the current status of these fisheries and the impact and benefits of the introductions of *C*. *carpio* and *P*. *aethiopicus* on subsequent catch returns.

Materials and methods

Commercial fishery statistics for both lakes were obtained from the Fisheries Department of the Kenyan Government. The Lake Naivasha fishery is based on the use of gill nets. The Lake Baringo fishery comprises both a gill net fishery and a long line fishery. As the gill nets have a with a minimum mesh size of 4 inches that is highly selective in capturing fish only of >250mm, additional samples of the juvenile fish stocks in both lakes were also collected in September 2004 using gangs of monofilament gillnets. Each gillnet was 60m in length, 1.5m depth, with mesh size changes every 5m (8 to 50mm). All major habitat areas were sampled, with nets set at first light and collected 5 hours later.

Results

The 4 species exploited in the Lake Naivasha commercial fishery were *Oreochromis leucosticus* (Trewewas), *Tilapia zillii* (Gervais), *Micropterus salmoides* (Lacépède) and *C. carpio*. Samples collected in September 2004 suggested few fish of these species present >200mm, for example, *O. leucosticus* (Figure 1). Commercial fishery returns revealed fluctuating performance between 1987 and 2005 (Figure 2). Following a period of poor returns between 1994 and 1997, catches increased from 21 t y^{-1} to a peak of 439 t y^{-1} in 1999 (Figure 2). Thereafter, a rapid decline in monthly catch returns occurred and the fishery was closed for the whole of 2001. Since re-opening, catch returns have continued to be depressed, with a total catch of only 39 t y^{-1} in 2003 and 60 t y^{-1} in 2004. A 3 month closed season is now also imposed annually. Returns in the initial 5 months of 2005 suggest some improvement, for the total catch was 66 t.

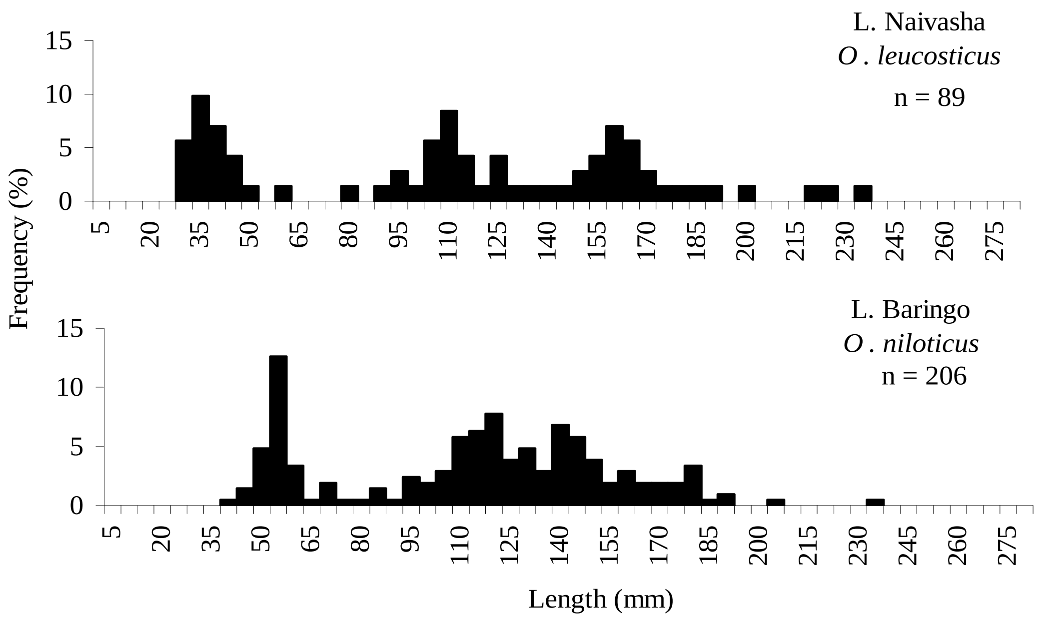

Figure 1. Length frequency distribution of the principal tilapia species in the fisheries of Lake Naivasha (top) and Lake Baringo (bottom), Sampled by gill nets, September 2004.

Periods of peak catch returns coincided with high contributions (up to 91%) of *O. leucosticus* in catches (Figure 2). In periods of low returns, the contribution of *O. leucosticus* also reduced. For example, in 1997 when the total catch reduced to 21 t y^{-1}, they comprised only 17% of catches. In 2004, there was a marked increase in the contribution of *C. carpio* to catches (Figure 2). By May 2005, the species comprised 90% of the total catch by weight, with the mean weight of individual fish being approximately 1.5kg, although individual fish were caught over 8kg. This reveals a fundamental shift in exploitation in the fishery, from *O. leucosticus* to *C. carpio*.

Within the fish community of Lake Baringo are four principal exploited species, *Oreochromis niloticus baringoensis* (Trewavas), *P. aethiopicus*, *C. gariepinus* and *Barbus intermedius australis* (Banister). Similar to Lake Naivasha, samples in September 2004 suggested few fish were present >200mm (Figure 1), although *P. aethiopicus* and *C. gariepinus* were unable to be sampled effectively using gill nets. Commercial catch returns also revealed fluctuating performance, with a similar trend of poor returns in the mid 1990s followed by increasing returns that peaked in 1999 and 2000 (Figure 3). Similar to Naivasha, a rapid decline followed, with the fishery closed in 2003 and 2004. Since re-opening, catch returns have continued to be depressed, with a declared catch of only 58 t y^{-1} in 2004 compared with 465 t y^{-1} in 2000 (Figure 3). Prior to fishery closure, *O. niloticus baringoensis* was the dominant species in the commercial catches and comprised up to 86% of total catch (1990). However, in 2004, they comprised only 4% of catches, whereas *P. aethiopicus* comprised 62% and *C. gariepinus* 33%. These species only feature in the long-line fishery. The length structure of the species exploited in the gill net fishery suggests there few fish present in size classes exploitable by legal mesh sizes (Figure 1) and in combination with their low commercial returns (Figure 3), suggests that this fishery may no longer be viable.

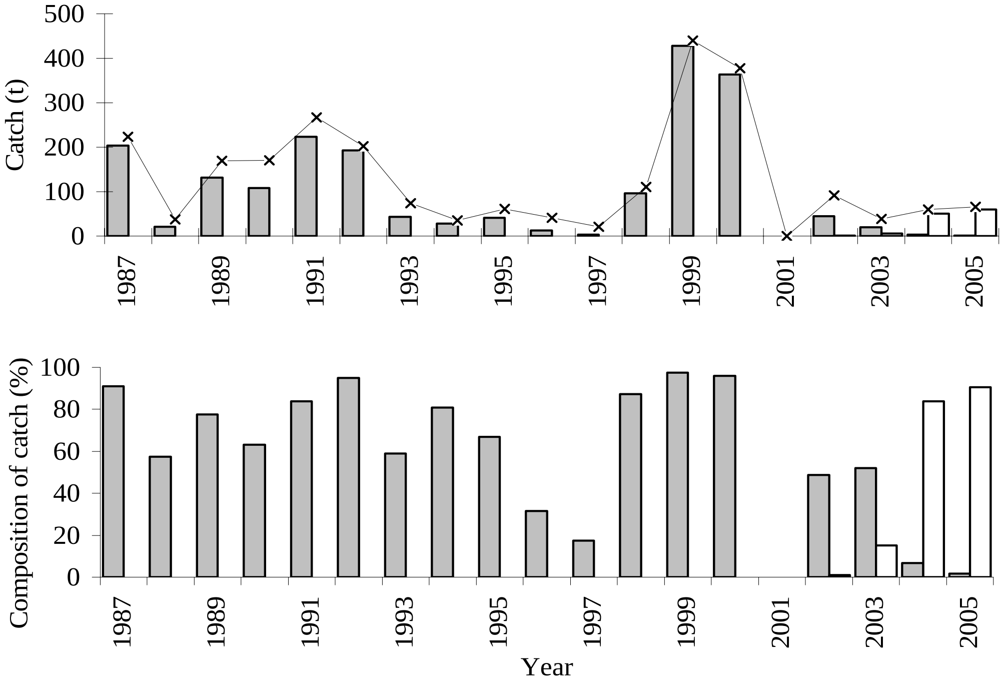

Figure 2: Lake Naivasha commercial fishery data: Top: Total annual catch (×), annual *O. leucosticus* catch , annual *C. carpio* catch . Bottom: Composition in annual catches of *O. leucosticus* and *C. carpio* .

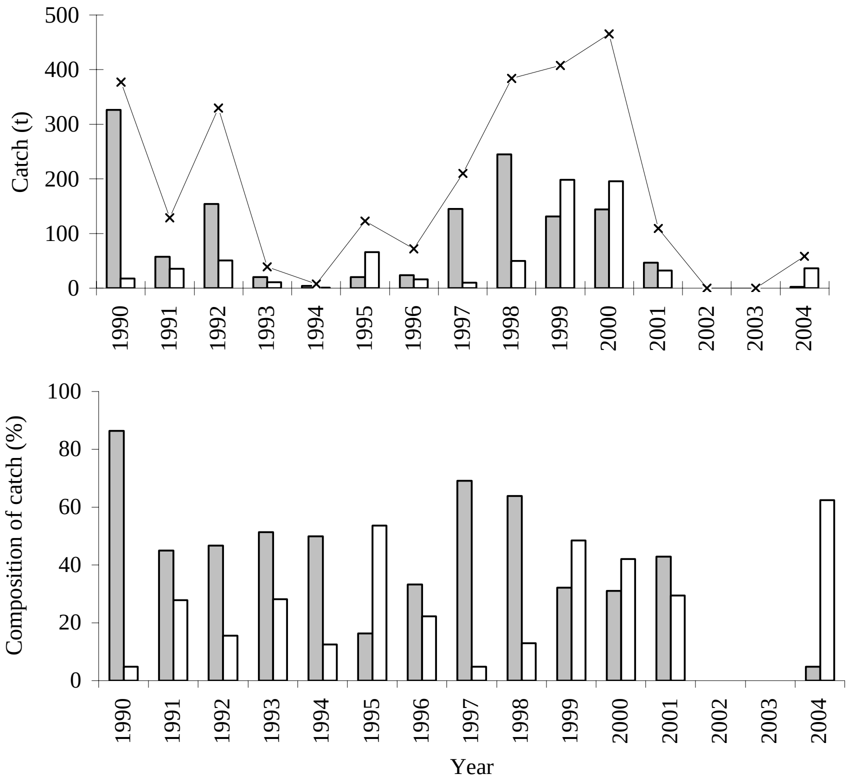

Figure 3: Lake Baringo commercial fishery data: Top: Total annual catch (%), *O. niloticus* catch , *P. aethiopicus* catch . Bottom: Composition in annual catches of *O. niloticus* and *P. aethiopicus* .

Discussion

In both lakes, the performance of the commercial fisheries continued to be poor in 2004, with catches still depressed from levels in 1999 and 2000, despite periods of closure imposed to allow stock recovery. This was mainly associated with continued low catches of the *Oreochromis* species, for historically they were the principal exploited species. In Lake Naivasha, catches of *O*. *leucosticus* have been strongly associated with fluctuating water levels; as water levels increased, there was a subsequent increase in catch returns (Hickley & Harper 2002). Similarly, catch returns in Lake Baringo increased as water levels increased and turbidity decreased (Hickley *et al*. 2004a). Therefore, the observed fluctuations in the catch returns are partially explained by erratic rainfall patterns (Figure 2, 3; Hickley *et al*. 2004a). *P*. *aethiopicus* and *C*. *carpio* are now the dominant species in the catches and if these species were absent, it is likely that the fisheries of both lakes would be no longer commercially viable.

Over exploitation was likely to be a factor in the catch declines that followed peak catch returns in 1999 and 2000, as a high intensity of fishing effort was expended. This compromised any potential benefit of fishery closure in restoring catch rates to their former levels. Whilst the depressed catch returns may relate to the continued low rainfall levels that the catchments have received (Hickley *et al*. 2004a), they may also reflect an inability in the fish stocks to regenerate to former levels due to the degree of habitat degradation in both lakes, for example, the increased turbidity, incidence of algal blooms and the loss of macrophytes (Hickley & Harper 2002). Whilst the role of habitat degradation in impeding stock recovery remains unclear, it is apparent the species now dominating catches in Naivasha (*C*. *carpio*) and Baringo (*P*. *aethiopicus*) both have wide environmental tolerances compared with the other exploited species. As *P*. *aethiopicus* can survive swamp conditions and periods of low dissolved oxygen, and feeding relies on non-visual cues (Greenwood 1986, Goudswaard *et al*. 2002), Baringo's degraded habitat is unlikely to adversely impact on their population. As *C*. *carpio* can survive in temperatures from 2 to 41^{o}C, pH from 5 to 10.5 and saturated oxygen levels as low as 7% (Koehn 2004), the current environmental conditions in Lake Naivasha are unlikely to pose an obstacle to their continued establishment.

Although limited remedial action is feasible in both catchments to initiate reversal of the degraded conditions (Hickley *et a*l. 2004a), the present conditions are likely to persist and so the presence of these tolerant species should continue to provide viable populations for exploitation. The life history traits of *C*. *carpio* - early sexual maturity, rapid growth, short generation time and high reproductive capacity (Koehn 2004) - ensure populations are resilient to high exploitation. In Naivasha, individual females attained lengths of 500mm in approximately 2 years, with production of up to 300,000 eggs per individual female (unpublished data). Therefore, it is likely that their catches in Naivasha will continue to increase. Fishers are increasingly utilising specialised gears to exploit the carp, with gill net mesh sizes up to 10 inches now used to selectively target individuals >5kg. These gears are unable to exploit the other species, with the reduced exploitation rate facilitating their population recovery. Whilst *P*. *aethiopicus* is a species that can still thrive in Baringo, their tolerance of high exploitation is poor compared with *C*. *carpio*. Although nest building and paternal care is utilised in the reproductive process (Goudswaard *et al*. 2002), the fecundity of individual females is low, as shown by females in Lake Baringo of lengths 770 to 1250mm producing only between 4,179 and 16,528 eggs (Mlewa & Green 2004). This results in population generation times being relatively long, increasing their vulnerability to the adverse impacts of high exploitation. This was demonstrated in Lake Victoria, where a 96% decrease in catch rate occurred between 1986 and 1990, with a principal factor being fishing pressure (Goudswaard *et al*. 2002).

Although the establishment of *C*. *carpio* in Lake Naivasha has brought demonstrable benefits to the commercial fishery, this may be to the detriment of the lake's already degraded habitat and affect efforts to restore the lake to its pristine state (Enniskillen 2002). Introductions of *C*. *carpio* into shallow waters usually result in subsequent severe habitat degradation, a consequence of their foraging behaviour (Wheeler *et al*. 2004). This involves sucking in sediments with prey items and retaining food organisms whilst sediment particles are expelled (Lammens & Hoogenboezem 1991; Sibbing 1988), uprooting aquatic plants, suspending sediment and increasing water turbidity (Petr 2000). This affects components of the ecosystem such as light availability for photosynthetic organisms and reduced visibility for fauna (Kirk 1985). Examples of such adverse impacts are available from countries including N. America (Crivelli 1983; Crowder & Painter 1991), India (Nandeesha *et al*. 1989) and Australia (Koehn 2004).

In summary, the performance of these commercial fisheries remains poor, with continued depressed catches of the traditionally exploited species. Periods of fishery closure have so far been ineffective in enabling stock recovery, possibly because exploitation levels during the open season have not been adequately controlled. The fishery management should recognise the slow and long term nature of the recovery process and aim to regulate long-term effort at appropriate levels. Reliance upon recently introduced species with wide environmental tolerances now provides the only commercially viable option for the fisheries of both lakes. However, the ability of *P. aethiopicus* to withstand exploitation appears low and the presence of *C. carpio* may be to the detriment of an already degraded habitat. Therefore, whilst their introduction may have provided these fisheries with a future, what will be the ecological cost?

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References

Crivelli, A.J. (1983) The destruction of aquatic vegetation by carp. *Hydrobiologia* 106, 37-41.

Crowder, L.B. & Painter, D.S. (1991) Submerged macrophytes in Lake Ontario: current knowledge, importance, threats to stability, and needed studies. *Canadian Journal of Fisheries and Aquatic Sciences* 48, 1539-1545.

De Vos, L., Pertet, F., Vanlerberghe, K., Nuguti, S. & Ntiba, M.J. (1998) Present status of the fish fauna and fisheries of Lake Baringo, Kenya*. Proceedings of International Conference for the Paradi Association and the Fisheries Society of Africa, Grahamstown (South Africa)*, 13-18. FISA/PARADI: Grahamstown (South Africa).

Enniskillen (2002) The Lake Naivasha Management Plan – consensus-building to conserve an international gem. *Hydrobiologia* 488, ix-xii.

Goudswaard P.C., Witte F & Chapman C.J. (2002) Decline in the African lungfish (*Protopterus aethiopicus*) in Lake Victoria (East Africa). *East African Wildlife Society* 40, 42-52.

Greenwood P.H. (1986) The natural history of African lungfish. *Journal of Morphology* (Supplement 1) 163-179.

Harper D & Mavuti K (2004) Lake Naivasha, Kenya: Ecohydrology to guide the management of a tropical protected area. *Ecohydrology & Hydrobiology* 4, 287-305.

Harper, D.M., Mavuti, K.M. & Muchiri, S.M. (1990) Ecology and management of Lake Naivasha, Kenya, in relation to climatic change, alien species introductions and agricultural development. *Environmental Conservation* 17, 328-336.

Hickley P, Muchiri M, Boar R, Britton JR, Adams C, Gichuru N & Harper D (2004a) Habitat degradation and subsequent fishery collapse In Lakes Naivasha and Baringo, Kenya. Ecohydrology & Hydrobiology 4, 503-517.

Hickley P. & Harper, D.M (2002). Fish community and habitat changes in the artificially stocked fishery of Lake Naivasha, Kenya. Chapter 20 of In *Management & Ecology of Lake & Reservoir Fisheries* ed. I.G. Cowx, pp 242-254. Fishing News Books, Blackwell Scientific Publications, Oxford.

Hickley, P., Muchiri, S.M., Britton, J.R. & Boar, R.R. (2004b) Discovery of carp (*Cyprinus carpio*) in the already stressed fishery of Lake Naivasha, Kenya. *Fisheries management and Ecology* 11, 139-142.

Koehn J.D. (2004) Carp (*Cyprinis carpio*) as a powerful invader of Australian waterways. *Freshwater Biology* 49, 882-894.

Lammens, E.H.R.R. & Hoogenboezem, W. (1991) Diets and feeding behaviour. In: *Cyprinid fishes: systematics, biology and exploitation* (I.J. Winfield and J.S. Nelson, eds). London: Chapman and Hall, pp. 353-376.

Mlewa C.M. & Green J.M. (2004) Biology of the Marbled lungfish *Protopterus aethiopicus* Heckel in Lake Baringo, Kenya. *African Journal of Ecology* 42, 338-345.

Muchiri, S.M. & Hickley, P. (1991) The fishery of Lake Naivasha, Kenya. *In Catch Effort Sampling Strategies: Their application in freshwater fisheries management* (I.G. Cowx, ed.) pp 382-392. Oxford: Fishing News Books, Blackwell Scientific Publications.

Nandeesha, M.C., Keshavanath, P., Basavaraja, N., Varghese, T.J. & Srikanth, G.K. (1989) Biological control and utilisation of aquatic weeds in India – a review. *Indian Journal of Animal Sciences* 59, 1198-1205.

Petr, T. (2000) *Interactions between fish and aquatic macrophytes in inland waters. A review.* FAO Fisheries Technical Paper No. 396. Rome: FAO, 185 pp.

Ramsar (1995) Information Sheet on Ramsar Wetlands (RIS) - Kenya, Lake Naivasha. http://www.wetlands.org/RDB/Ramsar_Dir/Kenya/ke002D02.htm

Ramsar (2002) Information Sheet on Ramsar Wetlands (RIS) - Kenya, Lake Baringo. http://www.ramsar.org/ris_kenya_baringo1.htm.

Sibbing, F.A. (1988) Specializations and limitations in the utilization of food resources by the carp, *Cyprinus carpio*: a study of aoral food processing. *Environmental Biology of Fishes* 22, 161-178.

Wheeler A.C., Merrett N.R. & Quigley D.T.G (2004) Additional records and notes for Wheeler's (1992) List of the Common and Scientific Names of Fishes of the British Isles. *Journal of Fish Biology* 65 (Supplement B) 40p.

The effect of water Hyacinth, *Eichhornia Crassipes,* infestation on phytoplankton productivity in Lake Naivasha and the status of control

John Momanyi Mironga

Department of Geography, Faculty of Environment and Resource Development, Egerton University, PO Box 536, Njoro, Kenya
Email: mmironga@yahoo.com

Abstract

The paper presents data collected in an assessment of effects of water hyacinth infestation on phytoplankton productivity in Lake Naivasha. A summary of the status of control and strategies for the future is given. The ecological effects of water hyacinth, Eichhornia crassipes, on Lake Naivasha have received little attention compared to the large body of work available on the weed's socioeconomic impact on the country's water ways and methods for its removal. This study was conducted to determine if water hyacinth infestation in Lake Naivasha affects phytoplankton productivity. Several sampling stations were set up in the lake at sites where the floating mats of the weed were present and sites where the weed was absent. Phytoplankton chlorophyll-a concentration and dissolved oxygen were measured at each station and used as proxies for phytoplankton productivity. The study findings show that phytoplankton productivity is reduced when water hyacinth is present, suggesting that the water hyacinth is not only a nuisance but that it can also alter the ecology within a lake by changing species composition and biodiversity. Although water hyacinth has continued posing serious ecological consequences, there is hope that the control strategies already adopted will continue to reduce deleterious impacts and allow sustained development in the Lake Naivasha Basin. There is, however, a great need to undertake research to quantify the levels of damage, and the costs of control, loss of livelihood, disease, and disruption of normal operations caused by water hyacinth.

Key words: Water hyacinth, Ecological Effects, Phytoplankton

Introduction

Water hyacinth, *Eichhornia crassipes* (Mart.) Solms. - Laubach (Figure 1) is considered one of the world's worst water weeds (Holm *et al*., 1977), invading lakes, ponds, canals, and rivers. It was introduced into many countries during the late 19th and early 20th centuries, where it spread and degraded aquatic ecosystems. It has such a high growth rate that, according to Ntiba *et al.,* (2001), it can double its area in only five days. It is still rapidly spreading throughout Africa, where new infestations are creating life-threatening situations as well as environmental and cultural upheaval (Cock *et al*., 2000).

The water hyacinth first appeared in Lake Naivasha, in 1988. It subsequently spread throughout the entire lake but was particularly prevalent in northern shallow inshore waters. The present-day cover by water hyacinth has remained relatively stable. It usually forms a narrow fringe 5-15 meters wide around much of the lake. *Eichhornia crassipes* remains the world's most problematic water weed despite widespread and various approaches to its control (Heard & Winterton, 2000). Its control at Lake Naivasha has focused upon biological control measures with the introduction of the Eichhornia weevil (*Neochetina spp.*) in the late 1990s. Because the enormous floating mats of the weed interfered with boat navigation, fishing, and even clogged up irrigation canals around the lake (Adams *et al*., 2002), much of the attention devoted to the water hyacinth in the literature has been concerned primarily with its socio-economic impact and methods for eradicating it from the lake. In contrast, fewer studies have focused directly on its ecological effects. It is known, however, that mats of water hyacinth reduce light to submerged plants, thus depleting oxygen in aquatic communities (Ultsch, 1973). The conditions under the water hyacinth mats are highly anoxic because of dead plant matter (Ntiba *et al*., 2001). The resultant lack of phytoplankton (McVea & Boyd, 1975) alters the composition of invertebrate communities (O'Hara, 1967; Hansen *et al*., 1971), ultimately affecting fisheries. Drifting mats scour vegetation, destroying native plants and wildlife habitat. Water hyacinth also competes with other plants, often displacing wildlife forage and habitat (Gowanloch, 1944). Higher sediment loading occurs under water hyacinth mats due to increased detrital production and siltation. In addition, the roots of the water hyacinth have been found to provide new habitat for gastropods that are intermediate hosts of the waterborne parasite that causes schistosomiasis (Masifwa, 2001).

Water hyacinth is known to cause a reduction on productivity of a lake's phytoplankton since the weed mats shade out any photoautotrophs (both phytoplankton and also submersed macrophytes) beneath them (Scheffer *et al*., 2003). The calming of the water by the floating mats reduces upwelling of nutrients from the sediments by wind action, making them less available to phytoplankton in the photic zone, and large aggregations of *Eichhornia crassipes* rapidly remove nitrogen and phosphorus from the water column (McVea & Boyd, 1975), out competing the phytoplankton for these vital nutrients. Exploitative competition among aquatic plants occurs for limiting resources, e.g. light, nutrients and suitable substrates (Barrat-Segretain, 1996). In addition to competition for limiting resources, aquatic plants sometimes compete with allelopathy, i.e. actively suppressing their neighbours by release of chemical compounds (Gopal & Goel, 1993). Novel mechanisms of competition, such as allelopathy, can affect native plants to a much larger extent than the alien's natural competitors (Callaway & Aschehoug, 2000). There is evidence of allelopathy by water hyacinth on phytoplankton (Yang *et al*., 1992).

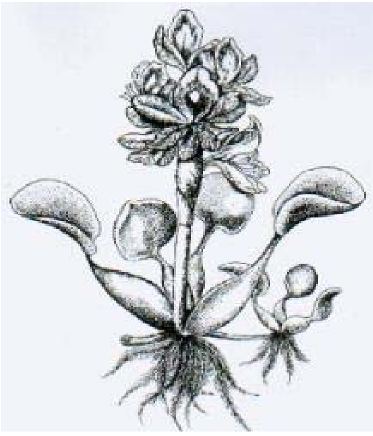

Figure 1. Water hyacinth.

The study findings show that phytoplankton productivity is reduced where water hyacinth is present in the lake. This is discussed in the context of water hyacinth altering the ecology of Lake Naivasha by changing species composition and biodiversity. In addition the current control strategies being used to combat water hyacinth are reviewed. There is, however, a great need to undertake research to quantify the levels of damage, and the costs of control, loss of livelihood, disease, and disruption of normal operations caused by water hyacinth in Lake Naivasha.

Study area

Lake Naivasha (0. 45ºS, 36. 26ºE), altitude 1890, lies in the Eastern Rift Valley and currently covers approximately 100 km^2 (Figure 2). It is the second-largest freshwater lake in Kenya (after the Kenya portion of Victoria). It is one of a series of 23 major lakes in the Eastern Rift Valley – eight in central Ethiopia, eight in Kenya and seven in Tanzania – spanning latitudes from approximately 7º N to 5º S. The overall climate of the Eastern Rift Valley is semi-arid. Most Eastern Rift Valley lakes are thus alkaline or saline. Lake Naivasha is unique within the central latitudes of the valley in being fresh, and indeed within the Kenyan series of lakes (from north to south are Turkana, Baringo, Bogoria, Nakuru, Elmenteita, Naivasha, Magadi) with a conductivity fluctuating between 250-450 µS cm^{-1}.

There are four, chemically distinct, basins at Naivasha (Gaudet & Melack, 1981). Crescent Island Basin, a small extinct volcanic cone, is the deepest part of the lake (up to 18m depth) and is half submerged and usually connected to the main lake over a shallow lip. The main lake has a maximum depth of 6m at its southern end. Oloidien is a smaller crater lake with a depth of 5m to the south end of the main lake, which has been distinct from it since 1982, increasing in conductivity from 250 to 3000µS cm^{-1} in this time. Crater Lake or Sonachi is located on the southwestern part of the lake and, in its own distinct volcanic crater, is a soda lake, fully independent from the main lake but its levels are believed to oscillate in harmony with the main lake as a result of groundwater connection. Table 1 illustrates some of the characteristics of the four water bodies.

Table 1. Water bodies characteristics.

Water body	* Area (Km^2)	* Volume (m^3 x 10^6)	Mean depth (m)	Maximum depth (m)
Lake Naivasha	145	680	4.7	7.3
Basin	2.1	23	11.0	17.0
Oloiden	5.5	31	5.6	6.1
Sonachi	0.6	0.62	3.8	6.1
Crescent Island				

(* Area and volume will depend on the lake level) Source: Melack (1976) cited in Goldson, (1993)

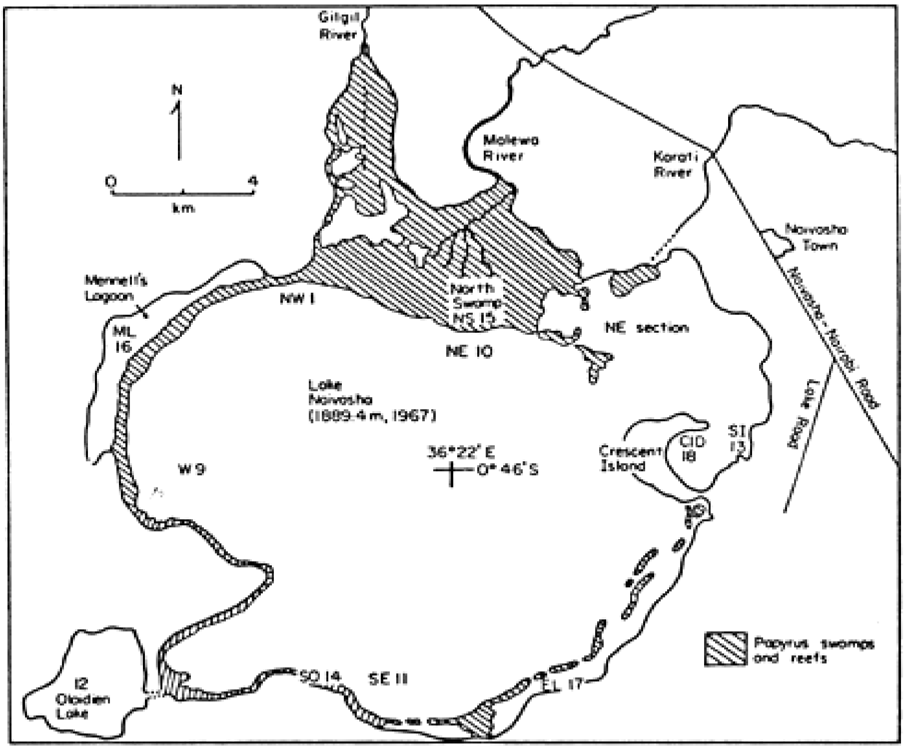

Figure 2. Lake Naivasha.

Since the early 1990s, the lake has become eutrophic. Its phytoplankton has showed a seasonal shift between diatom and cyanobacterial dominance and its assemblage is now dominated by a persistent *Aulacoseira italica* population, both numerically and in terms of contribution to overall primary production (Hubble & Harper, 2002). The concentrations of chlorophyll-a have increased from 30 $\mu g\ l^{-1}$ in 1982 to 110 $\mu g\ l^{-1}$ in 1988, and 178 $\mu g\ l^{-1}$ in 1995 and transparency has correspondingly declined to about 60 cm (but briefly rose to 160 cm in 1998-9 due to the diluting effect of the 'El Niño' rains) (Harper *et al.*, 2002c). 170 algal and cyanobacterial species have been identified (Hubble & Harper, 2002a). Most of the diatoms are indicators of moderate to high nutrient conditions. Total primary productivity of this phytoplankton population is approximately 160 mg C m^{-3} hr^{-1} (Hubble & Harper, 2002b). The sediments form a sink for phosphorus (Kitaka *et al.*, 2002), because they are rich in iron (Harper *et al.*, 1995) and the main lake is well mixed and does not deoxygenate enough to release this store of nutrients. However, Crescent Island lagoon does stratify temporarily and hypolimnetic deoxygenation occurs. Phosphorus is then released from the sediments, a process not seen in the main lake. This indicates that the rate of primary production in the water column could double if conditions change to allow lake-wide nutrient release from sediments (Hubble & Harper, 2002b). Kitaka, Harper & Mavuti (2002) showed that the lake did become 'hyper-eutrophic' on the OECD classification after the 'El Niño' rains in 1998, reverting back to eutrophic in 1999; this emphasises that most of the increase in trophic state of the lake comes from the wider catchment in the absence of the 'buffering' formerly provided in the North Swamp at the river inflows. The more alkaline Oloidien and Sonachi lakes are highly productive and *Arthrospira fusiformis* is significant in the latter.

Materials and methods

The study was conducted during the rain season in April 2004. Sampling took place in the lake at ten sites in a monitoring program previously established by the Kenyan Marine and Fisheries Research Institute, five of which were covered by the water hyacinth. Those stations that had no water hyacinth were used as controls. Chlorophyll-a concentrations and oxygen concentrations were measured at each station two times per week in the morning and in the evening for a total of five weeks. Chlorophyll-a concentrations were used as a proxy for phytoplankton production. As per Lung'ayia *et al.*, (2000), 0.5 ml saturated magnesium carbonate ($MgCO^3$) suspension was added to water samples of 50-500 ml and then the samples were immediately filtered through glass fiber filters. The filters were then extracted in cold 90% acetone for 18 to 24 hours. Dissolved oxygen at the surface was ascertained through Winkler titration as in Hecky *et al.*, (1994). A Hydrolab SVR-II profiling system, calibrated with the surface dissolved oxygen (DO) measured through Winkler titration was used to measure DO at depths of zero, 1, and 2 m at the shallower stations 1, 2, 3, 4, and 5 and at 5 meter intervals at station 6, 7, 8, 9 and 10 which was considerably deeper. The concentrations for each depth at each station were averaged at the end of the five weeks and the standard deviation for each was calculated.

Results

Impact of water Hyacinth on Phytoplankton productivity

Average surface chlorophyll-a concentrations at stations 1, 2, 3, 4 and 5, the stations with water hyacinth mats, were 4.6, 3.8, 5.0, 11.2, and 9.3 mg•m^{-3}, respectively. At the stations that were free of the water hyacinth, stations 6, 7, 8, 9 and 10, mean surface chlorophyll-a concentrations were 15.1, 14.6, 15.2 13.4, and 16.8 mg•m^{-3}, respectively (Figure 3). Mean dissolved oxygen concentrations at the stations under floating *Eichhornia* mats, sites 1, 2, 3, 4, and 5, were consistently lower than at control stations 6, 7, 8, 9 and 10 (Table 2).

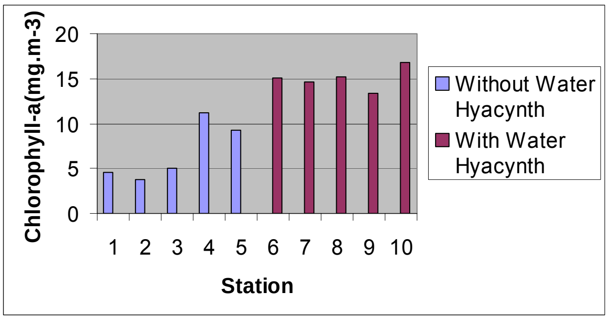

Figure 3. Bar graph of surface chlorophyll-a concentrations given in mg•m^{-3} at each station in Lake Naivasha. Lower chlorophyl-a concentrations were measured at the stations with *Eichhornia crassipes* cover, indicating lower productivity.

Table 2. Mean dissolved oxygen concentrations for each station at different depths.

Depth(m)	**Stations**									
	1	2	3	4	5	6	7	8	9	10
0	3.30	6.62	3.10	4.04	6.76	7.16	6.8	8.9	6.1	6.3
5	2.6	2.8	1.89	2.24	2.43	4.40	6.6	7.16	6.8	6.14
15	1.8	4.33	1.34	1.12	2.2	2.6	2.8	3.2	3.1	3.6

Water Hyacinth control strategies in lake Naivasha

Although water hyacinth has continued posing the above discussed impact on the phytoplankton productivity in Lake Naivasha basin, there is reason to hope that the control strategies adopted will eventually permit effective management of the weed.

Biological control

Since 1996, Kenya Plant Health and Inspectorate Services allowed KARI to import adult *N. bruchi* and *N. eichhorniae* from Uganda, South Africa and Australia for the biological control of water hyacinth in Lake Naivasha (Table 3). KARI established a second weevil rearing facility in December 1996, at the National Fibre Research Centre (NFRC), Kibos, near Lake Victoria. ‘Breeding stock’ for the Kibos rearing facility was obtained from the quarantined mass rearing facility at the National Agricultural Research Centre, Muguga, near Nairobi. The breeding material consisted of mature adult *Neochetina* weevils and host plants inoculated with weevil eggs. Later, adult *Neochetina* weevils were imported from Uganda for mass rearing. Julien *et al*., (1999) describe in detail rearing and harvesting techniques for *Neochetina* weevils from plastic tubs, rearing pools and galvanized corrugated iron sheet tanks, all of which have been in use at the Kibos rearing facility.

Table 3. Importations into Kenya of *Neochetina* weevils for biological control of water hyacinth in Lake Naivasha and Victoria.

Species	Year imported	Number	Purpose	Source
Neochetina bruchi	1996	1300	Mass rearing	Uganda
	1997	2000	Mass rearing/releases	Australia
	1998	1000[a]	Releases	South Africa
Neochetina eichhorniae	1997	5000	Mass rearing/releases	South Africa
	1997	2000	Mass rearing/releases	Australia
	1998	1000	Releases	South Africa
Total		12,300		

[a]Batch did not survive

From December 1996 to December 1999, the Kibos rearing facility and community rearing facilities produced approximately 100,000 adult weevils, of which 25,000 were for 'breeding stock' and for releases in Lake Naivasha. Between January 1998 and December 1999, approximately 23,200 adult weevils were released at several sites in Lake Naivasha.

Monitoring the establishment and spread of *Neochetina* weevils

Visual observations and pre and post-release sampling protocols have been used to monitor and evaluate the establishment, spread and impact of the *Neochetina* weevils on water hyacinth in Lake Naivasha. Weevils are now fairly established in all affected areas and have spread several metres from points of release. These natural enemies on the weed have been observed to have a significant impact and localised complete suppression of resident water hyacinth mats has been recorded at some sites some years after release Importation and mass rearing of additional biological control agents, the moth *Niphograpta albiguttalis* (previously called *Sameodes albiguttalis*) and mite *Orthogalumna terebrantis,* was attempted, but these did not establish well.

Evaluation of the impact of *Neochetina spp.* weevils on water hyacinth

In general, post-release sampling data collected (November 2003 to May 2004) at four selected release sites in the lake, indicated a suppression of plant growth parameters (fresh weight, leaf laminar area and leaf length) and substantial increases in number of feeding scars and adult weevils per plant. Fresh weight reduction was noted at a single site, mouth of River Malewa. Leaf length reduction was noted at two sites, while leaf laminar area reduction was evident at Hippo point and Crescent. The number of feeding scars and adult weevils per plant increased at all sites.

Estimations of weevil populations

Post-release sampling of water hyacinth at six selected sites in the lake (May–December 2004), gave a combined mean number of 6.0 *Neochetina* weevils per plant, with actual number of weevils per plant ranging from 0 to 32. *N. bruchi* was the dominant of the two weevil species, accounting for 73.3% of the total weevil population.

Physical control

Manual removal

The fisher-folk communities around Lake Naivasha have identified key sites for manual removal. These include fish-landing beaches, ports and piers, irrigation canals and water supply points and sources. Fish landing beaches in most of the affected areas are the prime targets for manual removal operations.

Mechanical control

Mechanical control operations are not common in Lake Naivasha and have so far consisted solely of chopping and dumping of the chopped pieces of water hyacinth and other weeds into the lake. Regrowth of the chopped weed is likely to take place, especially if most of the natural enemies are destroyed during chopping. In addition, shallow areas of the lake are likely to fill up with vegetation, especially along the shoreline, leading to drying up and subsequent reduction in the size of the lake. The future of mechanical control options in Lake Naivasha should be reassessed.

Ecological succession

Ecological succession (progressive displacement of one or more species of plants by other species) has made a significant contribution to the control of stationary mats of water hyacinth along the shores and banks of rivers entering Lake Naivasha. In the lake, pure mats of water hyacinth were invaded initially by aquatic ferns/ sedges (*Cyperus papyrus* and *Ipomea aquatica*) often to be followed by hippo grass (*Vossia cuspidator*) which invariably eventually started dominating and shading out the stressed and dying/rotting water hyacinth. By November 2004, stunted and disintegrated mats of water hyacinth and invading weed succession were clearly evident. Although water hyacinth will be a permanent feature in Lake Naivasha, currently hippo grass and not water hyacinth might form the dominant weed. The hippo grass is expected to die once the nutrients from dying water hyacinth are depleted.

Discussion and conclusion

Impact of Water Hyacinth on the Phytoplankton productivity

Invasions of water hyacinth have become a nuisance worldwide (Drake and Mooney, 1989). Originally perceived as a practical problem for fishing and navigation, water hyacinth is now considered as well a threat to biological diversity, affecting fish faunas, plant diversity and other freshwater life and the food chains, which depend upon it (Luken & Thieret, 1997).

Due to its physical presence water hyacinth greatly blocks sunlight and oxygen exchange and hence prevents growth of emersed and submerged plants. As a result, submerged macrophytes are scarce or absent in Lake Naivasha, while floating species dominate the macrophyte community in the littoral zones of the lake. Before the expansion of water hyacinth in the lake, submerged and rooted floating-leaved macrophytes were common in shallow parts (Gaudet, 1977). The loss of submerged macrophytes is dramatic as they have an important structuring and regulating role in the ecosystem: they stabilize the sediment (reduction of turbidity), compete for nutrients with phytoplankton; they increase the sedimentation rate and provide shelter from planktivorous predators for zooplankton species (Jeppesen *et al*., 1997).

The presence of the water hyacinth in Lake Naivasha has so far affected the productivity of the phytoplankton. Lower chlorophyll-a at the stations with the water hyacinth present indicates lower productivity by the phytoplankton in the water column. Dissolved oxygen was also reduced under the water hyacinth mats. This indicates reduced production by the phytoplankton and increased bacterial respiration from increased organic matter produced by the macrophytes (Voulion, 2004). The trend that Talling noted (1957), that chlorophyll concentrations are higher nearshore than offshore, is still evident in the water hyacinth free sites in Lake Naivasha. At all sites that were relatively deep there was a persistence of the hypolimnetic anoxia that Talling (1957) observed in his study. Dissolved oxygen at these deep stations averaged 1.79 mg O2•liter-1.

The reduced phytoplankton productivity in the presence of water hyacinth can be the result of a combination of factors, the most evident being the shade created by the floating mats (Rommens *et al*., 2003). McVea and Boyd (1975) found that water hyacinth mats hinders phytoplankton photosynthesis by shading out the algae. In addition, water hyacinth may be a better competitor than phytoplankton for limiting nutrients; in McVea and Boyd (1975), both phosphate and total phosphorus in the water column were quickly reduced once a small number of the floating plants established themselves, depriving the phytoplankton of this crucial nutrient and inhibiting their growth. Yang *et al*., (1992) found that water hyacinth can exhibit allelopathic effects on algae, biosynthesizing three kinds of algaecidal compounds in its roots and secreting them into the water to inhibit algal growth. They emphasize that the three compounds have a higher antialgal activity than the common algaecide $CuSO_4$ (Yang *et al*., 1992; Callaway & Aschehoug, 2000)). As a result of decreased phytoplankton production, McVea and Boyd (1975) observed a decline in the numbers of phytoplanktivorous fish in the small ponds that they studied. The effects of decreased algal productivity in Lake Naivasha could therefore extend all the way up to the piscivorous fish species at the top of the food web, and could pose a threat to the lake's fishery if the water hyacinth continues to thrive.

This study recommends that there is need for more research work to be done to understand why algal productivity decreases with water hyacinth cover in Lake Naivasha. This should be supported by other studies to measure total phosphorus and phosphate under the water hyacinth mats in order to confirm if the phytoplanktons are deprived of these nutrients by water hyacinth plants as stated in McVea and Boyd (1975). Finally, new surveys should be designed to do species counts to ascertain which kinds of phytoplankton persist in Lake Naivasha under the water hyacinth mats for similar studies elsewhere indicate that blue-green algae give way to Chlorophyta under water hyacinth cover (Yang *et al*., 1992).

Biological control of Water Hyacinth in lake Naivasha

Importation of additional biological control agents, the moth *Niphograpta albiguttalis*, the mite *Orthogalumna terebrantis* and the hemipteran bug *Eccritotarsus catarinensis*, to augment biological control efforts by *Neochetina* weevils, is recommended. Rearing pools, which are easier to manage and have a larger capacity, are preferred to be established near the lake. Releases on floating mats assisted in the redistribution and spread to non-release sites. Wind and water currents were responsible for the spread of weevils on floating mats of water hyacinth. Under the environmental conditions of Lake Naivasha, weevils established fairly slowly.

Ecological succession of water hyacinth by emergent plant species, mainly papyrus (*Cyperus papyrus*) and hippograss (*Vossia cuspidata*), has been noted. This phenomenon has also been observed in Lake Kyoga, Uganda, following the successful biological control of water hyacinth by Neochetina weevils (Julien *et al*., 1999). However, this is short-lived and the secondary vegetation will disappear after the degraded hyacinth substratum supporting it eventually sinks. The long-term approach to water hyacinth management should focus on curbing the discharge of effluents into Lake Naivasha from surrounding urban settlements, agricultural and industrial activities.

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References

Adams, C.S., R.R. Boar, D.S. Hubble, M.Gikungu D.M. Harper, P. Hickley & Tarras-wahlberg. (2002). The dynamics and ecology of exotic tropical plants in floating plant mats: Lake Naivasha, Kenya. *Hydrobiologia* 488, 115-122.

Gonzalez, S.H Hunt, M.E., and Koch, H.A. (2003). Physical attributes and eutrophication of Lake Atawapaskat, East Africa. *Journal of Atawapaskat Research* 1, 25-33

Hecky,R.E.,Bugenyi,F.W.B.,Ochumba,P.,Talling,J.F.,Muggidde,R.,Gophen,M., & Kaufmn,L. (1994). Deoxygenation of the deep water of Lake Victoria, East Africa. *Limnol. Oceanogr*. 39, 1476-1481.

K., and Lao,X.F.1992. Isolation and identification of antialgal compounds in root system of water hyacinth. *Acta. Phytophysiol. Sin.* 18: 399-402.

Lung'ayua, H.B.O, M'harzi, A., Tackx, M., Gichuki, J., & Symoens, J.J. (2000). Phytoplankton community structure and environment in the Kenyan waters of Lake Victoria. *Freshwater Biol*. 43, 529-543.

Masifiwa W.F., Twongo, T., & Denny, P. (2001). The impact of water hyacinth, *Eichhornia Crassipes*(Mart) Solms on the abundance and diversity of macroinverterbrates along the shores of northern Lake Victoria, Uganda. *Hyrdobiologia* 452, 79-88.

MCvea, C., & Boyd, C.E. (1975). Effects of water hyacinth cover on water chemistry, phytoplankton, and fish in ponds. *J. Environ. Qual.* 4, 375-378.

Ntiba, M.J., Kudoja, W.M., & Mukasa, C.T. (2001). Management issues in the Lake Victoria watershed*. Lake. Reserv. Manage*. 6, 211-216.

Scheffer,M.,Szabo,S.,GragnaniaA.,Vannes,E.H.,Rinaldi,S.,Kautsky,N.,Norberg,J.,Roijackers,R.M.M.,Fraknen,R.J. M. (2003). Floating Plant Dominance as a stable state. *P. Natl. Acad. Sci. USA.* 100, 4040-4045.

Talling, J.F. (1957). Some observations on the stratification of Lake Victoria. *Limnol. Oceanogr.* 2, 213-221.

Voulion, V.V. (2004). Contributions of major groups of autotrophic organisms to primary production of water bodies. *Water Resour. Res*. 31, 92-102.

Environmental implications of Water Hyacinth infestation in Lake Naivasha, Kenya

John Momanyi Mironga

Department of Geography, Faculty of Environment and Resource Development, Egerton University, PO Box 536, Njoro, Kenya
Email:mmironga@yahoo.com

Abstract

The proliferation of water hyacinth in Lake Naivasha during a period of about 16 years has imposed a variety of socio-economic and ecological impacts in the lake. This study examined whether water hyacinth floating mats were influencing the richness of flora and fauna in Lake Naivasha. The study further described the changes in the distribution of water hyacinth which have occurred in the lake since 1988, documented species of plant and invertebrate recorded on them and analysed aspects of the composition of different-sized floating islands of water hyacinth. Research results indicate that infestation of permanent mats of the weed along shores of Lake Naivasha had ecological disadvantages as well as some advantages. The shoreline fringe smothered the indigenous and floating macrophytes; and the water environment under the mat was largely devoid of oxygen except along the open water interface. Therefore, important fish breeding, feeding as well as fishing grounds were locked under hostile environment. However, diversity and abundance of macro-invertebrates were enhanced in the mats at the open water interface although both were poor the farther away into the water hyacinth mats. A marked increase in abundance of species of Bulinus known to be vectors of the agents of bilharzias was recorded in association with stationary fringe of the water hyacinth. An increase in abundance of some small fish species was realized. The progressive displacement of water hyacinth from the fringe by hippo grass through ecological succession created shore environments whose ecological advantages, disadvantages and sustainability require detailed and prolonged research to determine. Ecological impacts of the mobile mats were not systematically studied due to problems of access. However, it was proposed that edges of the weed mats were centers of high biodiversity and that mobility of the weed mats contributed to dispersal of the biodiversity. The major impacts due to mobile water hyacinth were socio-economic and they affected various sectors including the fishery, water transport, water abstraction and recreational activities. No quantitative evaluation was made. There is a great need to undertake research to quantify the levels of damage, and the costs of control, loss of livelihood, disease, and disruption of normal operations caused by water hyacinth.

Key words: Water hyacinth, Ecological Impacts, Invasive Species.

Introduction

Invasive species are of interest to ecologists, biological conservationists and natural resources managers due to their rapid spread, threat to biodiversity and damage to ecosystems. Invasive species may alter hydrology, nutrient accumulation and cycling (Polley *et al*., 1997). The global extent and rapid increase in invasive species is homogenising the world's flora and fauna (Mooney and Hobbs, 2000) and is recognized as a primary cause of global biodiversity loss (Czech and Krausman, 1997; Wilcove and Chen, 1998). Bio-invasion may be considered as a significant component of global change and one of the major causes of species extinction (Drake *et al*., 1989).

Guiding Principle 7.1, *IUCN Guidelines for the prevention of biodiversity loss due to biological invasion* (IUCN Council, 2000) encourage countries to act rapidly to eradicate or control new alien invasive species, even if there is scientific uncertainty about the long-term outcomes of the invasion. The best opportunities for eradicating or containing an invasive species are in the early stages, when populations are small and localized. These instructions for the approaches to alien species are applicable to most habitats and ecosystems but hard to implement in the fresh-water environment where invaders are often hidden from sight (beneath the water surface), or spread by water currents or able to disperse rapidly (IUCN Council, 2000). This is one of the reasons why fresh-water invasive species are often well-established by the time they are recognised and by the time a strategy is developed to control them. Nevertheless there have been several invasive species of fresh-water systems in tropical countries and in particular in Kenya since the late 80s which can give us ideas about how to tackle these problems in future.

The most deleterious of the invasive species and listed among the ten most notorious weeds in the world is the water hyacinth (*Eichhornia crassipes*) (Mart) (Pieterse, 1990a). Water hyacinth, *Eichhornia crassipes*, has become a nuisance for fisheries, navigation, water intake to hydropower plants, irrigation, and recreation in many tropical and subtropical lakes and facilitates the spread of such diseases as schistosomiasis and malaria (Mehra *et al*., 1999, Navarro & Phiri, 2000). It restricts photosynthesis through increased sedimentation and by shading the water column, leading to deoxygenation with a detrimental impact on aquatic organisms, especially fish. On the other hand, its capacity for accumulating heavy metals and organic contaminants, together with its wide tolerance to environmental conditions, make water hyacinth suitable for treating waste waters.

Its rapid growth and multiplication has led, among other things, to various applications as an animal food, paper and other products, or as compost (Mehra *et al*., 1999, Navarro and Phiri, 2000). This dual feature adds complexity to its management. In the past, chemical control with herbicides like 2,4-D and glyphosate was considered the most effective way of dealing with this weed problem. Biological control, primarily with water hyacinth weevils (*Curculionidae*), has had promising results in many places, although it has not been consistent at all sites (Centes *et al*., 1990). A combination of biological and chemical control in an early state of infestation and a well-coordinated flow of expertise and resources is likely to be the best long-term solution to the water hyacinth problem (Navarro and Phiri, 2000).

The water hyacinth first appeared in Lake Naivasha, in 1988. It subsequently spread throughout the entire lake but was particularly prevalent in northern shallow inshore waters. The present-day cover by water hyacinth has remained relatively stable. It usually forms a narrow fringe 5-15 meters wide around much of the lake. Its control at Lake Naivasha has focused upon biological control measures with the introduction of the Eichhornia weevil (*Neochetina spp.*) in the late 1990s. Because the enormous floating mats of the weed interfered with boat navigation, fishing, and even clogged up irrigation canals around the lake (Adams *et al*., 2002), much of the attention devoted to the water hyacinth in the literature has been concerned primarily with the methods for eradicating it from the lake. In contrast, fewer studies have focused directly on its ecological effects.

A workshop on the Control of Africa's Floating Water Weeds in 1991 stressed the importance of conducting ecological studies on aquatic ecosystems affected by water hyacinth in order to estimate its effect on biodiversity (Greathead and de Groot, 1993). To date, however, little attention has been paid to the potential importance of water hyacinth mats for maintaining the structural complexity of a lake and its species diversity. This lack of ecological research is probably due to the emphasis given to the many socio-economic problems caused by water hyacinth infestations with its eradication being given priority. This study attempted to evaluate the impact of water hyacinth in Lake Naivasha, a fairly eutrophic lake with some importance for tourism and fisheries, but which mainly provides water for a rapidly expanding horticultural industry around it. The impact of water hyacinth on the species diversity and livelihood of the communities around Lake Naivasha is the focus of this paper. Each time, the paper addresses the question whether the ecology of the lake changed in function of the presence of water hyacinth or not.

Study area

The Naivasha basin is situated in the Kenyan Rift Valley, about 80 km north west of Nairobi. The basin is located approximately between 0° 08' to 0° 55' S and 36° 00' to 36° 45' E. This part of the Rift Valley covers the three lakes of Nakuru, Elementeita and Naivasha to the south. The highest of the Rift Valley lakes, Naivasha (Figure 1) lies at about 1880 meter (6168 feet) above sea level, the lake level varies quite considerably - in 1926 it was reported to be 6 m higher. Lakes in the Rift Valley are normally saline unless water can escape through an outlet; however there is now no visible outlet to the Naivasha Lake. The supposition is that there is underground seepage maintaining the movement of fresh water brought into the lake by the Gilgil and Malewa rivers in the north.

The Lake consists of the main lake, a small separated lake Oloidien and a smaller Crater Lake Sonachi. The total catchment of the lake is approximately 3200 km2 (Morgan, 1998). The main lake (water surface) is approximately 120-150 km^2 plus 12 - 18 km^2 of swamp. LNROA (1996) reported that the lake has a mean depth of 4.7 m, with the deepest part at the Oloidien Bay (9 m) and around Crescent Island (17 m). In 1997 the mean depth of the lake was calculated at 3.8 m (Donia, 1998). Rupasingha (2002) have done the bathymetric survey during October 2001 and the result of calculated mean depth was 3.41 m at the level of 1886.38 masl.

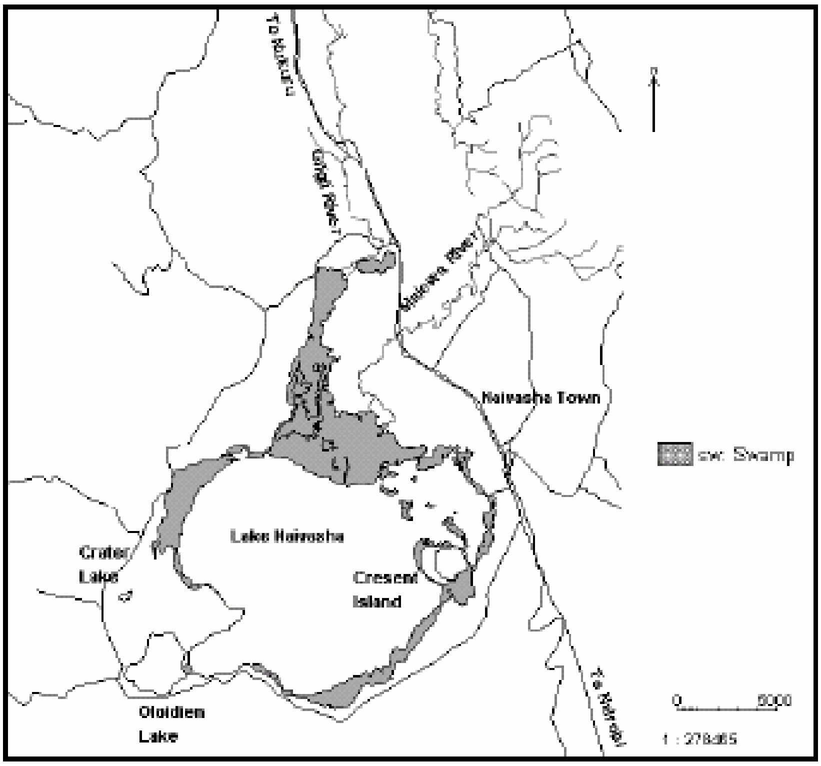

Figure 1. Lake Naivasha.

The ecology of Lake Naivasha is forever changing (Harper and Muchiri, 1986). The changes are brought about by alien invasive floating aquatic weed species (*S. molesta, E. crassipes* and to a limited extent, *P. stratiotes*) among other factors. The weeds have infested the lake ecosystem in the last three decades, suppressed and occupied ecological niches previously inhabited by native flora such as papyrus and water lilies, and thus disrupted plant-animal-physical environment interactions and balance. The formation and movement of their floating mats have influenced the whole lake and even led to re-distribution of seral stages - in that plant succession no longer follows a predictable sequence proposed by Guadet (1977) (Figure 2).

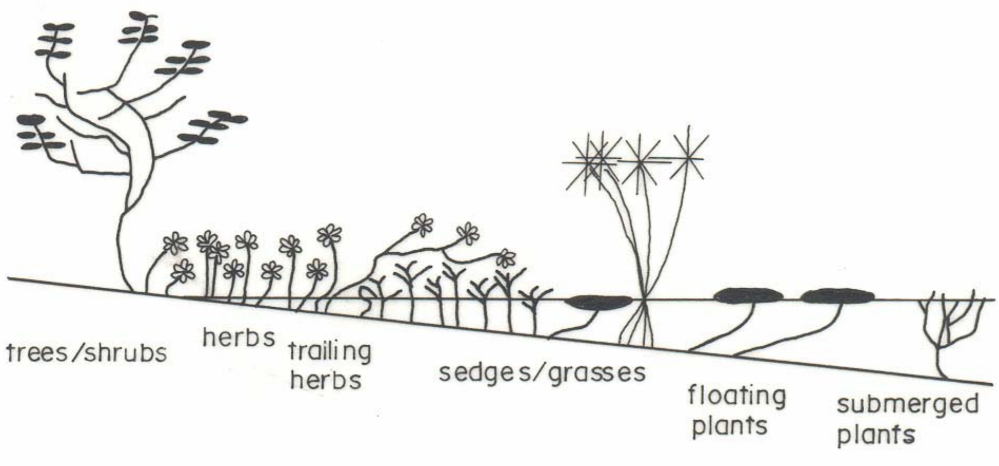

Figure 2. Hydroseral succession of plant species around the shore of Lake Naivasha in the 1970s. *After Gaudet (1977).*

These floating aquatic weeds (FAWs) are known to affect water resource management, the continued existence of human, riverine and wetland communities, and conservation of biodiversity. Waterways can be blocked; level of floodwaters increased substantially, water loss increased through evapotranspiration and the efficiency of irrigation and hydro generation impaired. People are affected by a reduction in the fish catch, difficulties in travelling by boat and consequent isolation from water sources, gardens, markets and health services, and also change in populations of vectors of human and animal diseases. Biodiversity can be reduced and conservation value affected. *E. crassipes, S. molesta* and *P. stratiotes* colonize open water at the margins of water bodies or occur as floating "islands".

Methods

Surveys were carried out in the months of April and May 2004, and involved lake inspections by boat and on land, interviews with local residents and stakeholders, literature and document review, and consultation with key persons. These surveys were intended to provide practical experiences on the impacts and management of water hyacinth in Lake Naivasha.

A questionnaire (Appendix 1) was used for the purpose of collating information from residents and stakeholders on the subject of the impacts of water hyacinth in Lake Naivasha basin. Additional surveys on the lake were made and the water hyacinth distribution status was assessed.

Results

Impact of Water Hyacinth in Lake Naivasha

The concern over water hyacinth

Water hyacinth infestation has resulted in serious socioeconomic and environmental problems for thousands of people living around Lake Naivasha. Over the years the weed has continued proliferating forming extensive floating mats that cause disruption in irrigation canals, navigation and fishing activities, and cause an increase in water loss through evapotranspiration. The weed also reportedly provides breeding grounds for schistosome (bilharzia)-carrying snails and malaria- carrying mosquitoes. The cost of water hyacinth infestation for Lake Naivasha is estimated to be of the order of millions of Kenya shillings. The rapid proliferation of water hyacinth in the lake Naivasha is a result of the widespread availability of nutrients in its waters. The nutrient enrichment of the lake is a result of pollution and other factors arising from the rapid increase of human population and corresponding activities in urban and rural areas around the Lake Naivasha. Large urban sewers and other effluent discharges are well known sources of point water pollution, while extensive use of improper agricultural methods, and land uses that often result in soil erosion, are a major source of non-point water pollution in this area.

The impacts of water hyacinth in Lake Naivasha are categorized in this study into social, economic and environmental. The information to be presented in this paper was collected through field measurements, interviews with officers-in-charge of district hospitals, fisheries, and water supplies in Naivasha municipality. Organizations, communities and activities affected by water hyacinth infestation were thus identified and are as shown in Table 1.

Table 1. Organisations, activities and communities affected by water hyacinth in Lake Naivasha.

Affected organizations/institutions/communities
Agricultural irrigation
Community development
Department of Fisheries
Electricity generating, KENGEN
Fisheries Department
Fishing community (fishermen, fishmongers, fish processors, consumers)
Health, Riparian communities
Kenya Agricultural Research Institute (KARI)
Kenya Marine Fisheries Research Institute (KEMFRI)
Lake transport
Local and provincial administration
National Environmental Management Authority
National Water and Sewerage Corporation
Navigation
Riparian communities
Various non-governmental organisations, e.g. Elsamere Foundation
Water supply

An indication relating to the perceptions of a cross-section of the communities and agencies in Lake Naivasha basin on impacts, as well as on control strategies of water hyacinth, was obtained through a survey, the results of which are recorded in Table 2. Lakeside communities deemed socioeconomic impacts more important than environmental impacts. The real costs and quantified impact levels were, however, not clear to those communities. Most of those interviewed identified decrease in fish catches, increase in certain diseases, increased difficulties in transportation, and difficulties associated with clean water availability as major negative impacts.

Table 2. Problems associated with water hyacinth in the Lake Naivasha Basin.

Category	Nature of problems
Social	Lack of clean water (debris-free)
	Less access to water points (domestic and livestock use)
	Societal conflict
	Increase in incidence of snake bite
	Disappearance of the aesthetic value of water
	Increase in disease outbreaks (schistosomiasis, cholera etc.)
	Reduction of riparian-based trade
	Migration of communities
	Percentage of overall responses
Economic	Reduced fish catches
	Increase in transportation costs
	Difficulties in electricity generation
	Difficulties in water extraction and purification
	Interference with irrigation (blockage of canals)
	Effects on tourism
	Effects of control on government budget
	Percentage of overall responses
Environmental	Decline in diversity and abundance of aquatic life
	Decline in water quality
	Increased water loss
	Increased siltation
	Increased potential for flooding

The impact of water hyacinth (Eichhornia crassipes) on water quality

This study evaluated the impact of water hyacinth on water quality in Lake Naivasha. The study examined the physical and chemical water quality and nutrient levels firstly along a transect from an inlet to the center of the lake and secondly following a random sampling design. Each time, the study addressed the question whether water quality changed in function of the presence or absence of water hyacinth or not.

Water quality of Lake Naivasha generally showed a dilution effect along a gradient over the lake from inflow (Malewa river) to the center part of the lake. Oxygen concentration and pH increased in downstream direction, while conductivity, ammonium, nitrite, nitrate and phosphate decreased. The reduction in ammonium along the transect was stronger than the reduction of nitrate. The polluting influence of the Malewa River caused an increase in ammonium and a decrease in oxygen concentrations. Algal biomass (mg chl-a/l) in the areas with water hyacinth was higher in the riverine part than in the lacustrine part of the lake. Superimposed on this general gradient in water quality from an inlet to center, most variables also differed in function of presence or absence of water hyacinth at each site along the gradient. Along the transect, the average ammonium and phosphate concentrations as well as transparency of the water were significantly (paired t-test, $p < 0.05$) lower in the areas with water hyacinth than in the corresponding areas without water hyacinth.

Differences in water quality between areas covered with hyacinth and other sites are further demonstrated when randomly selected sampling sites were compared (Table 3, one-way ANOVA). Ammonium, nitrate and nitrite concentrations were lower in the littoral zone especially in parts covered with water hyacinth. This difference was significant for ammonium and nitrate ($p < 0.05$). Total nitrogen, including organic compounds, however, was highest but not significantly different in the littoral vegetated parts of the lake. Chemical oxygen demand and chlorophyll-a were significantly higher under water hyacinth while oxygen levels were significantly lower than at other sites ($p < 0.05$). Transparency was significantly higher at sites without water hyacinth. There was over-saturation of oxygen only at the sites without water hyacinth. There were no significant differences among sites for the other variables.

Table 3. Differences in water quality (average value of 5 sites ± sd) among randomly selected open water (limnetic) and littoral sampling sites with and without water hyacinth in Lake Naivasha. P-level indicates significance of one-way ANOVA.

Water quality variables	Limnetic sites	Littoral sites with water hyacinth	Littoral sites without water hyacinth	P-level
Ammonium (mg N/L)	0.140±0.13	0.018±0.021	0.032±0.051	0.048
Nitrite (mg N/L)	0.017±0.009	0.011±0.005	0.011±0.006	0.362
Nitrate (mg N/L)	0.74±0.11	0.54±0.09	0.64±0.09	0.024
Total nitrogen (mg N/L)	1.38±0.35	3.52±1.98	2.10±1.76	0.124
Phosphate (mg P/L)	0.91±0.13	1.13±0.77	1.22±0.59	0.676
COD (mg/L)	59.4±23.5	180.4±116.4	56.6±55.5	0.037
Secchi depth (cm)	135.6±30.8	2.0±0.9	143.5±94.9	0.007
pH	8.91±0.29	8.37±0.58	9.08±0.41	0.076
Oxygen saturation (%)	149.6±27.0	92.0±40.5	153.3±52.9	0.067
Oxygen (mg/L)	12.0±1.6	7.4±3.2	12.2±3.7	0.040
Conductivity (µS/cm)	84.2±78.1	84.8±78.3	49.4±0.4	0.681
Temperature (°C)	17.6±0.9	17.9±1.1	18.9±2.0	0.372
Suspended solids (mg/L)	24.0±17.2	24.6±6.9	16.2±6.5	0.447
Chlorophyll-a (µg/L)	28.4±11.1	468.3±338.0	83.7±134.8	0.008

The impact of water Hyacinth on species diversity of lake Naivasha

This study also evaluated the role of water hyacinth in maintaining diversity in Lake Naivasha. In addition to species richness and diversity, the importance of water hyacinth mats as preferred sites for fish developmental stages was investigated while the associated macro-invertebrate community was inventoried.

Macrophyte diversity

It was estimated that 3.2% of the lake were covered with floating macrophytes, with the highest cover in sheltered parts of the lake where plants accumulated due to wind drift. Water hyacinth dominated the helophyte community (Appendix 2). Submerged macrophytes were rare and restricted to shallow zones of the lake. Fifty-one species of plants were recorded on/ or in floating mats of water hyacinth (Appendix 2), including submerged plants. Many water hyacinth plants were found apportioned between mat-forming species, grass and sedge species, herbs and shrub species. Cyperus pectinatus was the most frequent grass or sedge, and appeared to have an important role in binding the water hyacinth mats, facilitating colonization by other species. Present-day water hyacinth cover in Lake Naivasha has remained relatively stable. It usually forms a narrow fringe 5-30 m wide around much of the lake. Large mats occur in most sheltered bays and inlets.

There was an increase in Recognizable Taxonomic Units (RTU) number with increasing water hyacinth island size. The upper limit for plants richness was reached before that for invertebrates, as the richness of the latter continued to increase with island size (Figure 3).

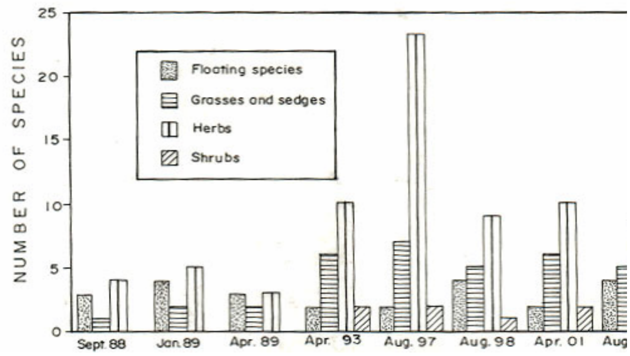

Figure 3. The species richness of groups of plants found growing on floating mats of water hyacinth in Lake Naivasha over the period 1988-2003

Macroinvertebrate diversity

The fauna associated with water hyacinth mats were concentrated in the layers between the leaves and the aquatic roots of the plant (Figure 4). The major groups of invertebrates found in this layer were *oligochaeta* (mainly *Alma emini*), insecta and Arachnida (both aquatic and terrestrial) (Table 4). Within this zone dead species of plants and invertebrate were found to be associated with water hyacinth.

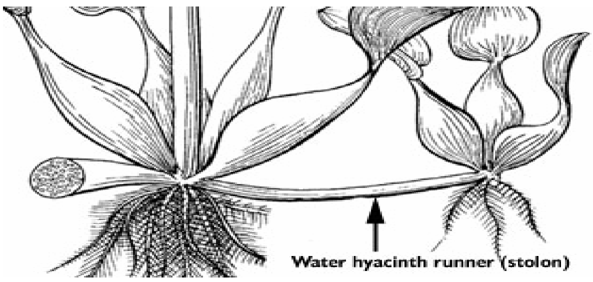

Figure 4. Morphology of water hyacinth plant that acts as a habitat of invertebrates.

Table 4. Taxonomic list of invertebrates recorded on water hyacinth at Lake Naivasha.

- Turbellaria
  - *Dugesia* sp.
- Oligochaetaea
  - *Alma emeni*
  - *Brachiura sowerbyi*
  - *Potamothrix* sp.
- Mollusca
  - *Bulinus* sp.
  - *Physa acuta*
- Aranaea
  - Hydracarina
  - Arachnida (terrestrial)
- Crustacea
  - *Procambarus clarkii*
  - Ostrocoda
  - Copepoda
- Insecta (terrestrial)
  - Collembolan
  - Thysanoptera
  - Orthoptera
  - Dermaptera
  - Cicadoidea
  - Staphylinidae
  - Formicidae
- Insecta (aquatic)
  - Coleoptera
    - *Hydaticus* sp.
    - *Rhantus* sp.
    - *Cybister* sp.
    - *Helochares* sp.
    - *Berosus* sp.
    - *Eochrus* sp.
    - *Canthyrus* sp.
    - *Hydrovantus* sp.
    - *Methles* sp.
    - *Synchortus* sp.
    - *Bidessus* sp.
- Hemiptera
  - *Micronecta* sp.
  - *Plea* sp.
  - Mesovoliidae
  - Lygaidae
- Diptera
  - Culicidae
  - Ceratopongidae
  - Chironomidae
  - Tipulidae
- Odonata
  - *Renellagma* sp.
- Ephemeroptera
  - *Cioeon* sp.
- Trichoptera
  - *Ecnomus* sp.

Further it was observed that plant material was continuously being broken down and wind-blown dust accreted in a soil-forming process, in which the earthworms *Alma emini* appears to be very important. The weevil *Neochetina eichhorniae*, which lives specifically on water hyacinth and is used as a biological control agent in the lake, was collected in six of ten samples in the water hyacinth mats.

Invertebrate density decreased with island size, in a power relationship, with density more or less constant for islands above approximately $10 m^2$ in area. No similar relationship was evident for plants. As plant species richness increased, so did invertebrate richness.

Fish Diversity

Using gangs of gill nets (Earth Watch gill nets) comprising of mesh sizes of between 8 and 50 mm, the fish of the lake was sampled at limnetic (P) and littoral zones with (L+) and without (L–) water hyacinth. The fishing method yielded 5 different fish species of which the principal species sampled at all sites was *Tilapia zilli* (Table 5). In terms of numbers *Oreochromis leucosticus* dominated the fish community representing 67.6% of the total catch, followed by *Micropterus salmoides* (20.7%) and *Tilapia zilli* (5.2 %). All other species represented less than 6.5% each. The dominant species in terms of weight were *Oreochromis leucosticus* (60.5 %), *Tilapia zilli* (22.7%) and *Micropterus salmoides* (16.8 %). The mean sizes (fork length) of these fish were: *Oreochromis leucosticus* 192 (S.D = 23 mm), *Tilapia zilli* (S.D = 18 mm), *Micropterus salmoides* (S.D = 17 mm).

Table 5. Relative species composition (% of the total catch over all the sampled areas) of the fish assemblage at each habitat of Lake Naivasha.

Species	Littoral zones with water hyacinth (L+)	Zones without water hyacinth (L–)	Limnetic zones (P)
Oreochromis leucosticus	4.5	56.8	70.1
Tilapia zilli	44.5	19.0	3.0
Micropterus salmoides	26.5	7.0	21.8
Procambrus clarkii	30.0	21.0	5.0
Barbus amphigramma	24.5	24.2	1.1

The distribution of the fish species in the lake like *Oreochromis leucosticus, Micropterus salmoides, Tilapia zilli* and *Barbus amphigramma* was more patchy, with their distribution reliant on the habitat sampled. *Oreochromis leucosticus* was found mainly in limnetic sites. *Micropterus salmoides* and *Barbus amphigramma* were found to be common in the littoral sites that were rocky margins.

Species diversity within the fish assemblage was, in general, higher in areas with water hyacinth than in those without. In addition, littoral habitats had more diverse assemblages than the pelagic part of the lake. Differences amongst diversity indices were, however, statistically not significant. In general, all the fish sampled revealed that smaller individuals preferred the littoral rather than the limnetic waters of the lake. This was illustrated by the crayfish, *Procambus clarkii*, whose average carapace size at the littoral zone was 17 mm and in the open water 45 mm. This reveals that the juvenile crayfish preferred the refuge of the littoral vegetation with gradual dissociation from these vegetated areas with increases in carapace size.

Socioeconomic Impacts of Water Hyacinth

Fishing Industry

Forty-five fishermen were interviewed, from various sites around the lake. Forty-two of the interviewed were male despite deliberate efforts made to interview more female respondents (Appendix 3). Although in most developing countries full time specialization in one field may be of great economic risk, 95 per cent of those interviewed were full time fishermen. This, therefore, implies that the water hyacinth has adversely affected their livelihood with no alternative.

Most of the respondents claimed to have first seen the weed on Lake Naivasha around early 1980s and very early 1990s (Appendix 4). This seems to concur with documented evidence that the weed was first reported in Lake Naivasha in May 1988. The responses on how the weed has affected fishing and other activities were much in line with earlier findings. These include: failure of boats to sail through waters covered by the weed given the thick mats created by the meshed roots, which lock up the boats and fishermen sometimes for hours. This leads to economic losses due to increase in catch delivery time. These delays have at times resulted in deterioration of the quality thus a reduction in the prices, and at most complete spoilage of the fish rendering it unsafe for human consumption. Fishermen have had to carry ice to maintain the quality. Higher costs to operators have also resulted from use of more fuel. Maintenance costs of engines have also increased due to knocks from the weed sucked into the engines.

The weed has led the fishermen to economic losses in terms of decreased price resulting from reduction in quality as evidenced by 32 (71.1%) of the responses. The weed traps them in the water where they spend a longer period than expected. The few who argued there had been an increased fish price (17.8%) based their argument on decreased numbers of fishermen who get access to the fishing sites. Another view is that there has been a reduction in reproduction because of the weed. The overall effect of the two is reduced supply of the fish vis-à-vis increased domestic and export demands. These declines may have been associated with the inability of fishermen to access the fishing grounds for those species because of water hyacinth infestation.

Generally therefore, as a result of water hyacinth infestation, accessibility to land and water has been hindered, resulting in reduced fish catches, especially of tilapia and mudfish which are found mainly along the shores. Fisherfolk, however, reported increased fish catches from suitable breeding grounds provided by water hyacinth e.g. tilapia, *synodontis, protopterus* and *labeo*. There is, however, need to clarify this conflicting information; in many more areas around the lake. A reduced fish catch would have an adverse effect on the quality of life of the communities around the lake and consequently affect sustainable development in the area.

Transport

Water hyacinth has affected transport in Lake Naivasha in many ways. For instance the delivery of fish from the open lake to the land has been disrupted often lowering the quality of fish. Water hyacinth infestation has caused delayed fish deliveries leading to loss of quality, lower prices, and operating costs especially on those fishermen relying on motorized boats. During delayed landings the fishermen realizes lower prices as the crayfish consumes netted fish (30 per cent of the fish catch is damaged by the crayfish). Crayfish also causes damage to the nets. Most piers are not accessible during the afternoons when dense mats of water hyacinth are blown by strong winds to these areas.

Movement of people, especially the research officers and other stakeholders, between the lake and the mainland has been disrupted. The Fisheries Office is worst hit because they are not able to navigate the entire lake to control illegal fishing. Thus there are escalating cases of illegal fishing in the lake, a situation that might disrupt the already delicate fishing industry in the lake.

The lake offers recreational facilities among them boating, water-skiing, sport fishing and bird viewing. All these have been affected seriously because water hyacinth mats moves into the navigation routes, thus increasing the operating costs and occasionally endangers lives.

Water supply

The town of Naivasha and riparian communities all depend directly or from wells and/ or boreholes adjacent to the lake. Water supply to both riparian communities and municipality in Lake Naivasha basin is affected by water hyacinth. In municipality water hyacinth interferes with the water intake points through blockage, which lowers the quantity of water pumped. In Naivasha town, the municipality reports that the quantity of water supplied has dropped from 10,000 m^3 to 7,000 m^3 per day. Water hyacinth infestations have been reported to lower the water quality in Lake Naivasha (in terms of colour, pH, turbidity (suspended solids) of water), and hence increase the treatment costs. Increased costs are associated with keeping the water intake points free of water hyacinth. For example, Naivasha Municipality in the year 2000 employed 12 casuals per day, 6 drivers and 6 boat operators, at a cost of 12,000 Kenya shillings per day. The villages bordering the lake have no access to the lake to draw water at times when the shores are heavily infested with water hyacinth. Even if they get access to the water, it is dirty and often smelly because of the rotting mats of the weed. This is true for all the points along Lake Naivasha which may be infested.

The weed in several ways has impacted on the urban and the riparian community of Lake Naivasha in several ways as outlined below:

1. Clogging of water supply intake has already been experienced. According to the records existing in the Ministry of Water, Naivasha Town, the weed has choked the pipes that feed Naivasha town more than ten times, in most cases during the rainy season.

2. Discoloration of the water from the lake is common. This is due to suspended decaying organic matters from the weed. This situation continues posing extra problems to the department in charge of water processing in Naivasha area. The department is ill equipped to deal with this situation and therefore many people are continuously getting affected.

3. Because after mats of the weed are blown away by the wind, muddy and dirty water filled with suspended, decaying, smelly organic matter renders the water unsuitable for domestic use. This situation has greatly affected those using the lake water directly. The undesirable taste and odour of water is also as a result of the many mollusks found associated with water hyacinth in Lake Naivasha. According to local residents many small pipes used in pumping water from the lake have been choked in several occasions by these mollusks.

The impact of water hyacinth on water supply in Lake Naivasha is comparable with other water systems in the world as reported by the works of Krishnamoorthi and Rajgopalan (1970); Richard (1999).

Health

Water hyacinth has affected the health of the lake side communities and that of the fishermen. In Lake Naivasha ecosystem water hyacinth mats have provided the habitat for agents of malaria and bilharzia and harbors snakes. This study found that presence of water hyacinth in Lake Naivasha has enhanced transmission of amoebae, dysentery, typhoid and also causes severe skin rashes. Analysis of information from 10 respondents and records existing in Health Centers around the lake showed that 90% of respondents were aware that presence of water hyacinth had contributed to increased number of mosquitoes and attributed this to increasing cases of malaria amongst the riparian communities. 40% of the respondents had suffered from acute skin rashes as a result of manually removing water hyacinth from the landing bays. Whereas a small percentage 2% reported to have suffered snake bites during fishing and /or clearing of irrigation canals from clogged by water hyacinth.

In the health centers, by end of the year 2003 10% of those treated for amoebae and dysentery were from the fishing community and such cases were prevalent in the late 90s, a period when water hyacinth was spreading on the lake. 6% of those treated from bilharzia in the whole Naivasha District came from the fishing community. The records and interviews conducted on the Ministry of Health officials indicate that such cases are becoming common on those living near the lake and more so the fishermen. There numbers of typhoid cases fell over the same periods. The statistics on health should be interpreted with great care as these apply only to incidences reported to the hospitals or health centres in the area. There may have been many more cases of disease that were not reported to hospitals.

These findings are supported by a survey done in the lake by Harper (1992) which found that there are 28 species of *Hemiptera* and *coleoptera*, as invertebrates found associated with water hyacinth (which are disease causing organisms). Elsewhere, research shows that water hyacinth concentrates *Vibrio cholerae*, the organism responsible for cholera.

Agriculture

The water hyacinth has interfered with irrigation process in the farms around the lake. It was observed that water pipes that get the water out of the lake into the farms around the lake regularly get choked by water hyacinth mats. This is common during the rainy season, when the weed spreads all over the shore of the lake. 70.9% of the farmers interviewed reported to have suffered from such blockages quite often. At the same time the water hyacinth grows in the irrigation canals, reducing water movement. Up to 50% of the farmers around the lake who rely on this method to get the water into their farms had their channels blocked by water hyacinth. The channels get blocked more frequently during the rainy seasons.

It was established that water hyacinth plants carry pathogens, which infect fruits and flowers grown around the lake. However, it was not established which type pathogens are causing more damage and whether actually they came from the water hyacinth mats. The study established that the beetles used in the control of water hyacinth are breaking loose into the farms around the lake, thus causing a lot of damage. 75% of the respondents have experienced increased costs through purchasing of pesticides meant to contain this problem.

The farmers are experiencing increased costs of managing their farms due to extra labour needed in removing water hyacinth mats from pumping stations and unblocking the pipes and canals. Buying of pesticides to contain some of the insects coming from water hyacinth mats to attack farm produce is an extra cost.

Discussion

The non-indigenous water hyacinth (*Eichhornia crassipes)* typically exploits eutrophic water bodies such as Lake Naivasha. The water hyacinth mats currently covered about 3.2% (or 83 ha) of the total lake surface, which is about ten times less coverage than in the period of peak infestation between 1998 and 1999. Despite the prevailing dry and rainy season conditions, its presence significantly affected the physical and chemical characteristics of the water in the vegetated areas of the lake.

Physical impact of water hyacinth on water quality

In Lake Naivasha, the physical presence of water hyacinth mats at any site acted as a trap for detritus which is likely to cause a cascade of effects on the water quality. Obviously, the high density of detritus caused a significant decrease in water transparency, relatively to open littoral or limnetic sites. Due to shading and the low transparency, light levels underneath the beds of water hyacinth are low, presumably resulting in net respiration of phytoplankton and in the stimulation of bacterial decay processes. The accumulation of organic material and increased bacterial activity resulted in the high total nitrogen load, in contrast with the inorganic nitrogen that decreased significantly at sampling sites covered by water hyacinth. The enhanced sedimentation and decay processes also explain the higher COD and conductivity and lower oxygen levels under water hyacinth than in the open water.

Nitrogen is the primary growth limiting factor in Lake Naivasha as illustrated elsewhere by Thornton (1982). This is also reflected in this study by a decrease in inorganic nitrogen compounds underneath mats of hyacinth and a decrease in nitrogen content in leaves of water hyacinth along a gradient from an inlet River Malewa to the center part of the lake. In the littoral zones of the lake, where the ammonium levels were lower (difference between L+ and L–: 0.01mg N/L versus 0.1mg N/L for nitrate, (Table 4) nitrate probably becomes the more important nitrogen source.

In comparison with early records of the water quality from rainy season samples in Lake Naivasha presented by Harper (1988) some marked differences with this study's measurements were noted: pH values in during this survey (averages ranging between 8.4 and 9) were much higher than in the earlier records (pH 7.1–7.2).

Also nutrient levels increased drastically: NO^3-N values have increased 10 to 30-fold while PO_4-P values were about 200 – 250 times higher in this study. Only the ammonium levels in the littoral zone were similar to the earlier records, but the values in the limnetic zone were about 10 times higher. These drastic changes relates to the ongoing eutrophication process already occurring in the lake as detected by Kitaka (2002).

The present low whole-lake impact of water hyacinth on the nutrient levels is due to the small cover (3.2% of the lake surface), especially when compared with the situation in the period 1988 –2002, when water hyacinth covered up to 5% of the lake surface and reduced nutrient levels considerably. After destruction of these fairly vast water hyacinth mats by the control methods in place, phosphorus and nitrogen might have increased between 2000-2004, not only due to the reduced uptake by the decreasing hyacinth biomass but also due to the decay of dead plant material. Similar observations were made at Hartbeespoort dam in South Africa where chemical destruction of water hyacinth resulted in aggravation of eutrophication problems and the development of a blue-green algal bloom (Davies and Day, 1998).

Uptake by the plants can only modify concentrations of nutrients in the water for some time if their biomass increases, or if the plants are exported. The present water hyacinth mats at Lake Naivasha, which are under biological and small scale mechanical harvesting, is unable to substantially reduce the nutrient levels of the lake. Further nutrient reduction can, however, possibly be achieved by the cultivation of hyacinth at the mouth region of the rivers like River Malewa discharging into the lake. This approach requires modeling of the best trade-off between rapid growth and fast removal or harvesting of water hyacinth. As Lake Naivasha also has a history of pollution from the flower industry around it (Kitaka, 2000), water hyacinth may also control these contaminants as was shown by Shine *et al.* (1998). At present, mechanical harvesting is only done on a small scale by fishermen with support of the LNRA along the shoreline (fish landing piers).

Impact of water hyacinth on biodiversity

Further to the foregoing, this study agrees with the findings of Drake and Mooney (1989) and Luken and Thieret (1997) that despite the onslaught of non-indigenous species worldwide, it is often difficult to determine what the congruent ecological effects are of such invasions. Oftentimes, sufficient monitoring is not available to document changes caused by a specific invading organism. As the study findings shows, water hyacinth can be characterized by distinctly different invertebrates. Therefore, presence of water hyacinth in Lake Naivasha is associated with minor to major shifts in invertebrate assemblages depending on the site, and can easily alter the 'lakes fish invertebrates' food web. Such community level effects are typical of habitat-altering invaders like the case of water hyacinth (Bertness, 1984) Posey, 1988; Vitousekm, 1990; Richardson *et al*., 1997; Woods, 1997; Crooks, 1998 and Crooks and Ichim, 1999) as hyacinth is not only widely abundant, but also provides structurally complex substrate to other organisms in both the aquatic and terrestrial zones of lake Naivasha.

Contrasting with the diversity at lower trophic levels, macro-invertebrates and fish apparently benefit from the presence of water hyacinth. When compared to the open water, the root and leaf structure of water hyacinth provides a complex habitat for these species. Mitchell and Marshall (1974) found a wide variety of species in mats of *Salvinia molesta* in lake Kariba, demonstrating the importance of floating weed mats for macroinvertebrate diversity. Olson *et al*., (1994) compared diversity and abundance of macro-invertebrates from three different macrophyte communities and an open water site in a Minnesota prairie marsh and found the largest numbers of organisms (mainly chironomid larvae) but the lowest diversity in the open water while the highest diversity occurred in the Typha sites. The high relatively diversity and population sizes of snails in this study could be related to the lake's eutrophic state (Kitaka, 2002). With their number and diversity of organisms, the areas with water hyacinth may have an important function as feeding places for birds and some fishes. In several other studies an association was shown between macro-invertebrate diversity and waterfowl use and productivity (Svingen and Anderson, 1998). Fish species diversity was, in general, higher at the areas with water hyacinth than in the open water and was higher in the littoral zone when compared to the limnetic areas. By means of gill netting, it was shown that the *Oreochromis leucosticus* which mainly feeds on benthic insects (had a strong preference for the areas with water hyacinth while the barb species *Barbus ampigrama* were only caught near rocky areas without hyacinths. *Tilapia zilli*, in turn, typically occurred in the pelagic zone. Apparent differences in habitat preference could also be due to diurnal movements into and away from warm shallow waters as is characteristic for many cichlid fishes and which, besides avoiding predators, could also be a means of improving physiological efficiency (Marshall, 1982). The most important function of water hyacinth for fish appears to be the provision of refugia and favourable feeding conditions. Smaller individuals of *Oreochromis niloticus, Tillapia zilii* and *Procambarus clarki* preferred the littoral zone rather than the open waters of the pelagic area. Small fishes and juveniles, especially cichlids, need to escape predation; they shoal in shallow 'nursery' areas (Marshall, 1982). As they become larger, their vulnerability decreases and they move into deeper water.

In conclusion, water hyacinth mats did not clearly support a higher diversity of aquatic organisms than the areas with water hyacinth in Lake Naivasha. Water hyacinth mats are evidently important for various macro-invertebrates that live on plant leaves (e.g. snails). In fish there was only a trend towards a higher diversity in the water hyacinth-covered zones. The most important function of the hyacinth mats seems to be a sheltering or nursery function for small size classes of fish. Such a function is, however, not only performed by water hyacinth and could also be so by other macrophytes that are currently present in Lake Naivasha.

The habitat-altering characteristic of water hyacinth may also affect restoring wetlands in the lake area, as canopies of hyacinth form on the marsh fringe and may be influential in the development of adjacent marsh community. There is vast interest in Lake Naivasha on predicting the rates and patterns of restoration of fringe wetlands, as exemplified by the Earthwatch Research Programme (Adams *et al.,* 2002). The case of succession in Lake Naivasha follows a pathway of submerged aquatic vegetation, emergent marsh vegetation as illustrated by Penfound and Earle (1948). This pathway of succession is also supported in other warm climates where hyacinth grows throughout the year as permanently floating islands are created which deposit large amount of organic matter (Trivedy *et al.,* 1978, Gopal 1987; Woods, 1997). Although this pathway is abbreviated in Lake Naivasha due to low temperatures that inhabit continuous growth of water hyacinth canopies, rates of wetland restoration could also be accelerated due to increased deposition of organic material. Growth of grass-species *Cyperus pectinatus* on canopies of water hyacinth could also help stabilize the canopies allowing emergent marsh vegetation to obtain a foothold. Such pathways warrant more research in Lake Naivasha as the current study focused primarily on social-ecological issues of the water hyacinth.

Conclusion

Establishment of water hyacinth in Lake Naivasha has had a broad effect on the utility and ecology of the lake. So far, the weed has adversely affected movement on the lake leading to increased operating costs for motorized boats, delayed fish deliveries and loss of quality.

Water level fluctuations, nutrients, water temperatures and pH are the driving forces behind water hyacinth spread in Lake Naivasha. Doubling time for water hyacinth is 14 days during high water levels compared to 40 days during low water levels in the lake. Relative growth rates follow the same trend. At the same time water temperature regime of Lake Naivasha ecosystem has contributed to a surprising low rate of spread of water hyacinth compared to other water bodies in the world. PH on the other hand as curtailed growth and spread of water hyacinth in certain sections of Lake Naivasha, especially in Lake Oloidien. As these factors and their combinations continue enhancing the growth and spread of the water hyacinth on the Lake Naivasha, the utility and the general ecology of the lake is getting undermined in several ways as discussed in this paper.

The spread of water hyacinth in the Lake has affected the irrigation of horticultural crops around the lake through blocking of the irrigation canals and water pipes. These horticultural farms produce fruits, vegetables and flowers. The horticulture industry around the lake is a source of employment to more than 30,000 people. At the same time, the dairy industry that is well established around the lake is completely dependent on irrigated lucerne farms and therefore is equally affected by the presence of water hyacinth in Lake Naivasha.

Lake Naivasha's outstanding aesthetic scenery and recreational potential has also been undermined by the presence of the water hyacinth. The Lake is well known for boating, water-skiing, sport fishing, game viewing and bird watching. Some 350 bird species have been recorded in and around Lake Naivasha (Bennum, 2002). Because of its outstanding scenery, there are many tourist hotels, campsites, hostels and marinas for accommodation. In 1994, a total of 41,000 tourists visited Lake Naivasha (LNRA, 1995) to witness its beautiful scenery. But in 1999 only 20,000 tourists visited the lake, recording a sharp decline. As the water hyacinth continues establishing itself in the lake it is therefore posing a serious threat to the tourism industry. Consequently, many people in the tourism industry have lost their jobs and many businesses brought to a stand still thus denying the country the much-needed foreign exchange.

The people living around the lake and the town of Naivasha all depend on the lake for their water supply. Presence of water hyacinth in Lake Naivasha has affected this water supply by blocking the pipes and lowering the water quality. Ol Karia Geothermal Station under the management of

Kenya Generating Company (KENGEN) use large volumes of water from the lake in drilling new steam wells and in condensing steam in the existing 65 MW geothermal power plant. The spread of water hyacinth in the lake may disrupt the ongoing second geothermal power project hoped to yield an additional 20 MW. This will pose a significant threat to power production in Kenya and even hamper the plans of exploiting geothermal resources in the country in the next 20 years.

This current situation calls for the lake to be accorded a proper scientific attention on the status of the weed. It is clear from the findings discussed in this paper that the continued existence of water hyacinth infestation in Lake Naivasha has far-reaching negative impacts on the people relying on the well being of the lake. The ecological functioning of the lake would undergo dramatic changes in future if at all the spread of water hyacinth on the lake is not contained. Therefore, every effort should be made to contain the rapid spread of this weed in an endeavor to restore the position of Lake Naivasha as a lifeline to the large number of people in the Rift Valley and as an important economic resource in terms of tourism and irrigation.

While this study presents important information on the infestation of Lake Naivasha by water hyacinth several weaknesses were identified during literature review related to access of information on water hyacinth, implementation of control mechanisms, and lack of research on certain aspects as the gaps that explain why Kenya is not promptly taking advantage of existing knowledge and capabilities to control water hyacinth whenever it invades its water systems. These gaps call for action to effectively mobilize, organize, and equip decision-makers, researchers, and representatives of communities, along with their organizations and institutions, to improve their response to the water-hyacinth problem.

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References

Adams, C.S. R.R. Boar, D.S., Hubble, M. Gikungu, D.M. Harper, H. Hickley & N. Tarras-Wahlberg, (2002). The dynamics and ecology of exotic tropical species in floating plant mats: Lake Naivasha, Kenya, *Hydrobiologia* 488 (Development Hydrobiologia 168), 115-122.

Centes, T. D., Confrancesco, A. F. & Balciunas, J. K. (1990). Biological control of aquatic and wetland weeds in the south-eastern United States. – In: Delfosse, E. S. (ed.): Proceedings of the VIIth International Symposium on Biological Control of Weeds, 6–11 March 1988, Rome, Italy, pp. 239 –262.

Crooks, J. A. & H. S. Khim. (1999). Architectural vs. biological effects of a habitat-altering, exotic mussel, *Musculista senhousia*. *Journal of Experimental Marine Biology and Ecology,* 240, 53– 75.

Czech, B., & Krausman, P. R., (1997). Distribution and causation of species endangerment in the United States. *Science*, 277:116-117.

Greathead, A. & DeGroot, P. (1993). Control of Africa's Floating Water Weeds. – Proceedings of a Workshop held in Zimbabwe, June 1991. Commonwealth Science Council, CAB International and Biomass Users Network, CSC Agriculture Program Series Number CSC (93) AGR-18, Proceedings 295, 186 pp.

Harper, D., Adams, C. and Mavuti, K. 1992. The aquatic plants of the Lake Naivasha wetland, Kenya: Pattern dynamics and conservation, 4th International Wetlands Confe. Ohio 1992.

Kitaka, N., Harper, D.M. & Mavuti, K.M. (2002). Phosphorus inputs to Lake Naivasha, Kenya, from its catchment and the trophic state of the lake, *Hydrobiologia* 488 (Developments in Hydrobiology 168), 73-80.

Krishnamoorthi, K.P., & Rajagopalan, S. 1970. Survey of mollusks nuisance in some water supplies of Calcutta. Proc. Symposium on Mollusca, Pt.III, 746-754.

Mehra, A., Farago, M. E., Banerjee, D. K. & Cordes, K. B. (1999). The water hyacinth: An environmental friend or pest? A review. *Resour. Environ. Biotechnol*. 2, 225 – 281.

Mitchell, D.S. (1976). The growth and management of *Eichhornia crassipes* and *Salvinia spp.* in their native environment and in alien situations. Pages 167-175 in C.K. Varshney and J. Rzoska, editors. Aquatic weeds in Southeast Asia. Dr. W. Junk Publisher. The Hague, Netherlands. 396 pp.

Mooney, H. A., & Hobbs, R. J., (2000). Invasive Species in a Changing World. Island Press, Washington, DC.

Navarro, L. & Phiri, G. (2000). Water hyacinth in Africa and the Middle East. A survey of problems and solutions. – International Development Research Centre, Canada, 120 pp.

Penfound, W.T. & Earle, T.T. (1948). The Biology of water hyacinth, *Ecol. Monogr*., 1, 92-101.

Pieterse, A.H. (1990a). Introduction of aquatic weeds. In: Aquatic A.H. Piertese and K.J.

Polley, H. W., Johnson, H. B., & Mayeux, H. S. (1997). Leaf physiology, production, water use, and nitrogen dynamics of the grassland invader Acacia smallii at elevated CO_2

Richards, J. H. (1982). Developmental potential of axillary buds of water hyacinth, *Eichhornia crassipes* Solms. (Pontederiaceae). *American Journal of Botany* 69, 615-622.

Richardson, J. L. (1972). Palaeolimnological records from Rift lakes in central Kenya. In van Zinderen Bakker, E. M. (ed) Palaeolimnology of Africa, the Surrounding Islands and Antarctica. Balkema, Cape Town: 131-136.

Shine, J. P., Ryan, D. K. & Ford, T. E. (1998). Annual cycle of heavy metals in a tropical lake – lake Chapala, Mexico. – J. Environ. Sci. Heal. A 33: 23 –43. Strickland, J. D. H. and Parsons, T. R. (1972): A practical handbook of seawater analyses. – Fisheries Research Board of Canada, Ottawa, 309 pp.

Tabitha, A. & Woods, J.W. 1962. History of hyacinth control in Florida, *Hyacinth Control J*. 1, 19-22.

Thornton, J. A. 1982. The algal community. – In: Thornton, J. A. and Nduku, W. K. (eds.): Lake McIlwaine: The eutrophication and recovery of a tropical African man-made lake. – DrW. Junk, The Hague, pp. 106 –109.

Webber, H.J. 1897. The water hyacinth and its relations to navigation in Florida, *Bull. Div. Bot., USDA* 18, 1-20

Wilcove, D. S., & L.Y. Chen., (1998). Management costs for endangered species. *Conserv. Biol*., 12, 1405-1407.

Williamson, M., (1989). Biological Invasions: a global perspective. Chichester: John Wiley and Sons.

The life history and survival of *Neochetina* in Lake Victoria basin: Basis for biological weed control

Stephen W. Njoka[1], Gerald R.S. Ochiel[1], J. O. Manyalar[2] and J. B. Okeyo-Owuor[3]

[1]Kenya Agricultural Research Institute / Lake Victoria Environmental Management Project, (KARI / LVEMP), P. 0. Box 680, Kisumu

[2]Moi University, School of Natural Resource Management and Environment, P. 0. Box 3900, Eldoret, Kenya

[3]Moi University, School of Environmental Studies, P. 0. Box 3900, Eldoret

Email:manyalajo@yahoo.com

Abstract

Using our own experimental data and published data, we review information on the control process of water hyacinth and its status in Lake Victoria basin. Experimental results show that the mean fecundity of the two weevils is 290 and 237 eggs per female laid over a period of 16 weeks, with an adult longevity of 98 and 112 days for Neochetina bruchi and N. eichhorniae respectively. There was significant difference between the egg laying capacities of the two weevil species (p=0.002). The survival rate of the two species was significantly different (p<0.05) for all life stages except for larvae to pupa. There was no significant interaction between the species and the method of experimental egg setting (p<0.05). The fecundity of both N. bruchi and N. eichhorniae significantly decreases with time in weeks (t = 4.09; p<0.01 and (t = 3.40.09; p=0.004 respectively). N. bruchi method had a significantly (p<0.05) high larvae to pupa survival percentage (33.8 ± 6.00 for Incision Egg Setting (IES) method as compared to the Free Egg Setting (FES) method (18.7 ± 2.6). In the case of N. eichhorniae, the percentage survival for IES (25.4 ± 4.6) was also significantly higher (p>0.05) than the FES method (19.7 ± 4.1).On the basis of these results, we review and discuss data on the damage caused by the two weevil species as a basis for large-scale biological control of water hyacinth in Lake Victoria basin.

Key words: Biological control, water hyacinth, Neochetina, weevils, Lake Victoria

Introduction

Water hyacinth (*Eichhornia crassipes*) was first found in Lake Victoria in 1989 (Twongo *et al*., 1995). The weed is believed to have gained entry into Lake Victoria through River Kagera. Its spread to the numerous small water bodies in the basin has been through dispersal by man, animals and wind through seeds and live plants, thereby, quicky colonizing several smaller ponds in the Kenya basin of Lake Victoria. The plant belongs to the family Pontederiacea and its native origin can be traced to South America. The plant is known to regenerate prolifically from fragments of the stem and its seeds can remain viable for several years. Its ways of regeneration makes it very difficult to control Lee (1979). Twongo *et al*. (1995) observed that the plant flourishes in nutrient-rich waters and shallow shores with muddy bottom. Fresh water hyacinth plants in the Nile River system has an average weight of 200-300 g. About 90-95% of this weight consists of water and the resulting dry matter is only 10-15 g (Little, 1979; Freidel, 1979). Water hyacinth has received by far the most attention among invasive weeds in the literature because it is more widespread than any other (Little, 1979). In the Kenya portion of Lake Victoria, the plants are larger and heavier; with a wet weight varying from 82 to 1290 g (Gunnarson & Mattson, 1997). Other authors have found the amount of dry matter to vary between 6.2 and 9.5% (ww) (Abdelhamid & Gabr, 1991; Bolenz *et al*., 1990; Chanakya *et al*., 1993). Few methods available for the control of water hyacinth include manual harvesting, mechanical harvesting, chemical control, biological control and integrated approaches.

It is desirable to employ the use of mechanical harvesters using machinery support to control the weed from areas with colossal invasions. Kamal & Little (1970) described a manual method for harvesting of water hyacinth from the Nile River in Sudan which uses a grapnel on a long handle, thereby reducing the risk of coming into contact with cercariae of bilharzia, liver fluke worms and occasional snakes. Boyd (1974) noted that large floating plants could be manually removed using hayfork.

Blanchard (1968) described a machine known as "Aquatic Scavanger" used for mechanical harvesting of the water hyacinth. Several authors have pointed out that the cost of mechanical harvesting of water hyacinth has prohibited commercial exploitation (Boyd, 1968; 1974; Bates & Hentges, 1976). Bruhn *et al*. (1970) have pointed out that only 10% of the cost of mechanical harvesting can be recovered from water hyacinth, prompting for a cheaper harvesting method especially in developing countries.

Bryant (1970) is of the opinion that mechanical control can be as low as 25% of the cost of chemical control. Besides, it eliminates the problem of algal blooms, which follow destroying weeds by herbicides or cutting weeds and then leaving the plants in the water to rot. Bates & Hentges (1976) are however of the opinion that chemical control is economically cheaper than mechanical control. The use of 2,4-D, a phenoxyl based on butyl isopropyl amine in Lakes Chivero and McIlwaine in Zimbabwe in addition to physical methods have achieved very limited success after spending over Zim $ 2.5 million. Tests have been done on the control of water hyacinth using Sesquiterpine lactone parthenin from another weed, parthenium (*Parthenium hysterophorus* L) which is toxic to water hyacinth (Pandey, 1996) at 100 ppm.

Biological control of water hyacinth, *Eichhornia crassipes* using *Neochetina* weevils offer good opportunities for control of the weed but despite their release into Lake Victoria, studies on their impact, establishment and control has yet to be done. *Neochetina* weevils feed on water hyacinth plants causing numerous scars that debilitate the plant by removing extensive proportions of epidermal tissues thus increasing water loss and reducing the photosynthetic area. Extensive feeding around the upper petiole girdles the petiole and kill the lamina above (Goyer and Stark, 1984). The larvae first make a tunnel into the lower petiole and crown, damage tissues and buds, initially preventing flowering. As damage on the plant increases, plant growth rate is reduced and the production of new leaves and stolons is reduced (DeLoach and Cardo, 1983). Plant parts size (petiole length, laminar area, fresh weight and stolon length) decline with increasing damage of the weevils. The duration from the release of weevils to plant death takes several months depending on a combination of factors, such as temperature, nutrient status of the weed, climate, the number of weevils released and their distribution through infestation (Julien, 2001). Following the phenomenal spread of *E. crassipe*s in Lake Victoria basin, this study was aimed at quantifying the impact of two *Neochetina* weevils on the water hyacinth in selected ponds in the basin.

Materials and methods

Fecundity of *Neochetina* weevils

For each species, one mating pair of freshly emerged weevils was placed in each of ten 500ml plastic containers. Three fresh water hyacinth leaves and a 5cm long bulbous petiole cut diagonally on both ends were provided for feeding and egg laying respectively. A small amount of water (100ml) was provided to avoid desiccation. The containers were then secured on a wooden bench in the open-air laboratory at the Kenya Agriculture Research Institute (KARI), National Fibre Research Centre (NFRC) - Kibos.

The adults were allowed to mate and lay eggs on the petioles for as long as they survived. Egg count was done every two days and recorded for each container while fresh petioles and leaves were provided after every count. To record the number of eggs laid by each female weevil, a disc of about 2 mm was sliced off from both ends of the petiole and observed under a hand lens. This exercise was continued until either no more eggs could be recovered or the female weevil died. Each of the 10 pairs placed in 500ml plastic containers formed a replicate giving a total of 10 replications. The oviposition rate was determined by counting the number of eggs laid by each female per day.

Data was also taken on daily temperature using a maximum and minimum thermometer while the relative humidity was recorded using a wet and dry bulb thermometer. Fecundity was obtained by counting all the eggs laid by each female throughout its lifetime. The duration of egg laying was found by counting the number of days from the onset to the end of oviposition.

Life cycle and development of *Neochetina* weevils

To determine the life cycle and development of the two species, *Neochetina bruchi* and *N. eichhorniae*, twenty sexed pairs of each species were harvested from the respective mass rearing tanks. Each pair was placed in 500 ml plastic container and provided with three water hyacinth leaves and a 5cm long bulbous petiole for feeding and egg laying respectively. To avoid desiccation, 100 ml of water was added to each container. The containers were covered with muslin cloth to keep off other insects. The containers were then placed on a wooden bench in the open-air laboratory at Kibos. This set up was used to study the life cycle and development of the two species using standard statistical techniques as follows:

Egg to larva duration

Two sets of nine 500 ml plastic containers each for *Neochetina bruchi* and *N. eichhorniae* were arranged in a Complete Randomised Design replicated three times in the laboratory to investigate the egg to larva duration. In each container, sets of 10 eggs laid on 2mm water hyacinth petiole discs harvested from the rearing tanks were introduced. A little amount of water (100 ml) was added to avoid desiccation. The sets were checked daily for hatching until no further hatching was observed. The number of eggs hatched gave fertility data. The duration (days) from egg to larva gave the period of incubation.

Larva to pupa duration

To study the developmental period from larva to pupa of *Neochetina bruchi* and *N. eichhorniae*, two sets of nine 20 litre plastic buckets were filled up to ¾ level with water. One healthy water hyacinth plant was then placed in each bucket. Sets of ten eggs borne on 2mm petiole discs were harvested from the respective species rearing tanks. The discs were then inserted at mid length of the plant's petioles using sterile scalpels to make the incision. Each bucket was then covered with an insect proof muslin cloth mounted on a 75 cm x 75 cm wooden cage. The duration (days) from the first instar larvae to pupal formation was recorded to give the larval development period. The experiment was conducted in an open-air laboratory at NFRC - Kibos. It was set up in a Complete Randomised Design and replicated three times.

Pupa to adult duration

Since the pupa stage of these weevils occurs in the submerged roots of the plant. To study the developmental period from pupa to adult for the two weevil species, two sets of nine 500ml plastic containers were arranged in a complete Random Design and replicated three times at NFRC Kibos. In each container, 10 freshly formed pupae borne on root hairs were introduced. The containers were then covered with an insect proof muslin cloth. The duration (days) taken from pupa to adult emergence was recorded.

Results

Fecundity

A two-way analysis of variance showed that there were significant ($P<0.05$) differences between the egg laying capacities of the two weevil species and also between the time in weeks during the egg laying period. After removing the effect of time in weeks, re-analysis of egg production using one-way analysis of variance confirmed that there were significant differences between the number of eggs laid by the two weevil species ($F_{318,\ 1} = 4.26$; $p < 0.05$) and the log-transformed data ($F_{318,\ 1} = 6.01$; $p < 0.05$). A two-sample t-test showed that there was a significant ($P<0.05$) difference in the mean number of eggs laid by the two weevil species with *Neochetina bruchi* laying more eggs (292) than *N. eichhorniae* (236) cumulatively (Figure 1).

The effect of time was analyzed for each species using regression of the number of eggs on the time in weeks: The results for *N. bruchi* are shown below:

$F = 33.4 - 1.79X$, Where F is fecundity and X is Weeks

With the following regression diagnostics: Constant ($t = 7.89$; $p < 0.05$) and Weeks ($t = 4.09$; $p<0.01$). This regression analysis showed that the fecundity of *N. bruchi* significantly decreases with time in weeks. The coefficient of determination was 54.4%, showing that 54.4% of the decrease in fecundity could be explained by the time factor (weeks). The regression line was itself significant with an $F_{14,\ 1}$ of 19.69 and p-value of 0.001.

For *N. eicchorniae*, the relationship between fecundity (F) and time in weeks could be best described by the following linear equation:

$F = 28.4 - 1.59X$, Where F is fecundity and X is Weeks

The regression diagnostics were: Constant ($t = 6.26$; $p<0.0005$) and Weeks ($t = 3.4$; $p=0.004$), while the coefficient of determination was 45.3%. Only 45.3% of reduction in fecundity for *N. eichhorniae* could be explained by variation in time (weeks). The regression line was also significant with an $F_{14,\ 1}$ of 11.58 and p-value of 0.004.

The mean number of eggs laid by *N. bruchi* was generally above 20 for the first 10 weeks of the study. The number of eggs laid by this species decreased steadily in the 11th week and by the 16th week, an adult female laid only 2 eggs on average. For *N. eichhorniae,* egg laying pattern was more irregular with a mean laying rate of 11-31 eggs per week for the first 10 weeks. By the 11th week, the mean egg laying per female had reduced to only 8 and by the 16th week, *N. eichhorniae* females were not laying any eggs (Table 1).

The cumulative number of eggs laid by *N. bruchi* reached 292 by the 16th week while the number laid by *N. eichhorniae* reached 236 during the same period of time.The overall daily oviposition rate for *N. bruchi* (2.6) was higher than that of *N. eichhorniae* (2.1). The egg laying capacity shows an initial increasing trend up to the 7th week for both species and then a general decline for both species up to the 16th week (Figure 2). The regression equations indicate that *N. bruchi* egg laying would completely cease by the 19th week while that of *N. eichhorniae* would completely cease by the 18th week.

The differences between replicate observations (individual females) was insignificant ($P>0.05$) for both *N. bruchi* and *N. eichhorniae*.

Life cycle and development time

The generation time of these weevils took 73 and 94 days for *N. bruchi* and *N. eichhorniae* respectively with the duration (days) of the distinct developmental stages as shown in Table 2. The durations taken by each developmental stage in both species were compared using a two sample t- test. The mean egg to larva duration was the shortest in both species taking 11 $\pm$ 0.2 days in *N. bruchi* and 13 $\pm$ 0.44 days in *N. eichhorniae*. These were not significantly different ($P<0.05$). The larval development stage took the longest time spanning 55 $\pm$ 0.79 days in *N. eichhorniae* and 31 $\pm$ 0.31 days in *N. bruchi*. This was significantly different ($P<0.05$). The physically inactive pupal stage took significantly (P<0.05) longer duration of 31 ± 0.49 days in *N. bruchi* compared to 25 ± 0.86 days in *N. eichhorniae*.

Table 1. Mean ± SE and cumulative number of eggs laid by *N. bruchi* and *N. eichhorniae* over a 16-week experimental period at KARI, Kibos 2001.

Weeks	Mean number of eggs per female per week		Cumulative egg laying	
	N. bruchi	***N. eichhorniae***	***N. bruchi***	***N. eichhorniae***
1	19 ± 4	11 ± 2	19	11
2	25 ± 3	27 ± 3	44	38
3	24 ± 3	22 ± 2	68	60
4	22 ± 3	23 ± 3	90	83
5	31 ± 3	14 ± 2	121	97
6	31 ± 6	22 ± 4	152	119
7	35 ± 6	31 ± 4	187	150
8	21 ± 3	20 ± 4	208	170
9	32 ± 5	31 ± 3	240	201
10	21 ± 3	21 ± 3	261	222
11	12 ± 2	8 ± 2	273	230
12	6 ± 2	3 ± 1	279	233
13	3 ± 1	1 ± 1	282	234
14	5 ± 2	1 ± 1	287	235
15	3 ± 1	1 ± 1	290	236
16	2 ± 1	0	292	236

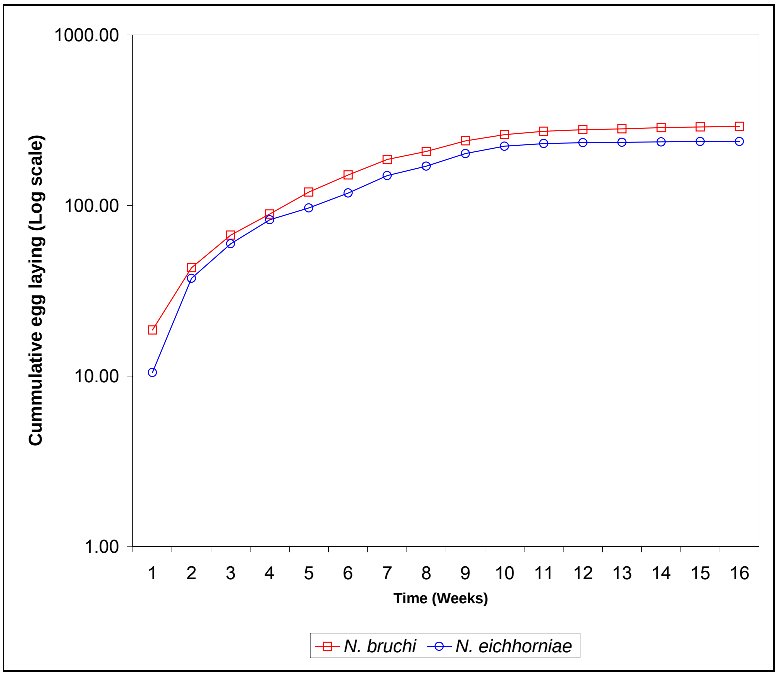

Figure 1: Cumulative egg production (log scale) by *N. bruchi* and *N. eichhorniae* over a 16 weeks experimental period at KARI, Kibos 2001.

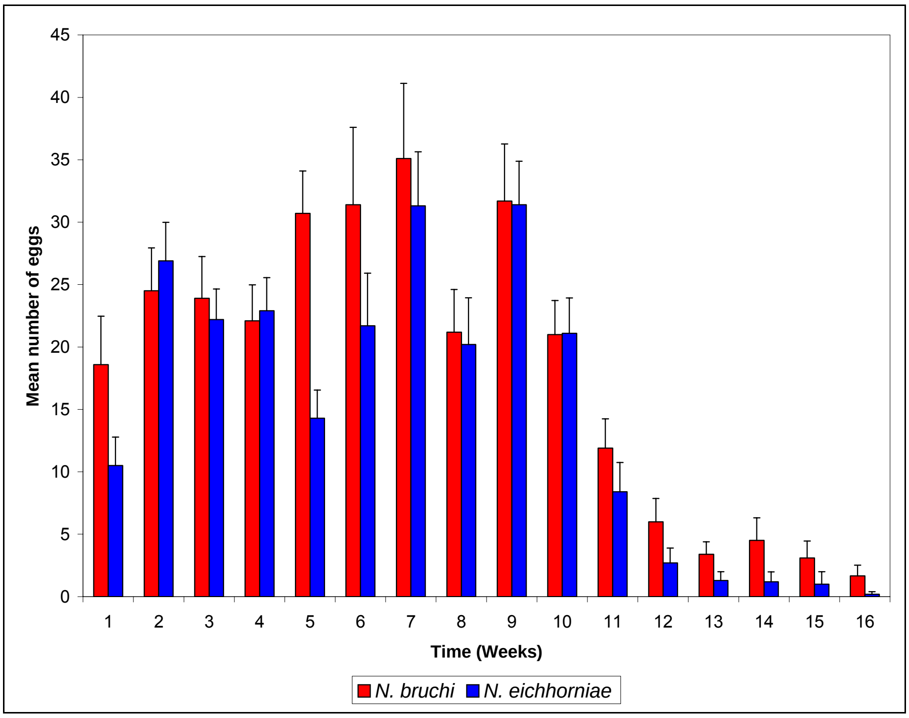

Figure 2. Mean fecundity of *N. bruchi* and *N. eichhorniae* over a 16 weeks experimental period at KARI, Kibos 2001.

Table 2: Developmental duration for *Neochetina* weevils' life stages.

Life stage	*N. bruchi* Mean ± SE	*N. eichhorniae* Mean ± SE	Significance (P>0.05)
Egg	11±0.2	13±0.44	NS
Larva	31±0.31	55±0.79	*
Pupa	31±0.49	25±0.86	*
Generation Time	**74**	**93**	

Effect of Temperature

During the period of these studies, the mean weekly ambient temperature varied from a minimum of 21.44ºC (week 11) to a maximum of 23.78°C (week 8) thus giving a range of 2.37°C. A graphical trend of egg laying versus temperature for the *Neochetina* weevils is shown in Figure 4.3.

From these results, it was possible to determine the relationship between ambient temperature and fecundity in both *N. bruchi* and *N. eichhorniae*. For *N. bruchi*, the relationship was weak, with coefficient of determination of only 28.69% (Figure 4.4). Temperature was found to be directly related to fecundity according to the equation:

F = 9.2X– 196.33, Where F is Fecundity and X is Temperature.

For *N. eichhorniae*, the relationship was slightly better than *N. bruchi*, with coefficient of determination of only 37.78% (Figure 4.5). Temperature was also found to be directly related to fecundity according to the equation:

F = 10.8X- 229.95, Where F is Fecundity and X is Temperature

Population projection matrices

From survival observations on life cycle stages, the rate of increase (r) in a stable population of *N. bruchi* was as 0.13141 while the expected number of replacements (R_o) was 1.16195 and the mean age of parents of offspring of a cohort (μ_1) was 1.15201. The estimated generation time (T) (time for increase of R_o) was 1.14225. The population projection matrix for *N. eichhorniae* not stable enough to enable calculation of the rates (Figure 3).

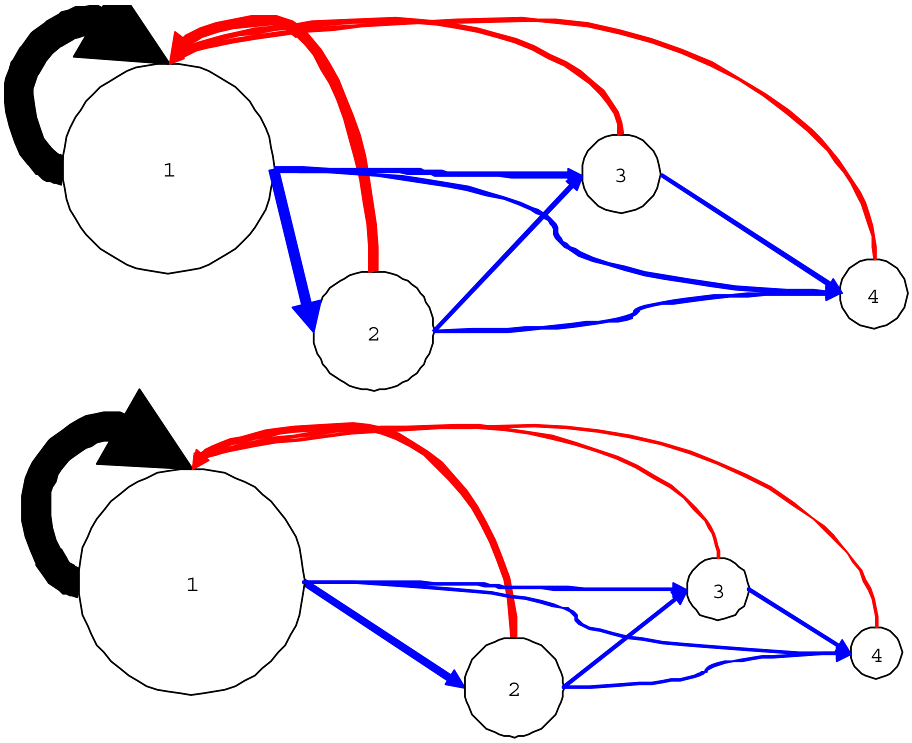

Figure 3: Life cycle of *N. bruchi* (above) and *N. eichhorniae* (below). The red arcs represent the proportional contribution to reproduction (the number of females per female that survive to stage 1), and the blue lines indicate survival rates during a single time step from egg to adult stage.

Table 3: The age structure and reproductive values of *N. bruchi* and *N. eichhorniae* based on age specific life tables and population matrices

Neochetina bruchi		**Age/stage structure**	**Reproduction value**
	Eggs	62.23	64.06
	Larvae	18.43	20.72
	Pupa	10.06	8.66
	Adult	9.28	6.55
Neochetina eichhorniae		**Age/stage structure**	**Reproduction value**
	Eggs	74.57	75.52
	Larvae	14.00	14.81
	Pupa	6.39	5.66
	Adult	5.04	4.01

Discussion

The fecundity of the two species *N. bruchi* and *N. eichhorniae* was shown to be significantly ($p<0.05$) different with *N. bruchi* laying more eggs (292) than *N. eichhorniae* (236). The former also had a higher oviposition rate at 2.6 eggs per day compared to 2.1 eggs for *N. eichhorniae*. Based on these fecundity factors, it seems logical to infer that *N. bruchi* is a more prolific producer of eggs than *N. eichhorniae* in the region under study. Ochiel and Njoka (1999) however noted that despite the higher egg production by *N. bruchi*, it was always out competed in numbers by *N. eichhorniae*, which is not fastidious in its diet. Under natural conditions, *N. bruchi* is a more fastidious feeder and is soon out competed by *N. eichhorniae,* which has no preference for succulent short bulbous petioles. Results of this study compare well with the work of Ogwang and Molo (1997) in Uganda.

For *N. bruchi*, the egg stage took a shorter duration (7.6 days) in Argentina (DeLoach and Cordo, 1976) compared to 11 days in Uganda (Ogwang and Molo, 1997) and 11 days in Kenya (This study). The larval duration, the most important in biological control of the weed, took between 31 days in Kenya, compared to 32 days in Argentina and the longest duration was recorded for Uganda at 35 days. The physically inactive stage of pupa took 30 days in Argentina, 31 days in Kenya and 33 days in Uganda. The total generation time took longest in Kenya at 73 days while it took 69.6 days and 72 days in

Argentina and Uganda respectively (Table 2). It would appear that the life cycle *N. bruchi* compare favourably and are similar in the 3 countries, making it easier to import them from Argentina and Uganda for biological control use against water hyacinth in the Kenyan part of Lake Victoria.

For the species *N. eichhorniae* (Table 2), the egg stage took 7-14 days in Argentina while it took 10 days in Uganda and 14 days in Kenya. De Loach and Cordo (1976) describe a long duration of between 75 –90 days for the larval stage of this species in Argentina while it took 58 and 55 days only in Uganda and Kenya respectively. The otherwise inactive pupa stage took 14-20 days in Argentina and longer durations of 28 and 25 days in Uganda and Kenya respectively. The entire generation time for *N. eichhorniae* took longest in Argentina (96-120 days), which is more or less similar to that in Uganda and Kenya where it took 96 and 94 days respectively. Thus although Argentinean conditions may be markedly different from that in the East African countries the similarity observed in the generation time may make it easier for *N. eichhorniae* to adapt well and be used for biological control of water hyacinth in any of these countries.

References

Abdelhamid, A. M. and A. A. Gabr (1991). Evaluation of water hyacinth as feed for ruminants. *Archives of Animal Nutrition 41(7/8):745-756.*

Bates, R. P and J. F. Hentges (1976). Aquatic weeds - eradicate or cultivate ?. *Economic Botany 30(1):39-50.*

Blanchard, J. (1968). Winter Parks' lake weed management programme. *Hyacinth Control Journal 7:30-31.*

Bolenz, S., H. Omran and K. Gierschner (1990). Treatment of water hyacinth tissue to obtain useful products. *Biological Wastes 33(4):263-274.*

Boyd, C. E. (1968). Fresh water plants: potential source of protein. *Economic Botany 22:359-368.*

Boyd, C. E. (1974). Utilization of aquatic plants. p 107-114. *In: Aquatic Vegetation and its use and Control. Edited by D. S. Mitchell. Paris, Unesco.*

Bruhn, H. D., R. G. Koegel and D. F. Livermore (1975). Utilization of aquatic vegetation. *Paper presented at the Annual Meeting of the Northern Atlantic Region of the American Society of Agricultural Engineering, New York. 13 p.*

Bryant, C. B. (1970). Aquatic weed harvesting - effect and costs. *Hyacinth Control Journal 8(2):37-39.*

Chanakya, H. N. B., G.S. Meena and K. S. Jagadish (1993). Solid phase biogas production with garbage or water hyacinth. *Bioresource Technology 46:227-231.*

DeLoach, C.J. and H.A. Cordo (1983). Control of water hyacinth by *Neochetina bruchi* (Coleoptera: Curculionidae: Bargoini) in Argentina. *Environmental Entomology 12:19-23*

Freidel, J. W. (1979). Population dynamics of water hyacinth (*Eichhornia crassipes* Mart. Solms) with special reference to the Sudan. *Berichte aus dem Fachgebit Hertologie der Univeritat Hohemheim. Herausgegeben von W. Koch und A Kemmer, Stuttgart..*

Goyer, R.A. and J.D. Stark (1984). The impact of *Neochetina eichhorniae* on water hyacinth in Southern Louisiana. *Journal of Aquatic Plant Management 22: 57 - 61.*

Gunnerson, C. and C. Mattson (1997). Water Hyacinth: Trying to turn an environmental problem into an agricultural resource. *M. Sc. Thesis. Swedish University of Agricultural Sciences. Minor Field Studies No. 25.*

Julien M.H. (2001) Biological Control of Water Hyacinth with Arthropods: a Review to 2000. *In: Australian Centre for International Agricultural Research Proceedings No.102.*

Julien M.H., Griffiths M.W. and A.D. Wright (1999). Biological control of water hyacinth. The weevils *Neochetina bruchi* and *N.eichhorniae*: biologies, host ranges, and rearing, releasing and monitoring techniques for biological control of *Eichhorniae crassipes*. *ACIAR Mongraph No. 60.*

Kamal, I. A. and E. C. S. Little (1970). The potential utilization of water hyacinth for horticulture in Sudan. *PANS 16(3):488-496.*

Lee, B. (1979). Insects for controlling water weeds. *Rural Research 105:25-29.*

Little, E. C. S. (1979). Handbook of utilization of aquatic plants: A review of World Literature. *FAO Fisheries Technical Paper No. 187.*

Pandey, D. K. (1996). Phytotoxicity of Sesquiterpene Lactone Parthenin on aquatic weeds. *Journal of Chemical Ecology 22(1):151-160.*

Twongo, T., F. W. B. Bugenyi and F. Wanda (1995). The potential for further proliferation of water hyacinth in Lakes Victoria, Kyoga and Kwania and some urgent aspects for research. *The African Journal of Tropical Hydrobiology and Fisheries 6:1-10.*

Macrophytes of Lake Victoria and succession after invasion of Water Hyacinth

Reuben Omondi[1] and Mwende Kusewa[2]

[1]Kenya Marine and Fisheries Research Institute, P. O. Box 1881, Kisumu, Kenya
Email:omondir2001@yahoo.com
[2] Kenya Agricultural Research Institute, P. O. Box 680, Kisumu, Kenya
E-mail: tkusewa@yahoo.com

Abstract

The distribution of Lake Victoria macrophytes is described. Succession of macrophytes in the lake became more dramatic and dynamic after invasion of water hyacinth. The weed pushes and smothers other free-floating macrophytes like *Pistia stratiotes*. It then provides substrates for the emergent *Vossia cuspidata,* which later reduces its population by competition for light and nutrients. The noxious weed is believed to have led to extinction of *Azolla nilotica* in the lake.

Key words: Lake Victoria, Macrophytes, zonation, succession

Introduction

Macrophytes are higher plants that grow in ecosystems whose formation has been dominated by water and whose processes and characteristics are largely controlled by water. Factors that influence the establishment of macrophytes include: depth, topography, type of substrate, exposure to currents and/or wind and water turbidity. The distribution of macrophytes is often related to their mode of attachment (Sculthorpe, 1976). In Lake Victoria, the numerous numbers of haplochromines before the introduction of *Lates niloticus* hindered the establishment of macrophytes in the inshore areas by constantly disturbing the substrate (Witte *et al*., 1992a). The distribution, permanency and quality of the water bodies available for their occupation govern the distribution and ecology of these plants. The most variable environmental factors of basic ecological importance for aquatic plants are the length of the period during which water is present, whether the water is still or moving, the availability of plant nutrients and the quality and quantity of light penetrating into the water.

Macrophytes can be subdivided into four groups on the basis of their water requirements and habitats (Plate 1). Submerged macrophytes are those that are completely covered with water. These have leaves that tend to be thin and finely divided adapted for exchange of nutrients with water. Floating leafed macrophytes are those that are rooted but have floating leaves while free floating are those that float on the water surface. The last group is the emergent macrophytes. These are rooted plants with their principal photosynthetic surfaces projecting above the water. Emergent macrophytes dominate the shoreline flora while the middle and lower littoral zones supports stands of floating-leafed macrophytes. Macrophytes usually show a succession of zones between the dry land and water, each zone with a dominating plant species. Variation in the latter depends on the duration of the flooding and may also be affected by ecological succession - where a plant community alters environmental conditions in a way that makes the habitat less favourable for its own survival but more favourable for the development of a different community. The present study was carried out to determine the distribution, zonation and succession patterns of macrophytes in Lake Victoria**.**

Materials and methods

The Kenyan portion of Lake Victoria can conveniently be divided into Nyanza Gulf and open waters. The Nyanza Gulf receives water from five rivers viz. Nyando, Sondu, Awach, Kibos and Oluch while the open waters receive waters from Rivers Yala, Nzoia and Kuja. Monitoring and surveillance of macrophytes in the lake was done monthly using a canoe with an outboard engine. Macrophyte species occurring in the various sites were recorded and the depths at which they occurred. Specimens were collected for herbarium preparation and further identification. Satellite image analysis was used to estimate the macrophyte population densities within the lake especially water hyacinth.

Results and discussion

A total of 20 families and 28 species common macrophytes were recorded from the different sites in the lake (Table 1).

More species of macrophytes were found at the river mouths and sheltered bays. This was probably so because these areas have muddy substrates suitable for attachment and the plant nutrients from the rivers enriched the environment. Comparison of open water station macrophytes and gulf station ones further shows that the distribution of macrophytes, especially emergents, is related to the type of substrate. Stations within the gulf were dominated by *Cyperus papyrus* while those in the open waters (Yala and Kuja/Migori) were dominated by *Phragmites australis*. Sediment deposition from the many rivers in the gulf seemed to favour papyrus.

Plate 1.

Table 1. Some common macrophytes recorded in Lake Victoria

Araceae	**Lemnaceae**
Pistia stratiotes	*Lemna* sp
Asteraceae	**Lentibulariaceae**
Enydra fluxuans	*Utricularia inflexa*
Sphaeranthus spp	**Malvaceae**
Azollaceae	*Hibiscus diversifolius*
Azolla pinnata	**Najadaceae**
Ceratophyllaceae	*Najas horrida*
Ceratophyllum demersum	**Nymphaeaceae**
Compositae	*Nymphaea lotus*
Melanthera scandens	**Onagraceae**
Convolvulaceae	*Ludwigia stolonifera*
Ipomea aquatica	*L. abyssinica*
I. cairica	**Polygonaceae**
Cyperaceae	*Polygonum setosulum*
Cyperus papyrus	**Pontederiaceae**
C. rotundus	*Eichhornia crassipes*
Elegant cyperus	**Potamogetonaceae**
Gramineae	*Potamogeton schweinfurthii*
Phragmites australis	*P. pectinatus*
Vossia cuspidata	**Trapaceae**
Echinocloa sp	*Trapa natans*
Hydrocharitaceae	**Typhaceae**
Vallisneria spiralis	*Typha domingensis*

Succession

Until the invasion of Eichhornia crassipes only small changes were seen in the macrophyte composition during the rainy season. Before water hyacinth invasion, the only common floating macrophyte was Pistia stratiotes and to a lesser extent Azolla sp. The grass, Vossia cuspidata formed a thin layer behind P. stratiotes followed by Cyperus, Aeschynomene, Phragmites and Typha. However, after the invasion of water hyacinth Pistia stratiotes and Azolla sp were either smothered or pushed towards the shore. These formed a compact substrate mass which led to the increase of Vossia cuspidata and Echinocloa sp communities that effectively grew afloat the substrate utilizing the nutrients within. Water hyacinth also reduced the populations of floating leafed and submerged macrophytes by cutting light and competition for nutrients. This could have led to the extinction of A. nilotica that was last seen at the mouth of River Nyando in the 1990s. The other macrophytes whose communities were decimated by water hyacinth included N. lotus, Ceratophyllum demersum, Najas horrida and Trapa natans.

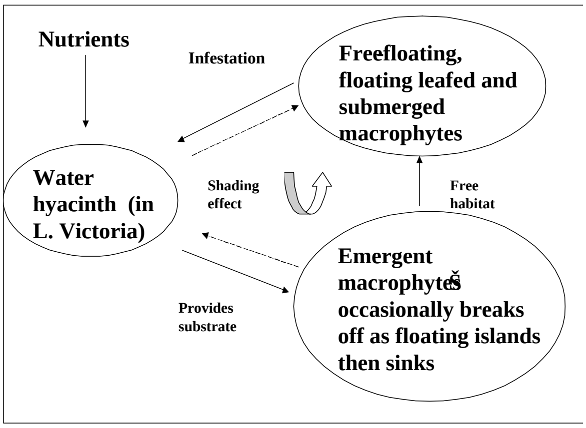

Figure 1. Macrophyte succession in Lake Victoria.

The establishment of water hyacinth biological control agents and the subsequent successful reduction of the hyacinth resulted in the increase of the above plants in the lake. It is believed that the emergent plant species contributed to the control of the hyacinth by out-competing it, particularly in the littoral zones (Figure 1). These plants, which were not-free floating, used water hyacinth as a substrate to grow upon. This was first observed in the Winam Gulf late 1999 and early 2000. This cycle is expected to continue so long as the biological control agents exist in the lake.

Conclusion

The invasion of Lake Victoria by water hyacinth led to the initial decrease in the population of emergent macrophytes as the hyacinth, which doubles its population in two weeks under favourable conditions, out-competed them. However, this trend was reversed with the reduction of water hyacinth population due to biological control. Macrophytes such as *Vossia cuspidata* and *Echinocloa* sp quickly increased in population along the shores of the lake. These, however, cannot survive in the lake once total decomposition and disintegration of the heterotrophic layer of the floating mats takes place. *Pistia stratiotes* and *Azolla nilotica*, which were the only floating macrophytes are now only found in ponds and irrigation canals along the lake shore and not in the main lake. Water hyacinth ‘hot-spot’ areas have been observed in the lake and unless environmental degradation around the lake is reduced, then the weed is expected persist and dominate these areas.

Recommendations

1. There is need to control environmental degradation in the Nyanza gulf catchment. This will prevent eutrophication in the lake, a perfect condition for re-invasion of water hyacinth.

2. There is need to introduce additional and/or alternative biological control agents in the hot-spot areas especially at the river mouths.

References

Sculthorpe, C.D. (1976) The Biology of Aquatic vascular plants. Edward Arnold, London.

Witte, F., T. Goldschmidt, P. C. Groudswaard, W. Ligtvoet, M. J. P van Oijen & J. H Wannink 1992a. Species extinction and concomitant ecological changes in Lake Victoria. *Neth. J. Zool.* **42**: 214-232.

Duckweeds spread in the Maracaibo Lake, Venezuela

Crisanto Silva Aguilera

Urbanizacion Villa Aliaza, sector 3 Carrera Comalapa N°4, Puerto Ordaz, Estado Bolivar, Venezuela
Email: crisantosilva@yahoo.com

Abstract

The Maracaibo Lake which is one of the biggest lakes in the world and the biggest one exposed to the sea, has since 2004 been victim to the spread of the lenna spp, commonly called duckweed or "lenteja de agua" (water lentil) in South America. Duckweed covered 2,025 km^2 of the Maracaibo lake surface in June 2004, which corresponds to 15% of the total surface. The lake basin is one of the most important in the country in terms of cultural heritage, ecological diversity and economical resources. Mechanical methods of control give an instant effect, which can last for a reasonable length of time. A careful watch, however, needs to be kept regarding re-growth of the remaining plants, and remedial action should be put into effect before the problem reaches nuisance proportions again. The removed weed can be composted with chemicals. Continuous removal of this plant would often be necessary. The accelerating growth of the duckweed in the Maracaibo basin is indicator of the high degree of degradation of the ecosystem of the lake and lake catchments.

Key words: Eutrophication; Duckweeds; Maracaibo lake; Venezuela

Introduction

The Maracaibo lake, which is one of the biggest lakes in the world and the biggest one exposed the sea, has for some months been victim to the spread of the *lenna spp*, commonly called duckweed or "lenteja de agua" (water lentil) in South America. The latest reports indicate that duckweed covers 2,025 km^2 of the Maracaibo lake surface, which corresponds to 15% of the total surface. The Maracaibo lake has a surface area of 13.500 km^2, and an approximate length of 160 km from North to South and 120 km from East to West. It is located in northwest Venezuelan state of Zulia (Figure 1).

The lake basin is one of the most important in the country in terms of cultural heritage, ecological diversity and economical conditions. The economy of the lake includes the location of the most important oil infrastructures of the country; one million barrels per day are extracted in the lakebed by means of hundreds of oil drilling platforms. Fishery is the second largest economical activity on the lake, and the main activity for most of the population living near the lake.

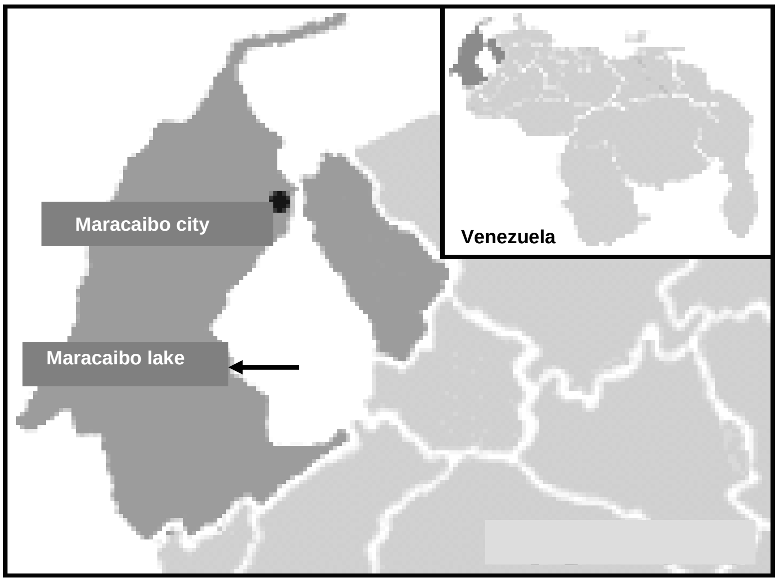

Figure 1. Maps of the Zulia state and Venezuela indicating the location of the Maracaibo lake.

Duckweeds

Duckweeds are small free-floating plants that often form dense mats on the surface of still or slowly flowing water. They grow best in eutrophic waters rich in nutrients, such as phosphate nitrate, and with an element of organic enrichment from leaf litter. The leaves are small, often not exceeding 5mm in length. They are very common in tropic lakes, reservoirs, water channels and ponds. The nutrients originate from pollution stemming from excessive usage of fertilizers or possibly from an imbalance of fish populations or waterfowl, which could result in excessive nitrogenous waste products in the water (Figure 3). The re-circulation of nitrogen and phosphorus from the cycle of growth and decomposition of duckweeds may also contribute to the high levels of these elements.

The duckweed mats can be composted and used as "green manure." They can also be fed to livestock, rabbits, poultry and fish. It has been estimated that 4 hectares of duckweeds could theoretically supply 60% of the nutritional needs of 100 dairy cows, the manure of which could be recycled to provide fertilizer for the thriving duckweeds. According to Harvey and Fox (1973) one hectare of water area is sufficient to raise 4000-7000 chickens and ducks during a vegetation period. According to Rejmankova (1981) one hectare of duckweed cover is sufficient to produce protein for 480 ducks during the warm season. The utilization of duckweeds as food for animals is summarized by Landolt and Kandeler (1987).

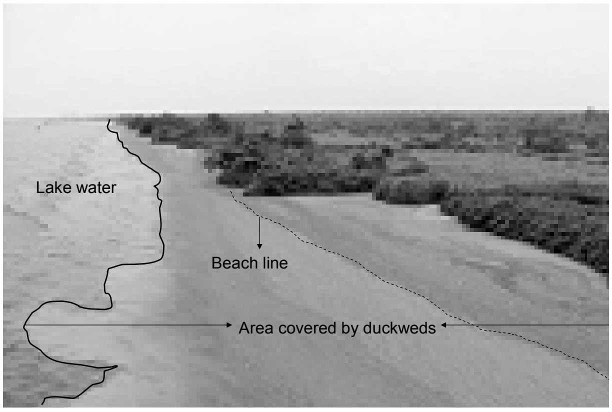

Figure 2. Duckweed in the coast of the Maracaibo lake.

Common methods to reduce the duckweed

Stopping the inflow of nutrients and the repetitive removal of the duckweed layer will greatly reduce the growth of duckweeds. Methods for controls the *Lemna* spread included biological, chemical and mechanical controls. Biological control had been proven to be successful. Grass carp, for example, will eat the *Lemna* species. The use of shade has been successful in reducing the amount of duckweed growth, although very deep shade is often required. Planting trees on the south side of a water body can result in sufficient shading. *Lemna* does not compete well with other floating leaved plants, such as water lilies, and planting species with floating leaves can substantially reduce the nuisance level of duckweeds. It is necessary to take into account that studies need to be proven before the usage biological controls, in order to avoid unwanted secondary effects in the lake ecosystem.

Chemical controls have also been proved successful; in general, duckweeds are very sensitive to herbicides. In fact, duckweeds are often used to test the toxicity of herbicides and to detect the presence of herbicides in water. Some algaecides are extremely toxic to some species of *Lemna*. *Lemna* species are susceptible to herbicides containing diquat, terbutryn and glyphosate. Reglone (diquat) is a liquid concentrate applied directly to the water (Newman, 2001). It is necessary to take into consideration that destroying the duckweed layer with herbicides does not solve the problem of excess nutrients in the water. In addition, the chemical herbicides may be toxic to the animal life, either directly or through biological magnification. Because of the exponential growth rate of duckweeds, herbicides must be used repeatedly (perhaps several times a year). Ideally, in terms of chemical control, it is best to eliminate the influx of concentrated nitrates and phosphates into the water and to avoid the use of concentrated fertilizers Fomin (1987). The most frequent and recommended method for the removal the duckweed is the manual or mechanical removal of the duckweed cover. This method can also remove a lot of the nitrogen and phosphorus nutrients.

Duckweed removal in the Maracaibo Basin

Manual or mechanical is the current method in use to remove duckweed from the Maracaibo basin. Up until now 90,000 tons have been removed in this manner; however, it has been calculated by Venezuelan Ministry of the Environment that 20 million tons are currently present (Figure 3).

Figure 3. Oil drilling platform surrounding by duckweed in the Maracaibo lake.

800 workers, consisting of a large number of fisherman, and around 30 special vessels and machines that are normally are used to attend to oil spills, are employed for this purpose. To combat the growth of the duckweeds the government has invested 300.000 US$, plus they established infrastructures of PDVSA and Municipalities. They are planning to invest 2,000,000 US$ more in the near future. Is necessary to take into account that it is impossible to remove every plant by mechanical means and that re-growth is inevitable. Mechanical

methods of control give an instant effect, which can last for a reasonable length of time.

A careful watch, however, needs to be kept regarding re-growth of the remaining plants, and remedial action should be put into effect before the problem reaches nuisance proportions again. The removed weed can be composted with chemicals. Continuous removal of this plant would often be necessary.

Duckweeds prefer still water and increasing the disturbance of the water surface can reduce the amount of duckweed. It has been proven that increasing the amount of boat traffic will reduce the competitive ability of the species and may contribute to their eventual elimination (Newman, 2001).

Conclusions

The accelerating growth of the duckweed in the Maracaibo basin is indicator of the high degree of degradation of the ecosystem of the lake and lake catchments. This implies that environmental controls until now are not enough. Better controls, however, still may not totally imply that the lake is unpolluted. Another strong source of pollution can also come from the lake's sub-catchments. Rivers and ravines effluents into the lake are known to have high amounts of pollution coming form fertilizer, pesticides, herbicides and organic materials from nearby farms. In addition, some chemicals used in the oil industry to clean the tanks and deposits are sources of pollution, which in some cases are not controlled, causing these chemicals to go directly into the lake.

It is important to mention that these environmental problems stem from the heritage of old practices from the last century, in which only the extraction of the resources was considered, with little consideration for sustainability or environmental risk. Now this situation is changed, with the new sustainable vision of the current central government, which through the Ministry of the Environment places the preservation of the environment as its priority.

Discussions

Under the Intituto para la Conservación del Lago de Maracaibo, ICLAM (Institute for the conservation of the Maracaibo Lake), adscript to the Ministry of the Environment, the research of the duckweed situation is still in progress. Studies of concentration levels and sources are needed, which includes monitoring of nutrients, organic matter, heavy metal, inorganic and organic pollutions and sedimentation in the lakebed, rivers and ravine effluents. This first step of information collection will then be coupled with GIS for the final processing of the data, which will assist in the forthcoming decisions concerning subsequent actions.

Efforts to prevent pollution in the lake basin is of interest for all the players and participants that interact in any way with the lake environment, including oil companies, farms, fishermen, nearby communities, and the central and local governments. Towards a joint effort amongst these groups for their mutual benefit, and for the sustainability of the lakes resources.

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References

Fomin, A., Moser, H., Pickl, C. & Arndt U. (1987). Ökotoxikologische Untersuchungen saurer Tagebaugewässer der Bergbaufolgelandschaft Lausitz während der Sanierung. (Hrsg.): Abbau von Bodenschätzen und Wiederherstellung der Landschaft. 29. Hohenheimer Umwelttagung, Verlag Günter Heimbach, Ostfildern, 165-178.

Harvey R.M. & Fox, J.L. (1973) Nutrient Removal Using Lemna minor. J. Water Poll. Control Fed. 45: 1928-1938.

Instituto para la Conservación del Lago de Maracaibo, Venezuela. http://www.iclam.gov.ve/

Landolt E. & Kandeler, R. (1987) The Family of Lemnaceae: A Monographic Study" in Veroff. Geobot. Inst. ETH, Stiftung Rubel 95 pp. 382-389.

Newman J. (2001) CAPM Information Sheet 25. Centre for Aquatic Plant Management, Broadmoor Lane, Sonning, Reading, Berkshire.

Rejmankova, E. (1981) On The Production Ecology of Duckweeds. Intern. Workshop on Aquatic Macrophytes, Illmitz, Austria.

Wayne's world, Professor Armstrong, USA. http://waynesword.palomar.edu/1wayindx.htm

Analysis of sediments, surface runoff and nutrient dynamics using Agricultural Non-Point Source Pollution Model (AGNPS) in Mirera sub-catchment of Lake Naivasha drainage basin, Kenya

Onyango Ruth Amondi[1], Onyando Japheth O. [2], Chemelil Mathew C. [3] and Becht Robert[4]

[1] MSc. Student, Dept. of Agricultural Engineering, Egerton University
P.O. Box 536, Njoro, Kenya
[2,3] Dept. of Agricultural Engineering, Egerton University, P.O. Box 536, Njoro, Kenya
[4] ITC Ecschede, Holland

Abstract

Sedimentation is a major threat to surface water bodies such as lakes and rivers where the eroded soils from the hill slopes are finally deposited. The extent of erosion depends on the magnitude of the surface runoff which transports the eroded sediments. Increased sediment concentration in surface water bodies can be detrimental to aquatic systems by; reducing light penetration, clogging aquatic vegetation and spawning gravels, and reducing water quality. Suspended sediments act as a media of transport for nutrients thereby accelerating the rate of nutrient accumulation in surface water bodies. Due to lack of data, it is necessary that other techniques such as modeling be used to analyze the dynamics of surface runoff, sediment and nutrient simultaneously. The Agricultural non-point Source (AGNPS) pollution model was found to be suitable to analyze these three variables together hence it was applied in Mirera sub-catchment located in Lake Naivasha drainage basin.

Mirera sub-catchment of 1.82 km^2 drains into a pond created through road construction works in 1998 and had accumulated sediments over the same period. Through use of GIS and remote sensing, it was possible to extract the biophysical parameters from the catchment for input into the AGNPS model. Grid cells of 50 m by 50 m were used and simulation done on cell basis using GIS ILWIS which gave the spatial distribution of surface runoff volume, sediment yield and phosphorus yield in form of thematic maps. The results showed that both runoff volumes and sediment yield were reasonably simulated with efficiencies of 0.88 and 0.92 respectively while the phosphorus simulations did not match well with the observed values at the catchment outlet. The results indicate the high potential of AGNPS model for analyzing the dynamics and constituents of flow required in catchment management. The model however requires further verification and testing especially with regard to simulation of nutrients.

Key words: Phosphorus, Runoff, Sedimentation

Introduction

Soil erosion is the removal of surface material by wind and water. It is a three-phase process consisting of the detachment of individual particles from the soil mass and their transport by erosive agents such as running water and wind and the deposition of the sediments. The severity of erosion depends upon the quantity of the material supplied by detachment and the capacity of the eroding agent to transport it (Morgan, 1986). As the runoff travels fast, it carries soil materials away through the hydraulic force of its flow and when it gets in contact with the soil surface, it also picks up an appreciable amount of dissolved and adsorbed material (Kirkby & Morgan, 1980). Soil erosion is a major cause of deterioration of water resources as runoff picks up and transport sediments and pollutants resulting from human activities ultimately depositing them into water bodies. Agriculture has been identified as the leading source of non-point source pollution (Andrew, 1990). The agricultural non-point source pollution is diffuse in nature, has complicated spatial and temporal dimensions and is influenced by the weather pattern or rather hydrologic events. Its characteristics make it very difficult and expensive to monitor on a continuous and widespread basis utilizing field treatments and experiments. Nutrients from commercial fertilizers and animal manure are transported to the water bodies by runoff either as particulate bounded to the soil particles in connection with erosion or as dissolved nutrient (Rode & Lindenschmidt, 2000). Biophysical simulation models have been developed as an alternative to estimate the impacts of agriculture on water quality.

Lake Naivasha drainage basin has become a major area of interest for agriculture and in particular, horticultural farming for export and a center of tourist attraction since it has a variety of wildlife living around it. At present, there are conflicting demands for water from water users whereby agriculture and municipality are now being challenged by power generation, navigation, fishing, wildlife, tourism and recreation; each contestant exerting its influence on both quantity and quality. As the economic pressures become greater for more usable water, the interrelationships between quantity and quality become more clearly defined, but the solution for a particular problem area becomes more complex. The quality of water is influenced by usage, natural pollution, drainage of urban and agricultural lands, solid waste disposal and recreation activities and therefore depends on the amount of total suspended sediment, the chemical constituents and the micro-organisms (bacterial load) in the water (Schwab *et al.*, 1981). The fluctuating lake levels have always been of concern to the horticultural farmers who really want to know the reasons behind it. The question was whether the reduction in lake levels is due to huge abstractions by the lake water users, or was it because or abstractions from the rivers at the upper catchment, or is it due to sedimentation which increases the surface area for evaporation? Several researches were therefore carried out to investigate the cause of reductions in lake water levels and the results were documented in ITC journal. However, it is not just the quantity alone but also the quality of the lake water is important. Lake Naivasha is a fresh water lake that is also used for fish production. The quality of the lake water thus affects the fish productivity. Lake Naivasha has of late experienced an increase in sediment and nutrients inputs. The transported sediments into the lake in 1998 was estimated to be 142 tons/year (Hamududu, 1998) and the phosphorus loading per unit area of the lake surface to be 1.41 g/m² (Kitaka, 2000). These two inputs, which mainly originate from agricultural activities in the catchment, are very costly to monitor since they are non point source pollutants, yet they are the major cause of deterioration of water resources. Modeling is a technique that can be used to simulate the activities in a catchment to see how these activities would affect the water resources. It takes into account the system characteristics, manipulates activities to create possible scenarios and gives the effect of the activities on the system as output. In most cases, modeling is carried out on small area and the results extrapolated for larger areas. Having knowledge of what goes on in a catchment forms the basis of decision making in catchment management. This in turn would prevent problems since one has data and idea of what would happen in the catchment and can take preventive measures to conserve water resources. This is about preventing rather than solving problems. Solving catchment problems can prove to be very expensive and therefore investing in low cost preventive measures should be encouraged. In order to conserve water resources, the use of a model that can estimate sediment yield, runoff volume and phosphorus yield should be investigated.

This study is an attempt to assess the effect of agriculture on the quality of the lake water by use of the Agricultural Non-Point Source Pollution Model (AGNPS). AGNPS model in this case was tested and calibrated in Mirera sub-catchment for simulation of sediment yield, runoff volume and phosphorus yield. Mirera sub-catchment is an example of many agricultural areas in Kenya which have undergone major changes due to man's interferences. Although it does not have a rugged terrain, or steep slopes, the 1.82 km^2 area at the south east of L. Naivasha drainage basin experiences erosion as it is evident on the open fields where there are several rills. The erosion in this area is due to activities carried out in the area.

Materials and methods

Model approach

The Agricultural Non - Point Source Pollution (AGNPS) model simulates runoff water quality from agricultural watersheds. It was developed by the U.S. Agricultural Research Service (ARS) in cooperation with the Minnesota Pollution Control Agency and the U.S. Soil Conservation Service (SCS) (USDA-ARS). It can be used in watersheds ranging in size from a few hectares to 20,000 ha. Estimates of runoff water quality can be made with AGNPS to evaluate potential pollution problems for a watershed, and remedial measures can be recommended on the basis of an assessment of the effects of applying alternative management practices. Data representing these management alternatives can be used as program input, and the resulting watershed responses can be evaluated and compared (Young *et al.*, 1987).

AGNPS is an event-based model, which uses geographic data cells of 0.4 to 16 ha to represent land surface conditions. Within the cells, runoff characteristics and transport processes for sediment, nutrients, and chemical oxygen demand (COD) are simulated for each cell. Flows and pollutants are routed through the channel system to the basin outlet. Point source inputs (such as nutrient COD from animal feedlots) can also be simulated and combined with the non-point-source contributions (Young *et al.*, 1994).

Basic model components include hydrology, erosion, sediment transport, and chemical transport. In the hydrology component, runoff volume is calculated by the SCS curve number procedure. Peak flow rate is estimated using an empirical equation which takes into account drainage area, channel slope, runoff volume and watershed length-width ratio. Erosion is calculated from a modified form of Universal Soil Loss Equation. Soil loss is calculated for each cell of the watershed. Eroded soil and sediment yield are subdivided into particle size classes; sediment routing is based on the effective transport capacity of the stream channels. In the chemical transport part of the model the transport of nitrogen, phosphorus and COD is calculated throughout the watershed. Chemical transport calculations are divided into soluble and sediment-adsorbed phases. COD is assumed to be soluble and accumulate without losses (Young *et al.*, 1987).

In AGNPS applications, a uniform grid is placed over the watershed. Storm rainfall and runoff produce erosion through sheet and rill erosion processes. Soil loss and sediment yield is calculated for each cell, and the upland transport of sediment and nutrients is determined for each cell outlet. Calculations for AGNPS occur in three loops: initial calculations for all cells are made in the first loop, runoff volume is calculated for cells containing impoundments and sediment yields for cells that no other cells drain into are computed in the second loop, and sediments and nutrients are routed through the watershed in the third loop.

AGNPS model simulates the runoff volume using the SCS curve number method, sediment yield using the modified USLE and phosphorus yield using CREAMS model. The runoff equation is

$$Q = \frac{(P - 0.2S)^2}{P + 0.8S} \quad \text{(eqn. 1)}$$

where Q is the runoff volume, P is the rainfall and S is a retention parameter, all expressed in uniform dimensions of length. The retention parameter is defined in terms of a curve number (CN), as follows:

$$S = \frac{1000}{CN} - 10 \quad \text{(eqn. 2)}$$

The curve number (CN) depends upon land use, soil type and hydrologic soil condition (Young *et al.*, 1987). The erosion and sediment component is calculated by the Universal Soil Loss Equation (USLE), which is;

$$A = RKLSCP \quad \text{(eqn. 3)}$$

where A = annual soil loss

R = rainfall factor

K = soil erosion factor

L = field slope length factor

S = field slope factor

C = cover and management factor

P = supporting practice factor

However, nutrients transport is adopted from CREAMS, and the nutrient yield in the sediment adsorbed phase is calculated using total sediment yield from a cell as follows:

$$Nut_{sed} = (Nut_f)Q_s(x)E_r \quad \text{(eqn. 4)}$$

where Nut_{sed} is nitrogen or phosphorus transported by sediment, Nut_f is nitrogen or phosphorus content in the field soil, and E_r is the sediment enrichment ratio, calculated as follows:

$$E_r = 7.4Q_s(x)^{-0.2}T_f \quad \text{(eqn. 5)}$$

where $Q_s(x)$ is sediment yield and T_f is an adjustment to correct sediment adsorbed nutrient enrichment ratios for sand and clay soils (Young et at., 1987). Soluble nutrient estimates consider the effects of nutrient levels in rainfall, fertilization and leaching. Soluble nutrients contained in runoff are estimated as follows:

$$Nut_{sol} = C_{nut}Nut_{ext}Q \quad \text{(eqn. 6)}$$

where Nut_{sol} is the concentration of soluble nitrogen or phosphorus in the runoff, C_{nut} is the mean concentration of soluble nitrogen and phosphorus at the soil surface during runoff, and Q is the total runoff.

The model has been tested for accuracy of sediment yield estimations with data from experimental watersheds located near Treynor, Iowa and Hastings, Nebraska. Sediment yield compared favorably with measured values from the Treynor watersheds (Young, *et al.*, 1987). AGNPS has also been tested on Augucho catchment in Ethiopia for estimation of runoff volume and sediment yield (Nigussie & Fekadu, 2002). The results indicate that the estimated sediment yield from the model compared favorably with the measured values but the results for runoff volume were less satisfactory.

Calibration of AGNPS

AGNPS is a physically based model. The input data must be prepared for all cells of the watershed. Calibration was done by determining the input parameters as specific to the area of study. Parameters were determined from field survey and by use of prepared Tables of given factors as they vary with land cover and soil type and antecedent moisture condition. Antecedent Soil Moisture Condition II (AMC II) was adopted for the study area. This is the soil moisture condition before a given rainfall event. AMCII means the soil moisture is normal. The most suitable value for each input parameter was selected from Tables for each given land cover and their maps created. The data was fed into the model and used to estimate the soil loss, runoff volume and phosphorus content of the runoff.

Simulation of runoff volume, sediment and phosphorus yields using AGNPS

In this case, every input parameter is considered for each and every cell. This implies that it takes into account the spatial variation of the input parameters. For a cell with various land covers, an average value of the parameter was used in the final computation. The following procedure was the used: (1)The input data was displayed – contour, drainage and boundary maps (2) The input maps were converted to polygons and then to raster formats (3) Digital Elevation Model of the study area was created (4) Grid overlay and unique cell numbering (5) Aggregation of model inputs (6) Creating a linking table (7) Creating a slope length map (8) Creating a slope shape map (9) Creating a channel indicator table (10) Creating a channel length and gradient map (11) Determining the final unique cell map of the watershed (12) Creating maps for land cover, Soil, K-factor, C-factor, P-factor, Manning soil, SCS-CN number, SCC, and COD (13) Determining the channel indicator map (14) Redefining the input variables according to the final unique cell map (15)

Running the conversion program (16) Editing the flow direction to remove sinks and to have only one outlet (17) Running AGNPS (Mannaerts *et al.*, 1998).

As seen in the above procedure, manual simulation involved creating maps for every input parameter. The Figure 1 shows the K-factor map, which is one of the input parameters required by the model. The value changes with the cover type as shown.

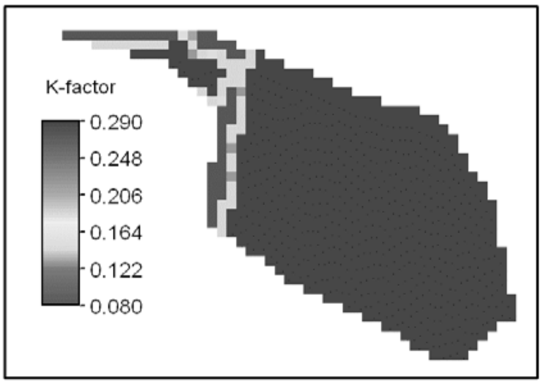


Figure 1: K-factor map for study area.

Step 16 in the procedure involved removing the sink holes from the study are. AGNPS does not recognize sink holes and when it finds any in the area; it gets confused and can not continue. Therefore to solve this problem, the flow direction must be checked to confirm that there are no sink holes and that there is only one outlet from the catchment. The sink holes are removed by giving the cell a flow direction such that the flow is continuous throughout the area. Figure 2 shows the flow direction, the arrow head gives the direction of flow. A sink hole is usually represented by a cross (✕) and must be removed before running the model.

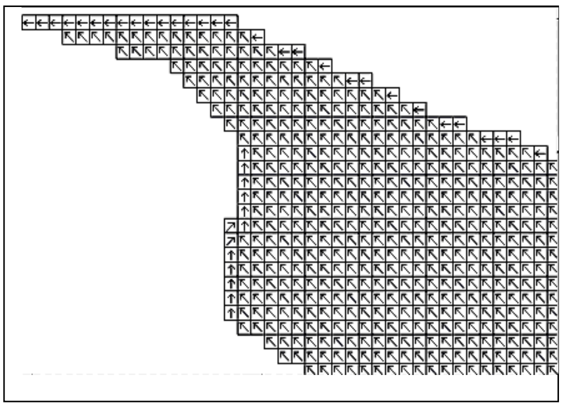


Figure 2: Flow Direction for the Study Area

Criteria for model evaluation

Evaluation of the model performance involved comparing the simulated and the observed results. This was done by visual observation through graphical displays and also statistical techniques. Different statistical methods have been outlined and tested for analysis of model performance. However, the choice of a statistical method depends on the objective of the study and the variable in question. In this case, the model was evaluated using percent error (%E), R^2, slope and Nash and Sutcliffe efficiency for runoff volume (Nash & Sutcliffe, 1970).

$$\%E = \frac{\bar{X}_o - \bar{X}_s}{\bar{X}_o} \times 100 \qquad \text{(eqn. 7)}$$

where *%E* is percent error, $\bar{X}_o$ is the sum of observed values and $\bar{X}_s$ is the sum of simulated values.

$$EFF = \frac{\sum_{i=1}^{m}\left(Q_{oi} - Q_{av}\right)^2 - \sum_{i=1}^{m}\left(Q_{oi} - Q_{si}\right)^2}{\sum_{i=1}^{m}\left(Q_{oi} - Q_{av}\right)^2} \qquad \text{(eqn. 8)}$$

where *EFF* is Nash and Sutcliffe efficiency, Q_{oi} is the measured values, Q_{si} is the simulated values and Q_{av} is the average of measured values, *m* is number of observations.

The smaller the percent error, the better is the model performance. For R^2 and gradient, a very good model performance is indicated when the values of R^2 and gradient are as close to 1 as possible. However, for Nash and Sutcliffe efficiency, a good model performance is indicated when its value is close to 100%

Results and discussion

The values in Table 1 below were adopted considering soil type, land cover and the antecedent moisture condition from prepared tables. Antecedent soil moisture condition two (AMCII) was adopted in this case because sampling was carried out during rainy season and the AMC were normal.

Table1: Parameters as determined for AGNPS model calibration.

Parameter	SCC	SCS	COD	K-factor	C-factor	Manning Roughness Coef.
Bare soil	0.05	82	0	0.08	0.55	0.055
Murrum Road	0.01	87	0	0.08	0.55	0.011
Vegetable	0.29	78	20	0.29	0.12	0.41
Grassland	0.22	86	80	0.29	0.066	0.04
Tarmack	0.01	98	0	0.08	0.55	0.011
Settlement	0.14	90	37	0.29	0.32	0.153
Homesteads	0.19	79	53	0.29	0.35	0.213
Fallow	0.05	79	0	0.08	0.55	0.055
Cropland	0.29	88	170	0.29	0.35	0.04

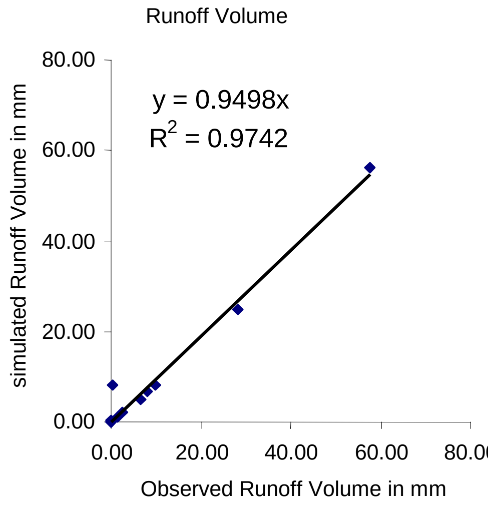
Table 2: Observed and simulated runoff volume in mm.

Date	**Runoff Vol. Obs (mm)**	**Runoff Vol. Sim (mm)**
07/04/04	0.06	0.00
08/04/04	0.01	0.00
10/04/04	9.85	8.13
11/04/04	28.40	24.89
18/04/04	2.74	2.03
25/04/04	0.18	0.25
29/04/04	1.29	1.02
01/05/04	6.48	5.08
31/05/04	0.27	8.13
15/06/04	8.21	6.60
Total	**57.49**	**56.13**

Runoff volume

The observed and simulated runoff volumes in mm are given in the table below. The results from visual inspection reveal a close simulation of the runoff volumes. After carrying out a statistical analysis, it is revealed that the simulated values of runoff volume were 97.6% of the observed values with a Nash and Sutcliffe Efficiency of 0.88. Similar results were observed in the Georgia Coastal Plain stream where the model simulation of runoff volume was 85% to 100% of the observed runoff volume (Suttles *et al.*, 1999). Also in the Piedmont region of Virginia (Mostaghini *et al.*, 1997), the predicted runoff volume was 100% of the observed runoff volume. In general, the model performance in runoff volume simulation was good. The model performance was also presented graphically as shown in Figure 3.

Figure (3a) shows that the gradient of the line is 0.95. This value is close to 1, which is an indication that runoff volume was well simulated by the model. The R^2 is 0.97, also approaching 1. The model performance in simulation of a given variable is rated as very good when the value for gradient and R^2 is as close to 1 as possible. It is therefore evident that the model results for runoff volume simulation are reasonable. Figure (3b) gives a quick indication of the model performance from a glance. The closer the plotted points are to the 45^0 line, the better the model performance. Also, one can easily see whether the model under or over predicted the values. This is determined by looking at the number of points below and above the line of equal values. If there are more points above the line, then it's an indication of an over-prediction, otherwise, it is an under-prediction. Hence, from Figure (3b), we can see that the model generally under-predicted the runoff volume. This could be as a result of the amount of rainfall that falls directly into the pond and also the runoff that comes from the tarmac road (South-Lake road), close to the pond. The pond surface area was estimated as 1941 m^2 and with high amounts of rainfall; the volume of water falling there is significant. These sources cannot be assumed to be negligible as their contributions increases with increase in rainfall amount.



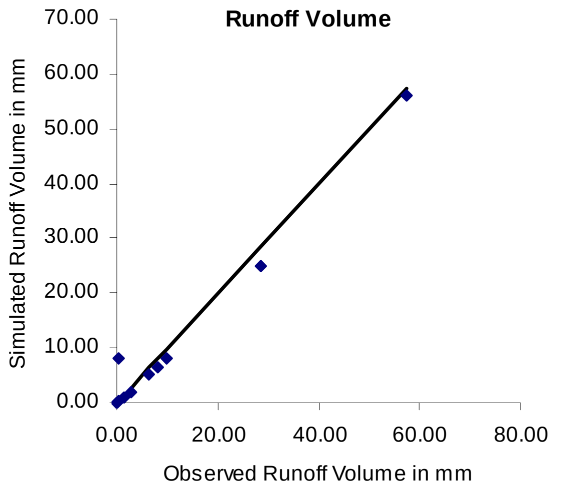


Figure 3: Observed (upper, 3a) and simulated (lower, 3b) runoff volumes in Mirera sub-catchment .

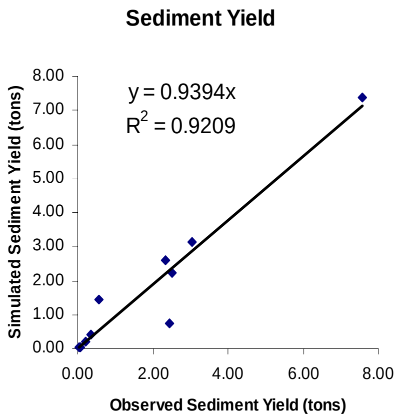
3.2 Sediment Yield

The observed and the simulated sediment yields are as given in Table 3. The model simulated values of sediment yield were 94.8% of the observed values. This is an indication of good model performance. Similar results were obtained in Piedmont region of Virginia using AGNPS (Mostaghini *et al.*, 1997); the simulated values were 88% of the observed values. However, in Augucho catchment in Ethiopia (Nigussie & Fekadu, 2002), the simulated values of sediment yield were 105.3% of the observed values. An over estimation of sediment yield was also recorded in central German catchments (Rode & Frede, 1997). These results don't deviate significantly from the one carried out in Treynor, Iowa (USDA-ARS, 1970) where the sediment yield estimates from the AGNPS model compared favourably with the measured values from the Treynor watersheds. A comparison of the observed and predicted sediment yields showed that the model over predicted by 2.3% with an efficiency of 0.95. In Illinois (Lee, 1987), the simulated and observed values of sediment yield were well represented when compared with observed values. The percent over-prediction and under-prediction in all the cases is however within acceptable limits and thus an indication of good performance by the model.

Table 3: Observed and simulated sediment yields in tons.

Date	Sed Yield Obs.(tons)	Sed Yield Sim (tons)
07/04/04	0.07	0.05
08/04/04	0.05	0.05
10/04/04	2.51	2.24
11/04/04	7.58	7.37
18/04/04	2.46	0.73
25/04/04	0.20	0.19
29/04/04	0.35	0.40
01/05/04	0.57	1.43
31/05/04	3.04	3.14
15/06/04	2.34	2.58
Total	19.17	18.18

The model performance was also analyzed graphically. This is presented in Figure 4.



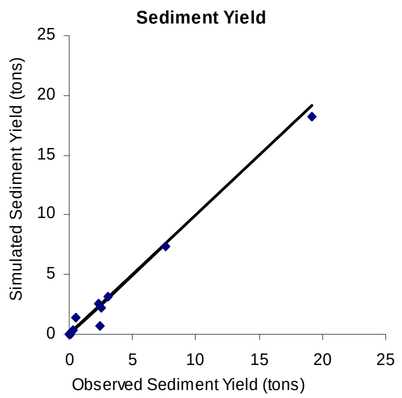


Figure 4: Observed (upper, 4a) and simulated (lower, 4b) sediment yields in Mirera sub-catchment.

Figure (4a) shows a gradient of 0.94 and R^2 of 0.92 (values close to 1), an indication of good performance of the model. Figure (4b) assesses the model performance in each simulated event. The Figure portrays an average in under and over-predicted events. It is however noticeable that the simulated and observed values vary significantly on 18th April 2004. Since the simulations for all other events were satisfactorily done by the model, the results for 18th April 2004 could be due to several factors. First, it is also possible that the measurements were not accurately done. The measurement of the sediment involved sampling runoff at a given time interval. It is possible that the time interval in the sampling for this event was so close such that in getting the mean sediment concentration, a higher value was used. The sediment yield is influenced by the transport capacity of the runoff. The runoff volume also changes with intensity; therefore we expect a change in sediment concentration with time. If the rainfall intensity variation within an event is not properly represented in the sampling, one is likely to get an error. This error when translated to the overall sediment yield from the catchment may give wrong values. The culvert on the south-lake road had just been cleared of the deposited sediment that was blocking the runoff flow. This could have been the source of the excess sediment as the loose sediment on the culvert floor was washed into the pond by the runoff. Also, the soil was generally dry. The area has poor grassland and also the main route for the animals to the lake. The trampling of the animals feet might have dislodged a significant amount of soil and with the high rainfall intensity for that rainfall event, caused high sediment yield. The effect of the animals trampling is clearly indicated in the murrum road, towards the outlet, where we record significantly high amounts of sediment yield.

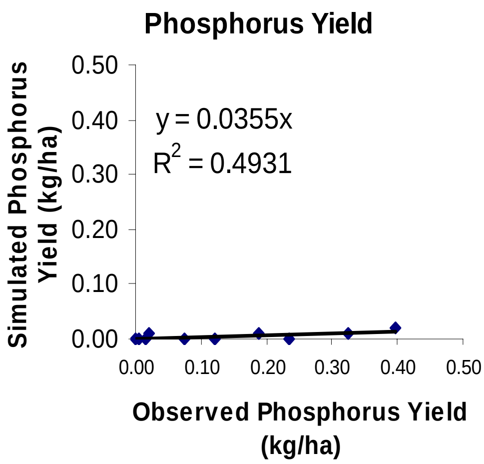
3.3 Phosphorus Yield

Phosphorus yield was simulated for ten rainfall events and the results compared with the observed values for the same rainfall events. The results are shown in Table 4 below.

Table 4: Observed and simulated phosphorus yields in kg/ha.

Date	TP Obs. (kg/ha)	TP Sim. (kg/ha)
07/04/04	0.01	0.00
08/04/04	0.00	0.00
10/04/04	0.33	0.01
11/04/04	0.40	0.02
18/04/04	0.12	0.00
25/04/04	0.02	0.00
29/04/04	0.07	0.00
01/05/04	0.23	0.00
31/05/04	0.02	0.01
15/06/04	0.19	0.01
Total	**1.38**	**0.05**

Simulation of phosphorus yield was very poor with a percent error of 96.4%. This means that the simulated values were 3.6% of the observed values. An under-prediction by the model was also experienced in the Piedmont region of Virginia, the model under predicted phosphorus yields by 41% (Mostaghini *et al.*, 1997). These results however vary significantly with researches from other regions where the model tends to over predict the phosphorus yield. Figure 5 is a graphical presentation of the simulated and observed phosphorus yields.



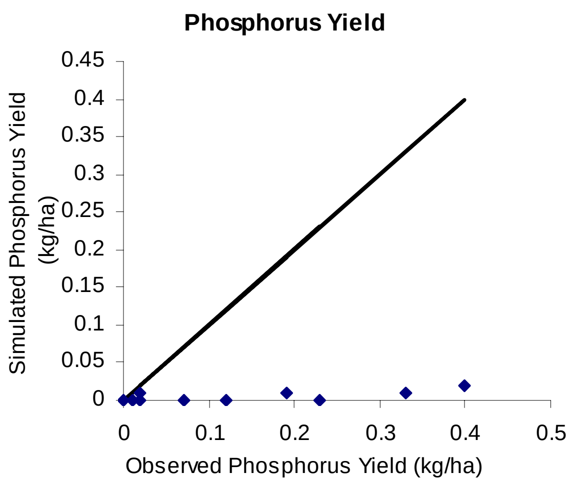


Figure 5: Observed and simulated phosphorus yields for Mirera sub-catchment.

Figure (5a) gives the R^2 and the gradient for the phosphorus yield simulation. The gradient is 0.04, a very low value implying a very poor performance of the model. The model under predicted phosphorus yield throughout as shown in the Figure (5b). It is only in the first two events and the second last event when the model simulation was very close to the observed values, and on the second event, the simulation is exact. However, the measured high concentration of phosphorus in the runoff raises a lot of questions. Analysis of soil samples from the area indicated lower concentration of phosphorus except for samples collected on the animal's track and near the settlement area. This was also revealed by the model as shown in Figure 6.

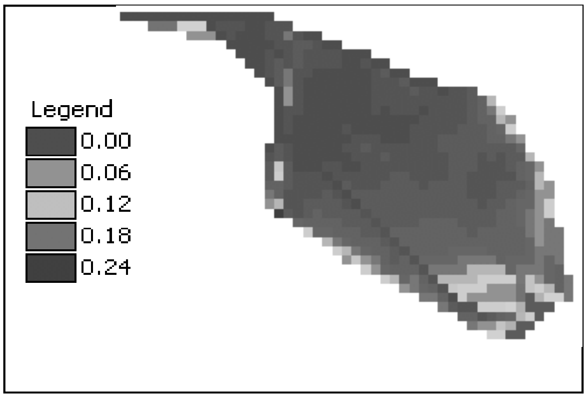


Figure 6: Soluble phosphorus yield in lbs/acre distribution for 15/06/2004.

High soluble phosphorus yield in this case is recorded in cropland and the borders between the homesteads and the grassland. Cow dung is no doubt a good source of phosphorus just as soap. However, the distribution as indicated in this map could not be justified.

Figure 7 below shows the distribution of sediment phosphorus in the study area for the same event as for Figure 6. There is a variation on the amount of soluble and sediment phosphorus for an individual cell. The analysis that was carried out in determining total phosphorus yield did not look at the soluble and sediment phosphorus separately, but rather as total phosphorus.

We can however justify the high discrepancy between the observed and the simulated values with the following possible factors: The catchment has the main route to the lake where animals from the catchment and outside the catchment follow as they go to water in the lake. As the animals move, they leave a lot of wastes on the path and also on the tarmac airstrip. The dung is also seen in the grassland. Animal waste (manure) has been found to be a good source of phosphorus. This could be the reason for high phosphorus concentration in the runoff. Also, the congested settlement in the catchment may be another source of high phosphorus concentration since there is no sewage system and the people pour out the wastewater from washing (grey water) on the ground. Detergents contain good amounts of soluble phosphorus. The human waste thrown on the pathways and also behind buildings during rainy seasons could also contribute to the high phosphorus concentration.

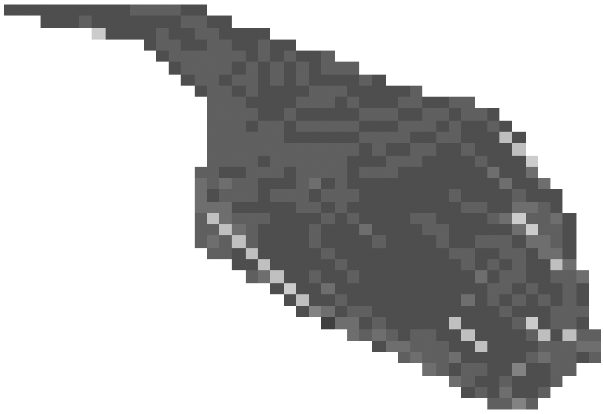


Figure 7: Sediment phosphorus yield in lbs/acre for 15/06/2004 rainfall event.

The horticultural farms at the catchment boundary could also contribute to the high phosphorus concentration as runoff during very high rainfall intensity, overflows into the catchment. There is a farm at the boundary of the study area near the catchment outlet that carries out outdoor farming. Runoff flowing over this farm carries both dissolved and adsorbed phosphorus from the fertilizers applied in the farm. It is therefore evident that the major sources of phosphorus in the study area are dynamic external sources.

References

Andrew G. 1990. The Human Impact on the Natural Environment. The MIT Press. Cambridge, Massachusetts.

Hamududu H.H. 1998. Erosion Assessment for Large Basins Using Remote Sensing and GIS: A Case Study of Lake Naivasha Basin, Kenya. MSc. Thesis.

Kirkby M.J., Morgan R.P.C. 1980. Soil Erosion. John Wiley & Sons. New York.

Kitaka, N. 2000. Phosphorus Supply to a Shallow Tropical Lake and its Consequences - Lake Naivasha, Kenya. PhD Thesis.

Morgan R.P.C. 1986. Soil Erosion and its Control. Van Nastrand Reinhold Company. New York.

Nash, J.E. and Sutcliffe, J.V. 1970. Riverflow Forcasting through Conceptual Models Part I-a Discussion Principles. Journal of Hydrology. 10:282-290.

Nigussie, H., Fekadu, Y., 2002. Testing and Evaluation of the Agricultural Non-Point Source Pollution Model (AGNPS) on Augucho Catchment, Western Hararghe, Ethiopia. Agriculture Ecosystems & Environment 99 (2003) pp 201-212.

Rode, M. and Lindenschmidt. 2000. Distributed Sediment and Phosphorus Transport Modelling on a Medium Sized Catchment in Central GermanyElsevier Science Journal Vol. 26, No. 7-8, pp. 635-640.

Schwab, G.O., Frevert, R.K., Edminster, T.W., Barnes, K.K. 1981. Soil and Water Conservation Engineering. Third

edition. John Wiley & Sons. USA.
Young , R.A., Onstand, C.A., Bosch, D.D., Anderson, W.P. 1987. AGNPS. Agricultural Non-point Source Pollution Model, A Watershed Analysis Tool. USDA, Conservation Research Report 35.
Young, R.A., Onstand, C.A., Bosch, D.D., Anderson, W.P. 1994. Agricultural Non-point Source Pollution Model. Version 5.00. AGNPS User's Guide. North Central Conservation Research Laboratory. USDA, ARS. Morris, Minnesota, USA.

Quick evaluation of lake water toxicity by the measurement of ingested fluorescent materials by *Daphnia magna*

Minori Kamaya and Kunio Nagashima

Kogakuin University, Faculty of Engineering, Dept. of Applied Chemistry
2665-1 Nakano-cho, Hachioji-shi, Tokyo 192-0015, Japan
E-mail: bt74290@ns.kogakuin.ac.jp

Abstract

The amount of food ingested by Daphnia magna decreases when it is exposed to a toxic material. The food items used were chlorella and yeast labeled with 5-DTF, a fluorescent substance. Daphnia magna were exposed to a toxic material for 30 minutes and then transferred to a solution containing the fluorescent materials for a 30-min period of ingestion. The Daphnia magna were then removed, homogenized, and subjected to fluorescence spectrophotometry to measure fluorescence. The amount of ingestion was determined from the intensity of fluorescence. An advantage of this method is that it can evaluate toxicity in about an hour, in contrast to the 24 hour requirement of the previously adopted acute immobilization test. This method has also been applied to evaluate the toxicity of lake water.

Key words: *Daphnia magna,* toxicity evaluation

Introduction

Bioassays using living organisms are often employed to survey the effects of chemical substances on ecological systems. Due to its ease of culture, *Daphnia magna* is widely used as a bioassay organism and standardized methods for its use have been adopted (JIS K 022, 1992). These methods include observations on rates of immobilization (JIS K 0229, 1992), quantity of motion (Aoyama *et al.,* 1997), heart rate (Presing & Vero 1983), and food ingestion (Lee *et al.,* 1997; Bitton *et al.,* 1996; Bitton *et al.,* 1995; Juchelka & Snell, 1995) However, there are drawbacks to many of these methods, for example dependence on visual inspection; analytical systems for measuring quantity of motion; data acquisition using fluorescent microscopy ; lengthy testing periods; expensive equipment; and, finally, the difficulty in obtaining accurate toxicity data due to irradiation stress incurred during *Daphnia magna* observation . We have determined that changes in *chlorella* injested by *Daphnia magna* can be measured by changes in absorbance (Kamaya *et al.,* 2001) or intensity of fluorescence (Kamaya *et al.,* 2004). However, these methods are relatively time-consuming since food concentrations changed slowly in test containers and a quick evaluation of toxicity was not possible. In the current study, a basic examination of toxicity based on ingestion was performed. Fluorescence-labeled chlorella and yeast were used as food. *Daphnia magna* were homogenized after ingestion and fluorescence measured to evaluate toxicity.

Materials and methods

Mechanism of ingestion by *Daphnia magna*

Ingestion mechanism

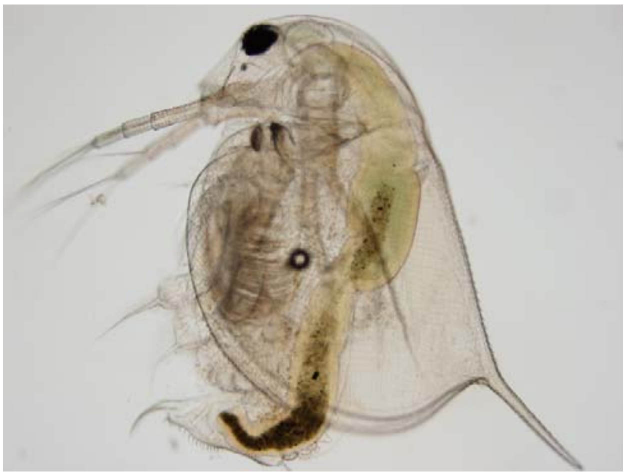


Figure 1: Photograph of *Daphnia magna*.

Three possible mechanisms for food ingestion in *Daphnia magna* have been proposed. The most commonly accepted mechanism is based on filtration, in which *Daphnia magna* ingests food items by filtration through the filtration hairs, spaced about 1 m apart, on the third and fourth peraeopods (Porter *et al.,*, 1983). However, since 0.2 m particles were retained on the filtration hairs, electrostatic action is likely an additional factor in the filtration mechanism. (Gerristen & Porter, 1982). A second theory (DeMott, 1986) proposed a positive ingestion theory. Although a positive association between feeding and food items with odors has been identified in several different kinds of zooplankters, this has not been confirmed with *Daphnia magna*. A recent study of the ingestion mechanism observed with a high-speed video camera has been reported. In this study, food movement was actually observed and the commonly accepted filtration theory was refuted. Although cladocerans such as *Daphnia magna* have often been called “filter feeders”, this designation was refuted by the video study. The second, third, and fourth peraeopods of *Daphnia magna* were found not to serve as sieves but instead functioned similarly to pump blades and created flow for food ingestion by irregularly phased piston movement (10-15 Hz). Thus, ingestion was found to depend on a mechanism of physical adsorption of food particles (Tanaka, 2004).

Ingestion method

We had previously evaluated toxicity by observing inhibition of chlorella ingestion (Kamaya *et al.,* 2001, 2004) in the presence of both chlorella and toxic substances. However, the adsorption of the toxic substances onto chlorella was not taken into account in this method. It is known that chlorella adsorbs heavy metal ions and the co-occurrence of chlorella and heavy metal ions leads to higher toxicity than with heavy metal ions alone (Taylor *et al.,* 1998). In addition, adsorption of metal ions to such solid materials as clay is also known to affect toxicity (Weltens *et al.,* 2000). Therefore, we judged it undesirable to perform an ingestion test in the presence of both particulate materials and toxic components. Hence, the current study examines two aspects: *Daphnia magna* exposure to a toxic substance and food ingestion.

Daphnia magna culture and apparatus

We cultured *Daphnia magna* and fed them Chlorella V12 by Chlorella Industry Co., Ltd., which was kept refrigerated and used within a month of purchase. Culture water was Elendt M4 culture water (Kamaya 2004) within an incubator, MIR-253 by Sanyo Electric Co., Ltd. Water was changed with fresh water and chlorella was fed every three days. Daphnia magna were homogenized with an Azuwan homogenizer-223A in a 1.5-mL micro test tube. Fluorescence measurements were taken with a F-3010 spectrophotometer by Hitachi Ltd.

Fluorescence-labeling of food

Fluorescence-labeling of chlorella

We followed the methods of Reiter (1997) for fluorescence-labeling of chlorella with 5-DTAF, as shown in Figure 2. Chlorella micropowder was used (Chlorella Industry Co., Ltd.). The micropowder was used without further purification; 0.5 g was added to an aqueous solution (pH, 8.5) of 5-DTAF, 5 mg/L, and the liquid was then left standing at room temperature for 20 hours. 5-DTAF is known as a labeling reagent for amino compounds (Blau & Halket ,1996). It binds with the cell wall in chlorella because the cell wall contains 20% of all proteins within the cell (Berliner 1986). The fluorescence-labeled chlorella was dried and then shaken with the culture water, centrifuged at 3000 rpm for five minutes, and washed repeatedly until intensity of fluorescence (excitation wave length, 460 nm; fluorescence wave length, 530 nm) was constant in the supernatants.

Fluorescence-labeling of yeast

Baker's yeast was used and labeled according to the methods of Lee *et al.,* (1997). Glucosamine, which is a component of yeast cell wall, is presumed to be the labeled component (Kollar *et al.,* 1995). The yeast was labeled by adding 0.5 g of the dry yeast to 50 mL of a 0.05 M Na_2HPO_4-0.85% NaCl solution. To this yeast suspension, 1 mL of 1% (w/v) 5-DTAF dimethylsulfoxide solution was added, and the liquid was left standing at 60˚C for two hours. After adding purified water, fluorescence-labeled yeast and the liquid was shaken vigorously, and then centrifuged at 3000 rpm for five minutes and repeatedly washed until intensity of fluorescence became constant in the supernatants. The labeled yeast was then dried and used.

Removal of solution during transfer of *Daphnia magna*

A small amount of the solution is transferred along with Daphnia magna when they are dispensed with a pipette, a possible source of error. To minimize solution transfer Konig *et al.,* (1981) placed a fluororesin fiber sheet (pore diameter, 70 µm) on dry filter paper, transferred Daphnia magna onto the sheet with a pipette, removed excess solution, and then transferred Daphnia magna with a stainless steel spatula. In the present study, a 42-mesh stainless steel net was used. The liquid containing Daphnia magna was passed through the mesh, the Daphnia magna washed with water, and the water droplets removed with absorbent paper.

Standard test procedures for toxicity evaluation

A toxic substance was diluted to five different concentrations with Elendt M4 culture water. Ten mL of the diluted solutions was dispensed into a glass container and 10 individuals of Daphnia magna (<24 hours old) were transferred to the container and exposed to the toxic substance for 30 minutes. They were then transferred to containers containing the fluorescence-labeled chlorella or yeast, and left for 30 minutes. The Daphna magna were then filtered through the stainless steel mesh, transferred to a micro test tube with 1 mL of water, and homogenized. After homogenization, the liquid was transferred to a 5-mL graduated flask and diluted to 5 ml with purified water. This diluted liquid was measured for fluorescence intensity at the excitation wavelength of 460 nm and fluorescence wavelength of 530 nm.

Results and discussion

Calibration curves for fluorescence-labeled yeast and chlorella

To examine the influence of the precipitate and flocculation of fluorescence-labeled chlorella and yeast, liquids of the fluorescent materials themselves were used to prepare calibration curves. Although the fluorescence-labeled materials were washed, 5-DTAF may not have been completely removed by washing. Hence, the liquids obtained for preparation of the calibration curve were filtered through a 0.45 m membrane filter to examine non-reacted 5-DTAF by determining fluorescence intensity of the filtrates. The results are shown in Figs. 3 and 4. The figures show there was a lesser amount of non-reacted 5-DTAF with the fluorescence-labeled chlorella, but the calibration curve was non-linear, whereas there was more non-reacted 5-DTAF with the fluorescence-labeled yeast with greater linearity.

Ingestion time and food concentration

The optimal time for ingestion by Daphnia magna was studied by altering the concentrations of the fluorescence-labeled chlorella and yeast stepwise from 50 to 150 mg/L. The results are shown in Figs. 5 and 6. The fluorescence-labeled chlorella did not yield either good reproducibility or constant fluorescence intensity. In contrast, as shown in Figure 6, the fluorescence-labeled yeast yielded constant fluorescence intensity beginning at around 20 minutes. Next, ingestion time was fixed at 30 minutes and the concentrations of the fluorescence-labeled materials were altered. The effects are shown in Figs. 7 and 8. The fluorescence-labeled chlorella tended to slightly increase in fluorescence intensity even at higher concentrations and was observed to precipitate as the concentration approached 150 mg/L. Conversely, the fluorescence-labeled yeast showed maximum fluorescence intensity near 75 mg/L and decreasing fluorescence intensity at greater concentrations. Depression of feeding rate was recognized higher critical concentration, this phenomenon has been described previously (McMahon & Rigler 1963). In addition, the ingestion time of about 20 minutes for fluorescence intensity to reach a plateau closely agrees with passage time for liquid through the intestine (Lavrent'yeva & Beym 1979). It further suggests that the fluorescent yeast was being digested as it passed through the gut, however 5-DTAF, a chemically modifying substance, was not changed by ingestion process in gut. The above results show that the fluorescence-labeled yeast was suitable as food, and thus a concentration of 75 mg/L and feeding time of 30 minutes were adopted.

Evaluation of detergents for toxicity

We attempted to evaluate the toxicity of a detergent in this study. Solutions of the detergent (containing 25% straight-chain sodium alkylbenzene sulfonate) were prepared at appropriate concentrations using aged tap water. These solutions were used to evaluate toxicity using fluorescent yeast in accordance with procedure 4 (above). Results are shown in Figure 9. The data reveal that a large amount of the fluorescence-labeled yeast was taken into the intestine at low detergent concentrations, but intake sharply decreased after the peak at 100 mg/L. The reason for the increased ingestion at lower concentrations was a result of increased motion of Daphnia magna stimulated by the toxic substance.

Conclusion

Daphnia magna were fed with fluorescence-labeled chlorella and yeast and the amount injested was determined for use in toxicity evaluation. Fluorescence-labeled yeast was suitable as a food and an ingestion time of 30 minutes was sufficient. In addition, the effects of differing detergent concentrations were studied to confirm that the method described in this study could be used for a quick evaluation of toxicity.

Acknowledgement

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References

Aoyama, I. Ruo, R. & Okamura, H., 1997. Monitoring of the hazardous chemical substances by the momentum analysis of Daphnia magna. *KankyoGijutu*, 26, 255-259.

Berliner, M. D., 1986. Protein in chlorella vulgaris, *Microbios.,* 46, 199-203.

Bitton, G., Rhodes, K., Koopman, B. & Cornejo, M., 1995. Short-term toxicity assay based on daphnid feeding behavior. *Wat.Enviorn.Res.,* 67, 290-293.

Bitton,G., Rhodes, K. & Koopman, B., 1996. An Acute toxicity test based on ceriodaphnia dubia feeding behavior. *Environ.Toxicol.Chem.,*15, 123-125.

Blau, K. & M.Halket J. / Translated by Nakamura, H., 1996. *Handbook of derivatives for chromatography.* pp.188 Maruzen,Tokyo.

DeMott, W. R., 1986. The role of taste in food selection by freshwater zooplankton, *Oecologia*. 69, 334-340.

Gerristen, J. & Porter, K.G., 1982. The role of surface chemistry in filter feeding by zooplankton. *Science*, 216, 1225-1227.
JIS, K 0229, 1992. Testing methods for determination of the inhibition of mobility of Daphnia by chemicals. *Japanese Industrial Standard,* Tokyo.
Juchelka, C.M. & Snell, T. W., 1995. Rapid toxicity assessment using ingestion rate of cladocerans and ciliates. *Arch.Environ.Contam.Toxicol.,* 28, 508-512.
Kamaya, M., Tanabe, J. & Nagashima, K., 2001. Toxic assesment by feeding rate of Daphnia magna. *Yousui to Haisui,* 43, 116-120.
Kamaya, M., 2004. Development of toxicity evaluation method with Daphnia magna using air stripping concentration. *Yousui to Haisui* 46, 3-8.
Kamaya, M., Tanabe, J. & Nagashima, K., 2004. A Rapid toxicity test based on inhibition of feeding rate with Daphnia magna. *Yousui to Haisui,* 46, 943-947.
Kollar, R., Petrakova, E., Ashwell, G., Robbins, P.W. & Cabib, E., 1995. Architecture of the yeast cell wall. *J. Biol.Chem.,* 270, 1170-1178.
Konig, P. & Muller, H-G., 1981. A rapid easy method to introduce Daphnia magna into test vessels. *Bull. Environm*. *Contam. Toxicol.,* 26, 22-23.
Lavrent'yeva, I.F. & Beym, A. M., 1979. Study of physiological process in daphnia using fluorochromes, *Hydrobiol.J.,* 14, 84-87.
Lee, Sang-III, Na, Eun-Jung, Cho, Young-Ok, Koopman, B. & Bitton, B., 1997. Short-term toxicity test based on algal uptake by ceriodaphnia dubia. *Wat.Environ.Res.,* 69, 1207-1210.
McMahon, J.W. & Rigler, F. H., 1963. Mechanisms regulating the feeding rate of daphnia magna straus. *Can.J.Zool.,* 41, 321-332.
Porter, K.G., Feig, Y.S. & Vetter, E.F. (1983) Morphology, flow regimes, and filtering rates of daphnia, ceriodaphnia, and bosmina fed natural bacteria. *Oecologia,* 58, 156-163.
Presing, M. & Vero, M., 1983. A new method for determining the heart beat rate of daphnia magna. *Water Res.,* 17, 1245-1248.
Reiter, M.A., 1997. A simple fluorescent labeling technique for marking of algae in mixed assemblages. *J.Freshwat. Ecol.,* 12, 163-165.
Tanaka, Y. (2004) Direct obserbation on zooplankton feeding mechanisms. *Japan Plankton Gakkaiho,* 51, 151-154.
Taylor, G., Baird, D.J. & Soares, A. M. V. M., 1998 Surface binding of contaminants by algae: Consequences for lethal toxicity and feeding to daphnia magna straus. *Environ.Toxicol.Chem.,* 17, 412-419.
Weltens, R., Goossens, R. & Puymbroeck, S.V., 2000. Ecotoxicity of contaminated suspended solids for filter feeders. *Arch.Environ.Contam.Toxicol.,* 39, 315-323.

Filter-feeding function and respiratory properties of a bivalve called Mashijimi (*Corbicula leana* in fresh water) in Lake Tega basin area in Japan

Go Konishi[1], Nobumasa Kitamura[1], Yosuke Yamashiki[2], Hitomi Matsushima[2], Kazuhito Murakami[3], Kazuo Taki[3], Norio Hayashi[4] and Takeo Mutoh[5]

[1]Grad. Student, Coll. of Sci.& Tech., Nihon Univ.,1-8 Kanda-Surugadai, Chiyoda-ku, Tokyo 101-8308 Japan
[2]CE Dept., Coll. of Sci. & Tech., Nihon Univ., 1-8 Kanda-Surugadai, Chiyoda, Tokyo 101-8308 Japan
[3]Life Sci. Dept., Eng., Chiba Inst. of Tech., Tsudanuma, Narashino, Chiba 275-8588 Japan
[4]Natural History Museum and Inst. Chiba, 955-2 Aobacho, Chuo, Chiba 260-8682 Japan
[5]Original Eng. Const., Co., Ltd., 1-1 ShinOgawa, Shinjyuku, Tokyo162-0814 Japan
[2]Email : hitomi@civil.cst.nihon-u.ac.jp

Abstract

A bivalve called Mashijimi in Japanese and classified into *Corbicula leana* in fresh water had been a native species at Lake Tega. It is located about 25 kms northeast from Tokyo and is featured by 650 has water area with the average depth of 0.86 m; and is known as one of the most eutrophicated lakes in Japan because of the inflow of the domestic and agricultural effluents from its basin area. Although Mashijimis do not live in the lake presently, they are still alive in the several small rivers inflowing into the lake and have an excellent filter-feeding function to remove algae suspending as organic matters. Mashijimi's physiological filter-feeding performance may be applied for removing the turbidity due to algae and thus for detoxifying the toxin like microcystin produced by blue green algae such as *Microcystis aeruginosa*, *M. wesenbergii* and *M. viridis*. Batch experiments were conducted with both open and closed systems to evaluate the Mashijimi's filter-feeding function and respiratory properties. In a closed system under dark condition to control photosynthesis reaction, the Mashijimi's filtering rate was estimated to be in between 0.13 and 1.23 liters/(g-wet bivalve-day), where the values were significantly affected by and proportional with the initial turbidity loading intensity in NTU (Nephelometric Turbidity Unit). However, if the initial NTU loading intensity becomes higher than 60 NTU-liter per unit wet-weight of a bivalve, the Mashijimi's filtering rate declines sharply. The Mashijimi's respiratory property as its oxygen consumption rate was determined to be 1.45～15.8 mg-O_2/(g-wet bivalve-day) for exogenous respiration, being proportional to the initial NTU loading intensity, and 0.04～0.20 mg-O_2/(g-wet bivalve-day) for endogenous respiration. Thus, a bivalve of Mashijimi like *Corbicula leana* in fresh water would be applied not only for removing algae, but also possibly for detoxifying microcystin.

Key words: *Corbicula leana*, filter-feeding, NTU loading intensity

Research background and objectives

Lake Tega located in the northwestern part of the Chiba Prefecture, Japan, is a fresh water lake, and has been the water frontal area for the residents' life activities including boating, fishing, irrigation and so on. At Lake Tega, however, the nutrient concentrations such as nitrogen and phosphorus have gradually rose mainly because of the inflow of domestic and agricultural effluents from the basin area due to the rapid development during the last 30 years. Consequently, the eutrophication phenomenon called water blooms has occurred by the excess growth of phytoplankton on the lake surface. Water blooms are mainly blue green algae, some of which have been known to produce a nerve and liver toxin like microcystin, and have caused serious problems in regard to the water-resources usage.

The present research work is related with the water quality preservation of eutrophicated lakes by using a native bivalve that is called Mashijimi in Japanese and classified into *Corbicula leana* in fresh water. Although the Mashijimis are not living in Lake Tega at present, but are still alive in the several small rivers inflowing into the lake, they have an excellent filter-feeding function for phytoplankton. The major objectives are to disclose the properties on the Mashijimi's predatory on blue-green algae suspending as organic matters and to examine the Mashijimi's filtering performance. Batch experiments in open system under light conditions resulted in obtaining the Mashijimi's excellent filtering capacity to clarify the eutrophicated lake water. Therefore, a series of the batch experiments in closed system under dark conditions to control a photosynthesis reaction and air entraining was carried out to evaluate the Mashijimi's filter-feeding performance and respiratory properties.

Open batch experiments under light conditions

Let each of the four 10 L glass-vessels contain 9 liters of the lake surface water, and then several pieces of the Mashijimis collected at a small river in the lake basin area were put into each vessel to adjust the Mashijimi's population density (PD in pieces/m^2) at PD 0 (the reference for control), PD 140, PD 350 and PD 700. The population density, PD, means the number of the living pieces of Mashijimi per unit area. Fluorescent light was being irradiated for the open system batch experiments to keep light conditions, while no light was irradiated for the closed system batch experiments in the following.

The Mashijimis collected for the experiments were being immersed into a deionized water for 24 hours before starting the experiments in order to excrete the wastes of the Mashijimi's living body. The average wet-weight of the Mashijimi for the experiments was about 1.65 g/piece. The water temperature in the glass-vessels was about 22 to 24 ℃ during the experiment period. A plankton net was used to collect the blue-green algae at Lake Tega, which was condensed to adjust the initial turbidity at about 30 degrees in Nephelometric Turbidity Unit (NTU) measured by a Model 2100P by HACH Company, Ltd. Each vessel was mixed slowly and thoroughly using a magnetic stirrer for the concentration of the suspending organic matters in the vessel to be uniform and in suspension. The results of the daily NTU turbidity removal in the open system experiments showed that the NTU turbidity removal rate becomes higher as the Mashijimi's population density becomes larger, which means the excellent improvement of the water quality in terms of the water clarity. This implies that the Mashijimi holds an excellent filtering capacity for the eutrophicated water. As for the control vessel in which a Mashijimi does not exist, the water clarity was getting worse day by day since the phytoplankton increased under the light conditions through photosynthesis reaction.

Experimental procedures for closed batch system under dark conditions

In order to control the growth of blue green algae under light conditions, a series of the experiments was conducted using a closed batch system under dark conditions. Simultaneously, the initial NTU turbidity loading intensity to the Mashijimis, defined as the value of the total NTU turbidity per unit wet-weight of the Mashijimi in the experimental vessel, was also examined about its influence on the Mashijimi's filtering and respiratory rates. The outline of the experimental apparatus for the closed batch system is as shown in Figure 1.

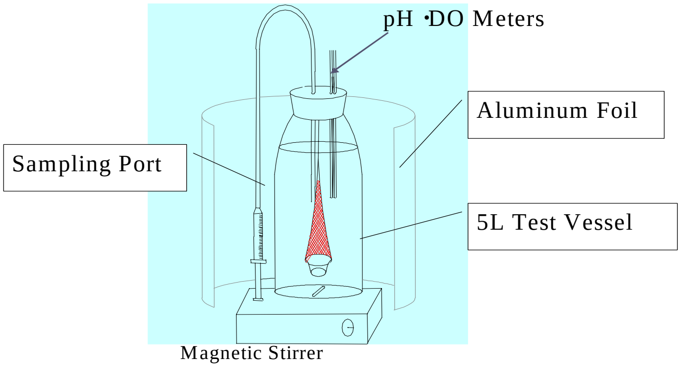

Figure 1 : Test vessel for closed batch system experiments.

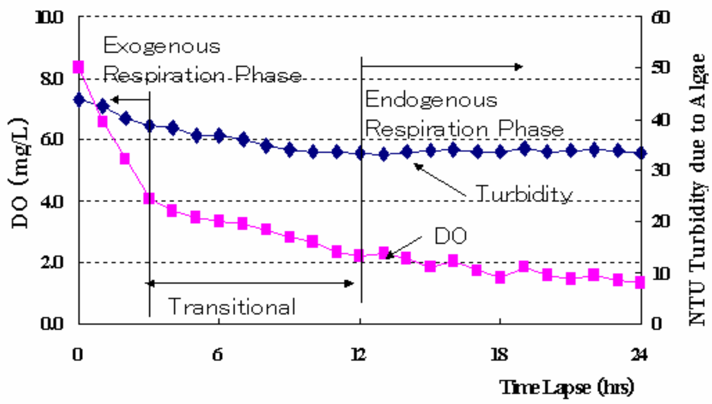

Figure 2 : Daily NTU turbidity and DO changes with time at P.D.700 for Mashijimi's filtering and respiratory rates.

The experimental apparatus, as in Figure 1, consists of four cylindrical glass vessels with a capacity of 5.0 liters. Each vessel with both a pH meter and a DO sensor is completely covered by a sheet of aluminum foil to control the photosynthetic reaction and is closed by a rubber cap to prevent air from entraining. Daily and timely NTU turbidity measurements were implemented similarly as in the open system experiments.

Relationship between Mashijimi's filter-feeding rate and its respiratory rate

From the profile of the DO (dissolved oxygen) concentrations changes in the closed system under dark conditions, the algae's respiratory rate will be estimated, and thus the Mashijimi's respiratory rates will be measured as well. The DO concentration profile decreases smoothly soon after pouring the sampled lake water containing algae into the test vessel, while the DO concentrations decrease sharply immediately after putting the Mashijimis into the test vessel. According to the DO change results of the phytoplankton in Lake Tega, the respiratory rate per NTU degree due to the algae was about 6.2×10^{-2} mg-O_2/(NTU degree-day). The DO and NTU turbidity changes at PD 700 are shown in Figure 2. Starting the batch experiments, the Mashijimis began to show their exogenous respiration phase, and then 10 hours later after starting the experiments the DO concentration profile showed a stable level to shift to the endogenous respiration phase from the exogenous respiration phase. As long as the Mashijimis are in the exogenous respiration phase, the NTU turbidity decreases sharply to show the Mashijimi's excellent filter-feeding function. Such a tendency at PD 700 at the exogenous respiration phase for several hours after starting was observed as well at PD 140 and PD 350. The Mashijimi's respiration phase, however, was considered to be at endogenous phase because of little change of the DO concentration, in which phase the filter-fed organic matters like blue green algae through the Mashijimi's filter-feeding function might have been digested, and probably some portion of the toxin of microcystin could have been detoxified physiologically.

The Mashijimi's filtering rate was defined to be the after-24-hours-filterate corresponding to the NTU turbidity removal rate, which is closely affected by the Mashijimi's population density. The NTU turbidity removal rates at various PD values are given in Figure 3. The NTU turbidity removal rate after 24 hours was observed to range at about 80 to 90 %, and the corresponding filtering rates per unit wet-weight of Mashijimi was about 0.13～1.23 liters/(g-wet bivalve-day), implying that the higher the initial NTU loading intensity, the larger the filtering rate of the Mashijimi. The Mashijimi's oxygen consumption rates as the respiratory rate were estimated to be ranging at about 1.45～15.8 mg-O_2/(g-wet bivalve-day) for the exogenous respiration phase and at about 0.04～0.20 mg-O_2/(g-wet bivalve-day) for the endogenous respiration phase. This is shown in Figure 4.

The influence of the initial NTU turbidity loading intensity on the Mashijimi's filtering rate is given in Figure 5, showing that the Mashijimi's filtering rate is directly proportional to the initial NTU turbidity loading intensity. However, the filtering rate decreases sharply when the initial NTU turbidity loading intensity becomes higher than about 60 NTU degrees-liters/(g-wet bivalve). Where, no effective filtering function was observed there.

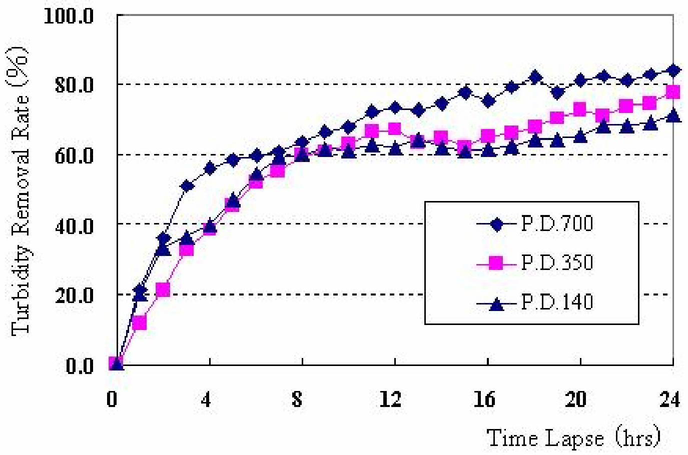

Figure 3 : Turbidity removal rate (%) with time lapse (hrs.).

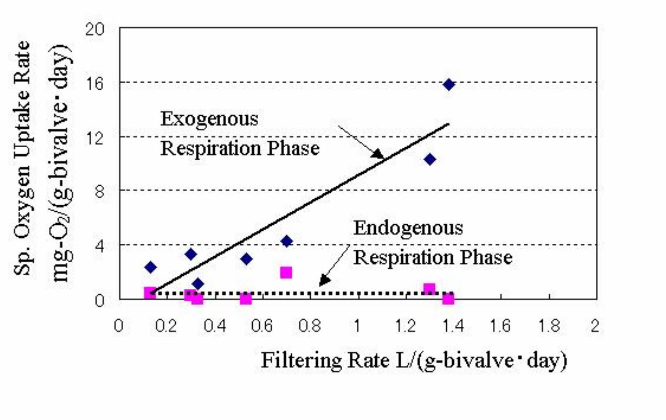

Figure 4 : Effect of filtering rate on Mashijimi's specific oxygen consumption rate.

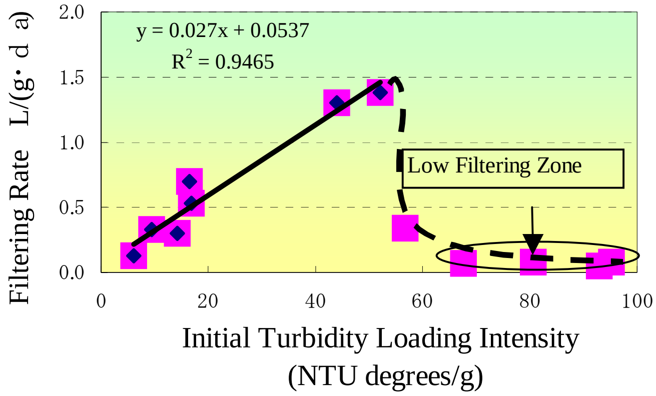

Figure 5 : Effects of the initial NTU loading intensity on Mashijimi's filtering rate.

Conclusions

Using the closed batch system under dark conditions, a native bivalve called Mashijimi in Lake Tega basin area, classified into *Corbicula leana* in fresh water, has been examined and discussed as above in terms of its filtering function for algae suspending as organic matters and its respiratory properties. Accordingly, the Mashijimi showed an excellent filtering capacity to clarify the eutrophicated lake water containing algae. Thus, the Mashijimi's filtering function is ecologically important for the natural water purification system. Some important results are summarized as in the following.

(1) Mashijimi showed clearly its filter-feeding function to remove the NTU turbidity due to phytoplankton to clarify the lake water with a good transparency. In addition, the pH level became to neutralize and maintain at a constant and stable level. Mashijimi's filtering rate was clearly affected by the initial NTU turbidity loading intensity, and was estimated to be about 0.13 to 1.23 liters/(g-wet bivalve-day), implying that the higher the initial NTU turbidity loading intensity, the larger the filtering rate of the Mashijimi. When it is higher than 60 NTU degrees-liters/(g-wet-bivalve), however, the filtering rate became to decline sharply.

(2) As far as the initial NTU turbidity loading intensity is relatively lower than 60 NTU degrees-liters/(g-wet bivalve), there was observed a clear relationship among Mashijimi's filtering rate, respiratory rate and the initial NTU turbidity loading intensity. The Mashijimi's oxygen consumption rates were estimated to be about 1.45～15.8 mg-O_2/(g-wet bivalve-day) for the exogenous respiration phase and about 0.04～0.20 mg-O_2/(g-wet bivalve-day) for the endogenous respiration phase.

Further experiments with a continuous flow system are now being conducted under light-dark conditions for applying the Mashijimis for eutrophicated lakes. Hyper-eutrophication of freshwater lakes and reservoirs are not only the obstacle for water supplies due to blue green algae causing water-blooms, but also the production of the toxin like microcystin from *Microcystis aeruginosa*; the toxicity of which is extremely high to living animals and even to human bodies. When eutrophicated and polluted by such a toxic substance, thus, the water usage becomes limited as for the drinking water resources. Therefore, it is important how to remove and detoxify the microcystin. Further experimental efforts are also now being continued on the effective detoxification using a bivalve of Mashijimi through its filter-feeding function.

References

T. Hoshi, H.Matsushima, *et al.*, "Filtering Property of a Bivalve of Mashijimi for Removing Suspending Organic Solids in Eutrophicated Lakes", ILEC 9th Int. Conf., Session Ⅳ, pp.228～231, Oct. 2001

M. Mori and H.Matsushima, "Filtering and Respiratory Properties of a Bivalve of Mashijimi", JSWE 37th Annual Meeting, Mar. 2002 (in Japanese)

R. Nakamura and H.Matsushima, "Mashijimi's Filtering Property on Suspending Organics using a continuous flow system", JSWE 39th Annual Meeting, Mar. 2004 (in Japanese)

K. Misono, H. Matsushima, *et al.*, "Detoxification of microcystin using the filter-feeding function of *Corbicula leana* in fresh water", IAGLR '05 Conference Abstract Book, p.131, May 2005.