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Review

Potential climate effects on nitrogen eco-toxicology of freshwater Lake, Victoria

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Lake Victoria has experienced changes that include introduction of alien species, over exploitation of fish, eutrophication and climate change. This review is a scenario of nitrogen cycle acceleration resulting in retention above the total inflow and the potential effect of climate on the N cycle. Excess nitrogen is attributed to nitrogen fixation, algal proliferation and decomposition. The nitrogen transformation like ammonia conversion to nitrate is enhanced over horizontal distance at higher temperature during dry season. Nitrogen (N) concentration in the vertical profile is related to climate variability of water temperature, lake water movement and differences in nitrogen loads from the catchment. Despite all, the effect of eddy currents or heat transfer caused by solar radiation on nitrogen processes is unknown. However, annual cycle of vertical oxygen distribution caused by stratification seems to provide potential condition for nitrous oxide production throughout the lake as compared to nitrogen gas. Therefore, understanding the relationships between organisms' diversity and community structure particularly of autotrophic and heterotrophic nitrogen bacteria, and their ecosystem functions in the entire freshwater lake is important for nitrogen budget. Due to scarcity of information, it is not possible to ascertain projection of climate influence on N dynamics in the Lake Victoria ecosystem. The ultimate suggestions on mitigation measures are to enforce policies that reduce both point and non-point sources of N into the lake and maintain riparian forests and wetlands.

Key words: Climate-effects, freshwater, Lake Victoria, nitrogen-ecotoxicology.

INTRODUCTION

Although Lake Victoria supports one of the largest fisheries, it has experienced changes that include introduction of alien species, over exploitation of fish, eutrophication and climate change (Hecky, 1993; Verschuren et al., 2002). Climate change is expected in an increased global temperature (IPCC, 2007). Already global mean surface temperatures have risen by 0.74°C over the past 100 years (1906 - 2005) and the warming rate of the last 50 years is almost double that over the previous 100 years (Trenberth et al., 2007). Africa countries in equatorial African region are warming at a slightly slower rate of about 1.4°C with respect to the 1961-1990 average (IPCC, 2001). Precipitation is also simulated to increase over Africa by 2050 (Hudson and Jones, 2002). Further, ozone depletion occurring over the latitudes that include much of Africa has potential effect on biogeochemical cycles such as alteration of sources and sinks of greenhouse gases and ozone (http://www.epa.gov/ozone/science/sc_fact.html). All these changes are likely to affect aquatic systems in Africa in complex ways (Lovejoy and Hannah, 2005). The changes may influence the water quality and biological processes through runoff from the catchment and hydrology processes within the lake system.

The global nitrogen cycle could come under increasing pressure, not only from direct anthropogenic perturbations but also from the consequences of climate change (Gruber and Galloway, 2008). In addition, the interactions of nitrogen with carbon and how these interact with the climate system is less emphasized (Falkowski et al., 2000), despite the climate change effects that may have started in tropical ecosystems including aquatic systems (Wandiga, 2003). The human impact on the dynamics is yet unknown. However, general responses to supply of nutrients and changing climate have been reported (Regier et al., 1990; Benke, 1993; Lovejoy and Hannah, 2005).

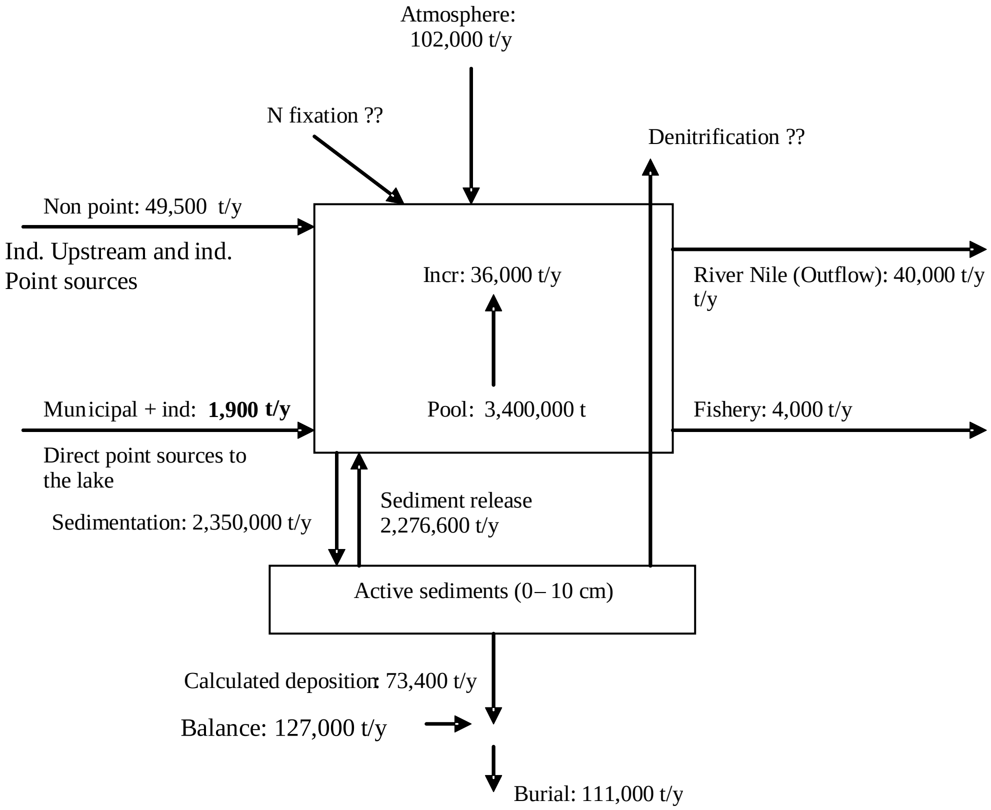

Figure 1. Nitrogen mass balance for Lake Victoria (Kayombo and Jorgensen, 2006).

The incorporation of ecosystem functions, as the rates of certain biological processes into regular monitoring program is becoming important (Bunn, 1995; Young, 2007). Many studies are conducted on linkages between climatic variables and effects on socio-economic status (Opera et al., 2007), sustainable use of natural resources (Niang et al., 2007) and biogeochemical cycles are now becoming of interest.

Nitrogen cycle acceleration affects the environment through the eutrophication of terrestrial and aquatic systems, and global acidification (Gruber and Galloway, 2008). In Lake Victoria, there are different catchment characteristics and there is uneven temporal and spatial nitrogen loads and distribution in the Lake system (Rutagemwa et al., 2005; Kayombo and Jorgensen, 2006; Pascal et al., 2007). Overall TN pollution loading (t/y) of the riparian countries (Uganda, Kenya and Tanzania) considering 50% load reduction by other treatment systems before discharge into the environment is 767 (21.88%), 2,019 (57.60%) and 719 (20.5) for urban wastewater and runoff, and 33 (7.97%), 57 (13.77%) and 324 (78.26) from industrial activities respectively (Kayombo and Jorgensen, 2006). However, the effect of nitrogen loads (inflow) into the Lake over a short period of time may not be visible (Biswas, 1976); although models indicate nutrients dispersion within the system (Banadda et al., 2011; Bongomin and Opio, Unpublished). The concentration of nitrogen compounds in aquatic systems are derived from allochthonous and autochthonous sources, while, in the system, nitrogen is either retained or released. The sources and the amount of nitrogen retained in freshwater, Lake Victoria are indicated in Figure 1.

The summary of the total nitrogen budget of the lake excludes nitrogen fixation and denitrification. The atmospheric deposition (102,000 t/y) is expected to increase with increasing nitrogen compounds in the atmosphere. Millennium Ecosystem Assessment (2005) estimated an increase of total reative nitrogen deposition from the atmosphere into the Lake Victoria surface water region to a threshold between 1000 - 2000 mg N m^{-2} y^{-1} in 2050. Nitrogen removed by fish harvesting is estimated at 4, 000 t/y. A total of 73,400 t/y of nitrogen which is above the total input (51, 400 t/y) is deposited in the sediment. This is attributed to nitrogen fixation, algal proliferation and decomposition (Bugenyi and Balirwa, 1998) and also explains the relatively higher concentrations of inorganic nitrogen (IN) (0.477mg l^{-1}) and dissolved organic nitrogen (DON) (0.406 mg l^{-1}) at the bottom of the lake (Pascal et al., 2007). The high values are contribution of accretion process (0.131 g m^{-3} d^{-1}) of nitrogen transformation or settling of nitrogen into the sediments (Pascal et al., 2007).

Accretion in the lake has not been quantified into the varied contribution of the dead organisms (i.e. plants, algae, plankton and bacteria) and wastes from the fishes. However, it depicts nitrogen ‘top-down’ effects. Mugidde (1993) however reported ‘bottom-up’ effects which is a result of eutrophication dynamics.

The inversion and prolific production of water hyacinth in Lake Victoria attributed to lack of natural enemies, ample space, optimal temperature and abundant nutrients (Opande et al., 2004) and caused significant changes in N dynamics of the lake. Water hyacinth is generally mobile except in lagoons and beaches that have little external interferences from wind actions. Open water is always clear of water hyacinth due to the frequent wave actions. The impact of water hyacinth on reducing fisheries harvests has been reported (Kateregga and Sterner, 2009). In addition, decomposition of the sunken water hyacinth cause prolong depression of dissolved oxygen to even anoxic levels close to the lake bottom and increase diversity and abundance of phytoplankton, macro invertebrates and fishes. Increasing infestation by water hyacinth is correlated with both *Chironomid* and mollusk densities (Bugenyi and Balirwa, 1998). Succession of the plant caused the disappearance of other free-floating macrophytes like *Pistia stratiotes* while providing substrate for emergent *Vossia cuspidate*. The weed is believed to have led to extinction of *Azolla nilotica* in the lake. This means a major change in the contribution of N retention and stock in the different floral and faunal components of the lake system.

The total organic nitrogen released into the lake undergoes a hydrolytic reaction, producing ammonia which provides a food source for the nitrifying bacteria that converts ammonia ($\mathrm{NH_3}$) to nitrite ($\mathrm{NO_2^-}$) and then nitrate ($\mathrm{NO_3^-}$) (Equations 1 and 2).

$$\mathrm{2NH_3^+ + 3O_2 \longrightarrow 2NO_2^- + 2H^+ + 2H_2O} \quad (1)$$

$$\mathrm{2NO_2^- + O_2 \longrightarrow 2NO_3^-} \quad (2)$$

The process of nitrification is mediated by *Nitrosomonas* (1) and *Nitrobacters* (2) that produce nitrite and nitrate respectively. Nitrification is however inhibited by free ammonia, nitrous acid and nitrite at low pH (Anthonisen et al., 1976; Metcalf and Eddy, 2003; Arceivala and Asolekar, 2008). Despite the significance of these processes in Lake Victoria, only values for Tanzania section of the lake is available (Pascal et al., 2007).

Although nitrification reduces toxicity of ammonia and contributes to biological oxygen demand (BOD), in Lake Victoria, more conversion of ammonia to nitrate in the horizontal distance occurs during dry season due to increased temperature, and at this time there is virtually no non-point allochthonous organic input (Banadda et al., 2011). The same study also reported similar concentrations for ammonia, nitrites, and nitrates at various vertical depths of the lake section in Gabba area in Uganda in the rainy season. However, an increasing concentration in the vertical profile (depth) was reported in the Tanzania section of the lake (Pascal et al., 2007). The differences in the N concentrations could be a result of the depth and accretion process at the different sites. In addition, shielded bays tend to have less mixing unless there is strong seiche from the open waters that cause dilution of the nitrogen compounds (Larsson et al., 2008). Moreover, nitrogen is introduced into the lake during the wet season due to non-point sources of pollution such as atmospheric deposition; precipitation and land run off through rivers from agricultural activities (Kayombo and Jorgensen, 2006; Pascal et al., 2007; Banadda et al., 2011). Therefore, the reportedly 10 - 20% increase in runoff as a result of climate change for most of Uganda (MLWE, 2002) is likely to affect nutrient inflow into the lake and their dispersion within the lake system.

Denitrification process within aquatic systems lowers nitrogen concentration (Equations 3 and 4).

$$\mathrm{2NO_2^- \longrightarrow N_2 + 2O_2} \quad (3)$$

$$\mathrm{2NO_3^- \longrightarrow N_2 + 3O_2} \quad (4)$$

In Lake Victoria, denitrification estimation has been reported within the general range of 250 - 500 kg N km^{-2} y^{-1} (Seitzinger et al., 2006). The daily average value for denitrification in the Tanzania section of the lake is presented in Figure 2. These processes affect the overall nitrogen budget estimation in the lake system.

In aquatic systems, aerobic nitrification may be followed by anaerobic denitrification and for that reason; there is high potential of nitrous oxide production as compared to nitrogen gas (Takaya et al., 2003). Such conditions may accumulate nitrous oxide ($\mathrm{N_2O - N_i}$) in the lake although this has never been estimated. Nitrogen budget of the lake indicates that there is a lot of nitrogen accumulating in the lake bottom through accretion (Kayombo and Jorgensen, 2006; Pascal et al., 2007). The nitrogen would be lost or locked up in the sediment if it were not the ‘bottom-up’ effect reported by Mugidde (1993). The general process affect the total nitrogen (TN) and total phosphorus (TP) ratio (Van Ginkel, 2002) with ultimate reflection in the type, distribution and abundance of plant species in the lake. TN:TP ratio of greater than 10:1 favors green algae, which may be less problematic to manage while any ratio less than 10:1 encourages *Cyanobacteria* that is able to fix atmospheric nitrogen. Cyanobacterial blooms have been reported in the inshore of Lake Victoria and is attributed to light as the limiting factor since dissolved organic matter (DOM) attenuate most of the light in the

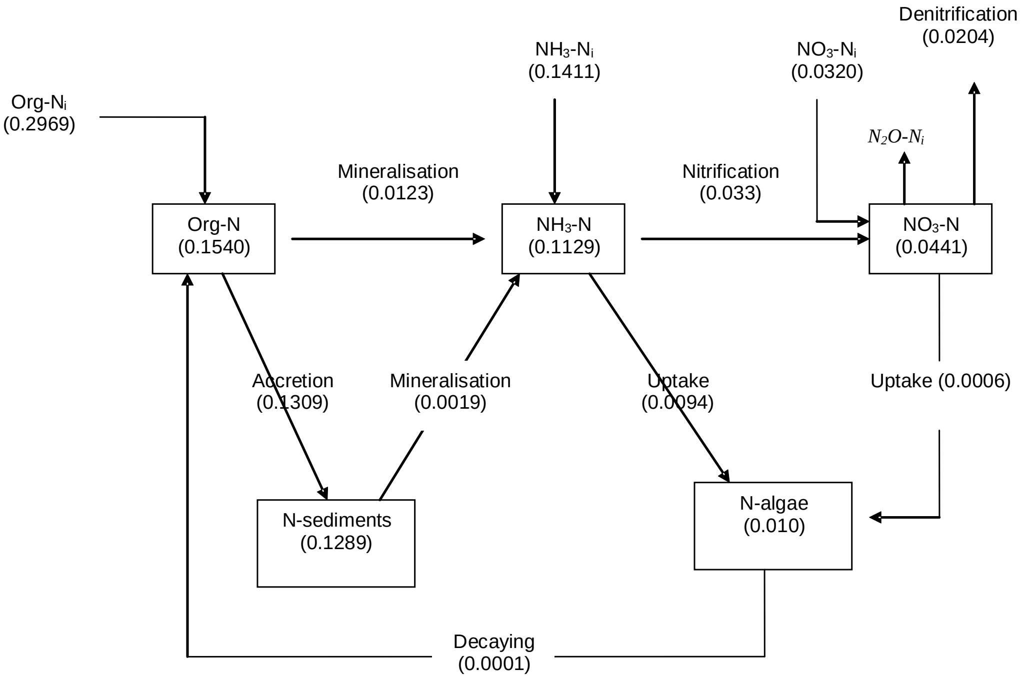

Figure 2. Nitrogen mass balance for Tanzania section of Lake Victoria (Pascal et al., 2007). All values are in g m^{-3} d^{-1}.

inshore zone (Verschuren et al., 2002). This also corresponds with the positive correlation of nitrogen bacteria to nitrate-N concentration in aquatic systems (Chia and Bako, 2008). TN:TP ratios and especially DIN : SRP ratios in Lake Victoria decrease with the wetland presence along the coastline, showing a higher probability of N limitation in the inshore waters where large wetlands are present (Co´zar et al., 2007). The results therefore points to denitrification processes in the wetland ecotones as the cause of this trend.

Solar radiation effects on nitrogen related factors

The visible solar radiation range is important in aquatic productivity. The shorter wavelength of solar radiation (ultraviolet (UV)) has potential effects on aquatic biogeochemical cycles. The general effect of solar radiation in Lake Victoria waters is the buildup of thermal differences resulting in stratification of the waters. In addition, the radiation triggers off conventional movement of water that transfer heat from the southeastern area to the northeastern colder region of the lake (Song et al., 2004). The wave of heat transfer is thought to vary on the intra-seasonal, inter-annual, inter-decadal and palaeo-climate time scales causing eddy currents. Therefore, the role of the eddy currents in the transport of energy and momentum may impact on the spatial and temporal nitrogen processes. The role of eddy currents could be similar to the water movement in the great lakes or tides in marine environment that organizes the structure and function of the ecosystems (Keough et al., 1999) particularly the nitrifying and denitrifying bacteria which are sensitive to temperature changes.

Solar radiation intensity within water column is strongly related to water quality and therefore vary from inshore to the open waters (Dattilo et al., 2001). General increase in UV radiation in aquatic systems has been reported (Gies et al., 2004; Mckenzie et al., 2003) corresponding with stratospheric ozone depletion (Solomon, 2004). The increasing intensity affects aquatic food chain (Häder et al., 2007). Many factors influence the depth of penetration of radiation into natural waters i.e. dissolved organic compounds (Dattilo et al., 2001; Häder et al., 2007) which are likely to be influenced by future climate change. Owing to the high input of inorganic and organic decaying materials, coupled with the high level of eutrophication, Lake Victoria ecosystems has a enormous potential of UV absorption (Loiselle et al., 2001; Bracchini et al., 2004). The inshore area has high phytoplankton compared to open water zones and is attributed to nutrients load and less stable water column stratification in the inshore areas (Loiselle et al., 2001). Meanwhile the open water zone is characterized by atmospheric nitrogen deposition and seasonal mixing. In addition, variation of DOM from the fringing wetlands directly influences phytoplankton biomass and productivity (Loiselle et al., 2001). The effect is usually enhanced during the wet season as a result of flushing out of DOM that reduces the deleterious effects of UV radiation. Changes in phytoplankton composition and species diversity in the lake therefore impact on the nitrogen processes particularly the fishery and sediment retention.

Exposure to solar UV radiation has been shown to affect both orientation mechanisms and motility in phytoplankton, resulting in reduced survival rates (http://www.epa.gov/ozone/science/effects/index.html).

The effect with other stress factors has been studied in bacteria and *Cyanobacteria* and other primary producers. Excessive visible radiation, non-optimal temperature, toxic heavy metal, and changes in salinity synergistically increase the inhibitory effects of UV on growth, reproduction, ecosystem structure and food dynamics (UNEP, 2000). Lake Victoria catchment is dominated by industrial activities and the chances of heavy metals increasing into the lake cannot be ruled out. This notwithstanding, a number of new compounds absorbing UV have been identified in *Cyanobacteria*, phytoplankton and macroalgae and their role as photoprotectants during evolution is now recognized (UNEP, 2000). The effect of such changes on nitrogen processes in the open Lake Victoria waters is unknown. Analysis of the inshore areas indicates protection/attenuation from high irradiances of UV compared to open waters (Dattilo et al., 2001).

Thermal conditions and nitrogen changes in Lake Victoria

Water temperature usually falls with water depth, because less sunlight penetrates the water at a greater depth. However, temperature change attributed to climate variation in Lake Victoria has been reported (Bugenyi and Magumba, 1996; Wandiga, 2003) and thermal gradients weakness over the last decade in the Lake water column is also documented (Marshall et al., 2009). The sensitivity of fresh water Lakes to climate warming over a narrow range of high water temperature have been reported (Ndebele-Murisa et al., 2010). Therefore, moderate climate warming may destabilize nitrogen cycles and nutrients distribution (Spigel, and Coulter, 1996), as well as occurrence of nitrogen fixation, nitrifying and denitrifying bacteria. For example in Nyanza area of Winam gulf in Lake Victoria, high abundance of nitrogen fixing (*Cylindriospermopsis africana*) and potentially toxic (*Anabaena sporoides*) species during uniformly mixed period is attributed to low availability of dissolved inorganic nitrogen (Gikuma-Njuru et al., 2011). The mixed condition results into uniform oxygen distribution in lower depths; thereby promoting conversion of low oxidized forms of nitrogen to nitrate that is easily assimilated into organic growth. Mathematical models have also indicated the role played by the assumption of uniform mixing factor in nitrification process. The series model for nitrogen dispersion revealed shorter distance for ammonia-N and nitrite-N, and longer distance for nitrate-N (Bongomin and Opio, Unpublished). The information obtained by using principle of conservation of mass modeling indicated greater distance for all the nitrogen compounds (Banadda et al., 2011). In the later modeling result, there is assumption of uniform mixing and the results from the models also show the effect of thermal stratification on the nitrification process.

Like any other aquatic systems, the general organic nitrogen decrease in Lake Victoria agrees with first order kinetics (Kadlec and Knight, 1996) and temperature impact on the processes like decomposition (Webster and Benfield, 1986). Oxygen consumption and dissolution rate during decomposition depends on temperature (Boyd, 1998). In this regards, nitrogen loading from organic substances is expected to increase with rising temperature range. Ammonia fraction also dominates over ammonium at high temperature values (Boyd, 1998). Therefore, in eutrophic freshwater lakes such as Victoria, daily temperature rise increasing pH value to more than nine (9), shifts the balance of the total ammonia towards un-ionized ammonia (Balirwa, 1998).

Extreme pH values are detrimental to many micro-organisms that influence nitrogen cycle. High pH values may imply high NH_4^+ ion concentration (Boyd, 1998). Research using molecular analysis have also revealed novel nitrifier sub-strains of *Nitrosospira briensis* (3b cluster) being tolerant to high NH_4^+ concentration while activities of *Nitrosospira NpAV* (3a cluster) strain are inhibited at high concentration (Webster et al., 2005). However, the pH ranges suitable for denitrifier growth and optimal ammonification process is 7 - 8 and 6.5 - 8.5 respectively and usually low pH increase the generation of N_2O and NO (Princic et al., 1998).

The survival of aquatic also organisms depend on temperature and warm waters are naturally productive with species that flourish being harmful sometimes (Poff et al., 2002). For example ‘nuisance’ blooms of algae that occur in warm and nutrient rich conditions are expected to increase in frequency. Large fish predators that require cool water temperatures may also be lost resulting into more blooms of ‘nuisance’ algae, reduced water quality and pose potential health problems. However, in some cases, i.e. the abundance of nitrogen fixing bacteria such as *Dactyllococcpsis* and *Gomphosphaeria* species are reported to be negatively correlated with temperature (Chia and Bakia, 2008). Optimal nitrification also occurs at the range of 25 - 35°C while temperature range suitable for denitrifies is estimated between 5 °C to 25°C (Kadlec and Knight, 1996). Changes in thermal gradients in the Lake Victoria water column were reported (Marshall et al., 2009) and the effect on spatial and temporal nitrogen processes is yet unknown.

In addition, ammonia and DO concentrations, water mixing and light attenuation are known to influence nitrification process in water hyacinth systems (Webster and Tchnobanoglous, 1985; Todd and Josephson, 1996). However, nitrification process is most sensitive to DO concentration and temperature only affects the rates of physiological processes and vitality of bacterial attachment sites in the water hyacinth (Webster and Tchnobanoglous, 1986). Therefore, nitrification in poorly oxygenated water hyacinth system is limited by oxygen transport through the plant. Elevated temperature only affects nitrification rates when it increases or decreases oxygen transport capacity.

Dissolved oxygen concentration and nitrogen processes

Oxygen flux into aquatic system is driven by mass transfer from the atmospheric sources constituting the first order process (Kadlec and Knight, 1996) and during autotrophic production of aquatic plants. In Lake Victoria, the annual cycle of oxygen vertical distribution in 1990 - 1991 was compared with data of 1960 - 1961 (Hecky et al., 1994). The results showed high oxygen concentration in the mixed layer for 1990 - 1991 with nearly continuous super-saturation in surface waters. The hypolimnetic waters had lower concentration for a longer period; the values were < 1 mg/l at 40 m compared to the shallowest occurrences of > 50 m in 1961. The dissolved oxygen (DO) trend was attributed to increased nutrient loading, altered climate and food web changes (Verschuren et al., 2002). In such a situation of high BOD levels, growth of heterotrophic bacteria that competes with nitrifying bacteria is favored (Tiedje, 1988). DO concentration of less than 0.50 mg l^{-1} is thought to limit nitrification (Metcalf and Eddy, 2003). Low DO has been reported in the lake sediment area since 1950s (Talling, 1957, Hecky et al., 1994). The general condition seems to have worsened with increased eutrophication. However, lack of complete mixing and redistribution of oxygen enhance anoxic denitrification in the sediment layer of aquatic systems (Mortimer et al., 2004), as oxygen acts as a better electron acceptor and there is high amount of energy produced when DO is used. For that matter, decreased DO favors denitrification process since the only source of oxygen is that bound to the nitrate or nitrite. Breaking of nitrogen-oxygen strong bonds in nitrate requires high amount of energy, which the denitrifiers tend to avoid (Kadlec and Knight, 1996). In this regards, denitrification process in bulk water of Lake Victoria will cease under high oxygen condition but the process continues to occur in the microscopic anoxic zones, for instance, at the sediments particularly when the overlying water column is highly productive. Laws (2000) reported denitrification occurring when oxygen concentration drops to below 0.2 gm^{-3}. Aerobic denitrifiers (*Paracoccus denitrificans, Microvirgula aerodenitrificans, Thaurea mechernichesis* and *P. denitrificans)* reduce nitrate even at oxygen saturation levels (Loyd et al., 1987; Robertson and Kuenen, 1990; Takaya et al., 2003). Whether the rate of denitrification is the same irrespective of the conditions is yet to be investigated for freshwater, Lake Victoria. Otherwise, nitrogen reduction has the potential of occurring throughout the lake waters since stratification plays a major role in DO profile of the lake. Depending on the lake conditions, aerobic nitrification may be followed by anaerobic denitrification. Under such conditions, there is high potential of nitrous oxide production as compared to nitrogen gas. Typical aerobic denitrifiers as *P. denitrificans* produce more N_2O than nitrogen gas under aerobic conditions meanwhile more of the nitrogen is produced by *Pseudomonas stutzeri* TR2 and *Pseudomonas species* (K50 strain) under the same conditions (Takaya et al., 2003). Therefore, the observed changes in invertebrate community structure and species abundance could also be caused by DO changes for example in the case of *Caridina and chironomids* (McMahon et al., 1974).

Influence of aquatic carbon and nitrogen ratio on nitrogen processes

Increasing atmospheric CO_2 due to industrial activities, burning and climate change is bound to result into more CO_2 dissolving into aquatic systems. CO_2 uptake changes the chemical equilibrium and increase the lowering of pH. However, decreasing pH and rising temperature act to reduce CO_2 aquatic buffer capacity and the rate at which aquatic systems take up CO_2 (IPCC, 2007). Such changes therefore influence the C/N ratio that in turn affects the rate of organic matter mineralization in aquatic systems (Boyd, 1998). Carbon affects the activities of autotrophic nitrifying bacteria and denitrifying bacteria (Kadlec and Knight, 1996). When C:N ratio increases, bacteria and other micro-organisms of decay remove nitrate and ammonia (immobilization of nitrogen) from water for use in decomposition; rather less is added to the water (Boyd, 1998). Utilization of some carbon sources such as methanol by denitrifiers is coupled with production of alkalinity (Kadlec and Knight, 1996). In addition, bacterial decomposition of organic matter is also known to depend on both biological parameters such as synthesis of enzymes capable of hydrolyzing the organic substance and chemical characteristics such as compound structure (Wetzel, 2001; Sangkyu and Kang - Hyun, 2003). Lake Victoria waters receiving substances with high carbon will therefore farvor nitrifying autotrophic bacteria. The carbon and nitrogen elemental ratio is also correlated with growth and fecundity of secondary trophic levels in aquatic system (McMahon et al., 1974).

Conclusion and recommendations

N_2O in Lake Victoria is not characterized though large N inputs and DO changes occurring due to stratification create the potential for the production. The nitrogen and carbon cycle, particularly nitrogen fixation and denitrification are reported as processes that need special attention (Gruber and Galloway, 2008). At the moment, no climate and nitrogen projection for the Lake Victoria ecosystem has been done therefore; it is not possible to ascertain climate change effect on the dynamics of N cycle in the lake. The ultimate suggestions on mitigation measures are for the riparian countries to enforce policies that reduce both point and non-point sources of N into the lake and maintain riparian forests and wetlands. This is because aquatic systems have limited ability to adapt to climate change. Reducing the likelihood of impacts to aquatic systems will therefore depend on human activities so as to reduce N sources, aquatic ecosystem stress and enhance adaptation capacity.

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