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# Effects of point source pollution on water quality, phytoplankton diversity and abundance in lake Victoria, Kenya

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**Abstract**
Water quality deterioration remains a major problem in most fresh water lakes. This study was conducted to establish the extent of water quality deterioration in Kisumu Bay of Winam Gulf, Lake Victoria. The overall objective was to assess the effects of point source pollution on water quality and phytoplankton community structure. Sampling stations were selected randomly all over the bay and marked with a global positioning system (G.P.S). Triplicate samples of dissolved oxygen concentration, conductivity, ammonia, nutrients, chlorophyll-*a* and phytoplankton were taken for laboratory analysis. Mean overall dissolved oxygen concentration was 7.41±1.39 mg $l^{-1.}$ Mean conductivity level was 142.39±63.786 µS $cm^{-1}$ whereas mean soluble reactive phosphorus level was 330.02±311.9 mg $l^{-1}$. The mean overall chlorophyll *a* level was 592.9± 604.4 mg $l^{-1}$ indicating that the bay is eutrophic. The Duncan multiple range test showed significant variability of the means of this parameters at different sampling sites at p=0.005. At 46%, the diatoms were the most dominant phytoplankton family in the bay. There were significant differences in distribution of different phytoplankton taxa at different sampling sites (p=0.0001). The results of this study can be used to formulate management options for the improvement of water quality in the Kisumu Bay of Lake Victoria.

**Keywords:** Kisumu Bay, Water Quality Chlorophyll-a, Nutrients, Management options

## 1. Introduction

Freshwaters are among the most seriously threatened ecosystems on the planet, having suffered intense human impacts over the past century (Sala *et al.*, 2000; Cowx *et al*, 2002, Dudgeon *et al*., 2005). Lake Victoria the largest tropical lake is not an exception. It has experienced fundamental changes in ecology due to habitat degradation (Hecky, 1993; Sitoki *et al.*, 2010) [5, 18] eutrophication remains a major problem in Nyanza Gulf of the Lake Victoria ecosystem and is the most stressed (Gichuki, *et al.*, 2006) [14] in relation to other parts of the Lake. The growing population is among the major cause of degradation and pollution of the lake (Africa atlas, 2006; U.N, 1998) [48] which is attributed to increased anthropogenic activities in the watershed (Hecky, 1993; Hecky *et al*., 1994; ILEC, 2005; Davies and Hirji, 2003) [5, 18, 11] In particular, industrialization, urbanization and agricultural activities have led to nutrient loading into the Lake waters (Heathwait1 *et al*., 1998; Veschuren *et al*., 2002).

For instance, the effluents from industries directly find their way into the gulf. The water is further polluted by large discharges of untreated and chemical wastes from the urban centers such as Kisumu, Homa Bay, Kendu Bay and Port Victoria. UNEP (2006) observed that the lake is the final destination of factory effluents, oil, grease and sewage from the urban centers and oil spillage from transportation. In addition, Lake Victoria waters receive 2.3 mm per year of sediment loads -silt, phosphorus, nitrogen and others (Odada, *et al.*, 2003; Veschuren *et al.*, 2002 and 2004). Point source pollution is through many rivers which are the major conduits for the pollutants. Therefore, deteriorating water quality is due to point and non-point pollution fuelled by human activities in the catchment areas (Charles and Alexander, 1993). Eutrophication is manifested in surface waters through proliferation of algal blooms and aquatic macrophytes. This is due to disturbed balance of trophic levels resulting to destabilization of species dominance and distribution. This results to explosive growth of a few species which disturb the biochemical cycles (Awange *et al.*, 2006) [3]. Hence, water quality is compromised (Hecky, and Bugenyi, 1992) [16].

### 1.1 Objectives
The overall objective of the study is to assess the effects of point source pollution on water quality and phytoplankton community structure.

### 1.2 Specific objectives
1. To determine the spatial-temporal variability of physical-chemical and biological parameters within Kisumu Bay.
2. To determine the spatial-temporal variation in the concentration of selected nutrients within Kisumu Bay.
3. To determine occurrence percentage composition and diversity indices within Kisumu Bay.
4. To determine the relationship between phytoplankton community structure and water quality parameters.

## 2. Materials and Methods
### 2.1 Study area
The study area constitutes the Kisumu Bay which is situated at the North Eastern end of Nyanza Gulf The bay has an average depth of three metres and receives water from rivers Kisian, Kisat, Auch-Nyamasariaz**.** The bay also receives seepage from the large Nyalenda papyrus swamp and surface runoffs from Kisumu City. The bay is 4.5km long and 1-3km wide covering an area of 8 km$^2$. Rainfall in the catchment is about 1150 mm/yr especially in the Eastern side (average -1550 mm/yr) and is seasonal with two maxima. The main rain seasons are from March –May with subsidiary rains in August and November. The monthly maximum air temperatures range from 27.5 $^0$C - 3.0 $^0$C, while the minimum ranges from 16.0 $^0$C -18.0 $^0$C. The winds at Kisumu are mostly South Westerly and are strongest during the afternoons. The shallow nature of the bay coupled with the wind results in the bay being well mixed vertically.

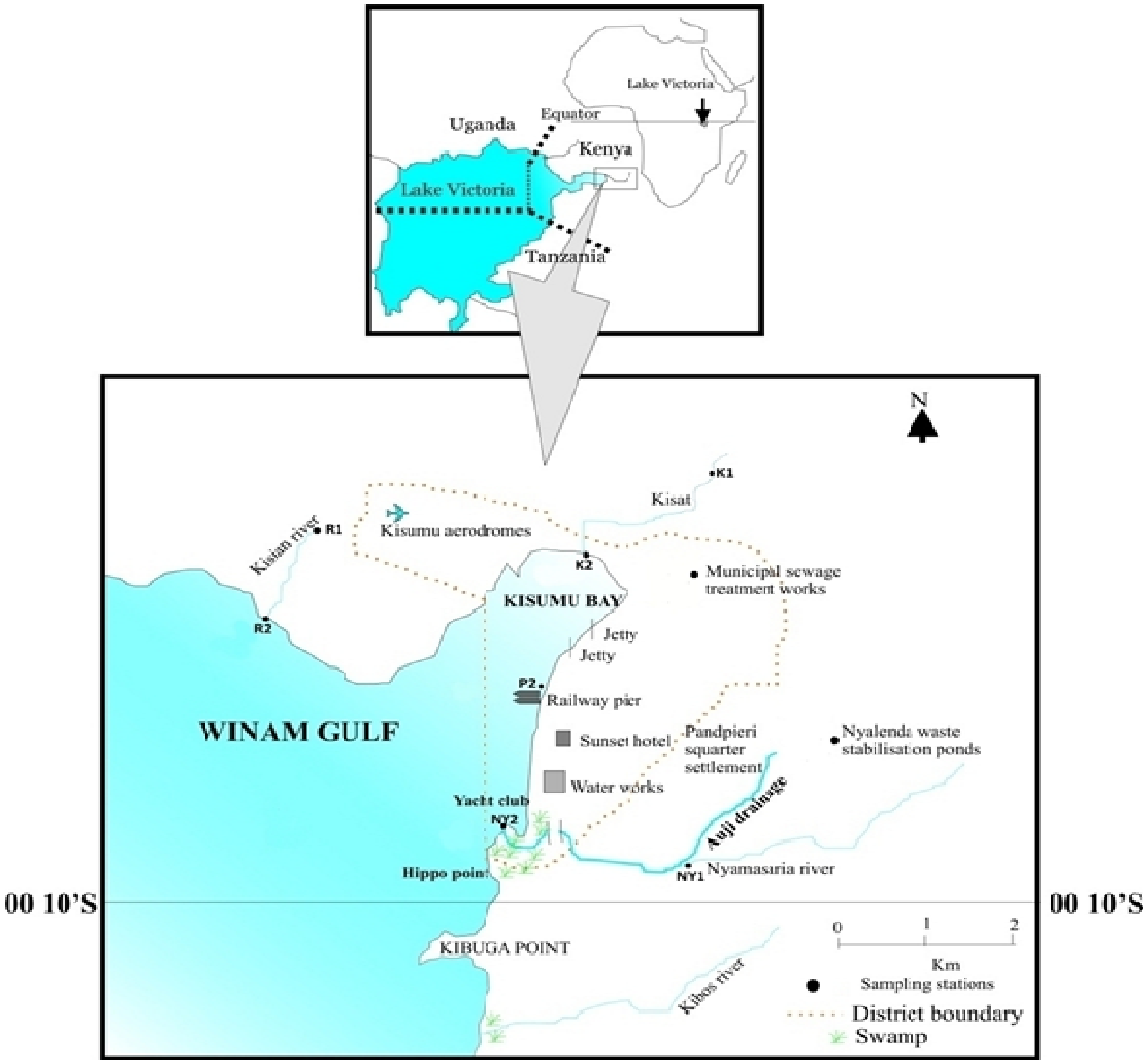



**Fig 1:** A Sketch map showing Lake Victoria and the sampling sites

### 2.2 Sampling and laboratory analysis
Data was collected on a monthly basis from five inshore and two offshore stations between November 2013- May 2014; sampling sites were marked using a Magellan Global Positioning System (GPS) 315 meridian. All samples were taken at a depth of 1m while *in situ* parameters were measured before sampling. Water samples were collected using Van Dorn water sampler of a capacity of 2.5 L, placed in sterile plastic sample bottles and frozen for further nutrient analysis in the laboratory. Phytoplankton samples were collected using

phytoplankton nets, water samplers, fixed using acidic Lugol's solution and stored in plastic vials for laboratory analysis. Secchi depth was measured using a Standard Secchi disk of 20 cm diameter. Turbidity was measured using a Hatch Turbidimeter 2100 P. pH was measured using a digital Mini Model 49- pH meter. Depth, temp, conductivity, chlorophyll *a* were measured using a submersible Conductivity-Temperature-Depth profiling system (CTD, Sea-bird Electronics®), programmed to take measurements at 5 seconds intervals. Alkalinity and hardness were determined using the methods highlighted in APHA (1995). For phytoplankton analyses, a 2 ml sub-sample was taken from the preserved water sample and placed in an Utermöhl sedimentation chamber and left to settle for three hours. Phytoplankton cells were identified to species level where possible and counted using an inverted microscope at 400x magnification. Ten fields of view were counted for the coccoid cyanobacteria. A 12.42 $mm^2$ transect was counted for the larger algae. The whole bottom area of the chamber was examined for the big and rare taxa under low (100x) magnification. Counts were taken on all individual cells, colonies or filaments. Identification was done using the methods of Huber–Pestalozzi (1968) [20] and Cocquyt *et al.* (1993) [9].

## 3. Results
### 3.1 Physico-chemical Parameters
The Mean values of physical-chemical parameters shown in Table. 1, similarly, the table depicts the overall minimum and maximum dissolved oxygen levels across all the stations were 5.7±1.9 mg/l at R2 and 8.4±1.4 mg/l at NY2 respectively. Dissolved oxygen showed significant differences between sampling stations ($F_{(8,44)}$ = 2.22, p = 0.0491). Duncans' multiple range test further revealed that stations above point source pollutions varied from those below the point source pollution except the NY1.

Conductivity levels were highest at R1 (201.6 ± 6mg/l) and lowest at NY2 (90.1 ± 27.4). The variation in Conductivity was significant between stations ($F_{(8,44)}$ = 4.72, p = 0.0005) with pair-wise comparison showing that most stations varying from each other. Temperature levels did not vary significantly between stations though NY1 and NY2. The average minimum and maximum temperatures recorded in the entire study were at NY1 (25.1 ± 2.9 $^0$C) and R1 (27.8 ± 1.5 $^0$C).Turbidity level was highest at R2 (401.2 ± 43.6 NTU) and lowest at P1 (157.1±35.9 NTU). The differences were significant ($F_{(8,44)}$ = 2.72, p = 0.0137) DMRT revealed that this variation was largely due to station R2. PH did not vary significantly between stations. DMRT *post hoc* test further showed that all the stations were in the same subset. Physico-chemical parameters (Mean±SD) at different sampling sites **(**Means with different superscripts in the same column are significantly different at P<0.05).

| Stations | pH | DO (mg/l) | Temp ($^0$C) | Turbidity (NTU) | Cond. (µS/cm) |
|---|---|---|---|---|---|
| R1 | 7.8 ± 0.8$^b$ | 6.7±0.4$^b$ | 27.8±1.5$^b$ | 273±53.4$^b$ | 201.6±62.4$^b$ |
| R2 | 7.4 ± 0.4$^b$ | 5.7±1.9$^a$ | 27.8±3.4$^b$ | 401.2±43.6$^d$ | 195.6±53.9$^a$ |
| K1 | 7.5 ± 0.8$^b$ | 8.0±0.7$^c$ | 26.1±2.8$^b$ | 232.7±87.7$^b$ | 100.4±40$^{bc}$ |
| K2 | 7.3 ± 0.4$^b$ | 7.4±1.7$^b$ | 26.7±3.0$^b$ | 153.9±21.9$^a$ | 109.9±33.4$^{bc}$ |
| NY1 | 7.6 ± 0.3$^b$ | 8.4±1.4$^c$ | 25.1±2.9$^a$ | 241.1±91.9$^b$ | 93.4±26.0$^c$ |
| NY2 | 7.7±0.7$^b$ | 8.0±1.1$^c$ | 25.4±3.2$^a$ | 222.1±110$^b$ | 90.1±27.4$^c$ |
| P1 | 7.5±0.5$^b$ | 7.9±1.0$^b$ | 26.6±3.1$^b$ | 157.1±35.9$^a$ | 125.7±17.1$^c$ |
| F-Value | 1.04 | 2.22 | 0.71 | 1.68 | 4.72 |
| *p – value* | 0.7252 | 0.0491* | 0.6771 | 0.0137* | 0.0005* |

### 3.2. Dissolved Nutrients
Figure 4.2 below shows the mean concentrations for dissolved nutrients within the sampling stations. The minimum and maximum SRP levels in all the sampling sites combined were 4.78 mg/l and 632.0 mg/l respectively, with a range of 627.2 mg/l. Stations K2 and K1 had the highest levels at 330.1 ± 32.86 and 210.7 ± 19.45 respectively and was lowest at R1 (44.2 ± 4.3). There was no significant difference in soluble reactive phosphorus levels between different sampling sites ($F_{(8,44)}$ = 6.42, p = 0.001). Duncan Multiple Range Test further showed that stations K1 and K2 were significantly different from other stations.

The minimum and maximum Total Phosphorus levels in all the sampling sites were 8.11 mg/l and 1281.4 mg/l respectively, with a range of 1273.29 mg/l. The mean overall phosphorus level was 330.02±311.9 mg/l. The concentration was however highest at station K2 (987.7 ± 70.7) and lowest at P1 (157.6 ± 8.7). Total phosphorus levels were significantly different between sampling sites ($F_{(8,44)}$ = 8.8, p = 0.001), with DMRT further showing that total phosphorus levels at K1 and K2 were significantly different from all the other stations.

The ammonium levels was highest at station K2 (885.9 ± 122.3) followed by K1 (764.1 ± 53.3) and was lowest at NY2 (82 ± 6.4The mean ammonium level in all the stations was 319.29±397.35 mg/l. Ammonium concentrations was significant different between sampling stations ($F_{(8,44)}$ = 4.49, p = 0.0008). Duncan Multiple Range Test further showed that the significant variation in ammonium levels was attributed to stations K1 and **K2** The mean total nitrogen concentration was 592.9±604.38 mg/l. Stations K2 and K1 recorded highest values (1251.4 ± 233.3, and 853.5 ± 84 respectively). The variation between these stations is not significant (p = 0.051). The mean silicate concentrations ranged between 3.2 ± 0.1 in K1to 4.3 ± 0.2 in NY2. The variation in silicates between stations were significant (p = 0.048).

### 3.3. Chlorophyll a
The mean minimum and maximum chlorophyll *a* of 1.46mg/l and 289.1mg/ was recorded at K1 (102.1±15.2) and R1 (18.4 ± 3.4) respectively (Fig 4.3). The mean overall Chlorophyll *a* level was 592.9±604.38 mg/l. These variations were significant (p = 0.031) but DMRT post hoc test showed that its stations K1 and K2 that were at different subset from other stations.

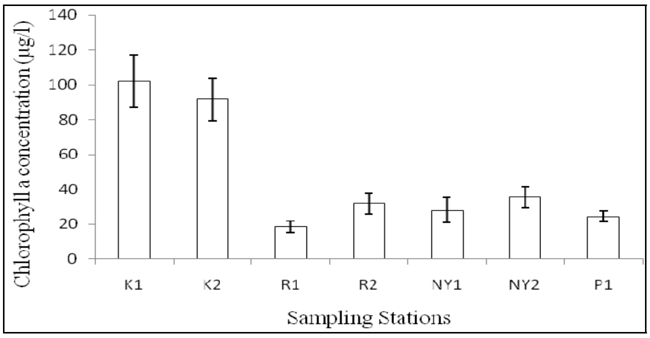



**Fig 4.3:** Mean Chlorophyll *a* levels as measured at different sampling sites.

### 3.3.1 Spatial variation in Phytoplankton

Diatoms were the most prominent algae in all the stations except at P1 and R1 where Cynophyceae was dominant. Highest proportions of diatoms were recorded at R2, followed by NY1 and the lowest at P1. NY2 recorded almost similar proportions of Chlorophyceae, Bacillariophyceae and Cyanophyceae (Figure 4.5).

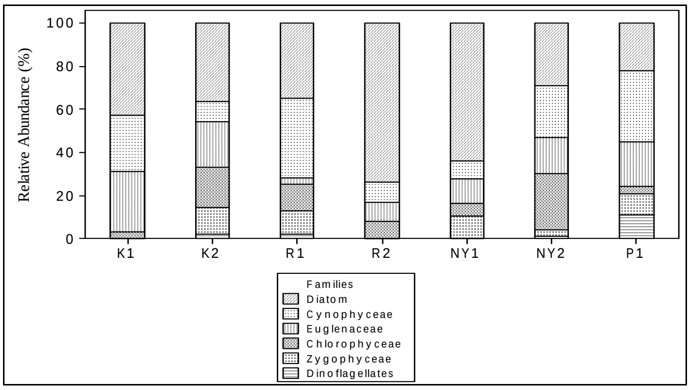



**Fig 4.5:** Percentage algal composition per site

Among the Cyanophyceae, the most abundant species were *Microcystis*, *Anabaena* and *Chroococcus spp*, while *Aulacoseira* spp, *Nitzschia* spp and *Navicula* spp dominated the family Bacillariophyceae, while the dominant Euglenaphyceae were *Euglena* spp, *Phacus* spp and *Trachelomonas* spp. The families Zygnemaphyceae were dominated by *Closterium* spp and *Cosmarium* spp. The shanon-weiner diversity index was highest at P1 (2.36 ± 0.14) followed by K2 (1.91 ± 0.08) while was lowest at R2 with an index of 0.76 ± 0.04

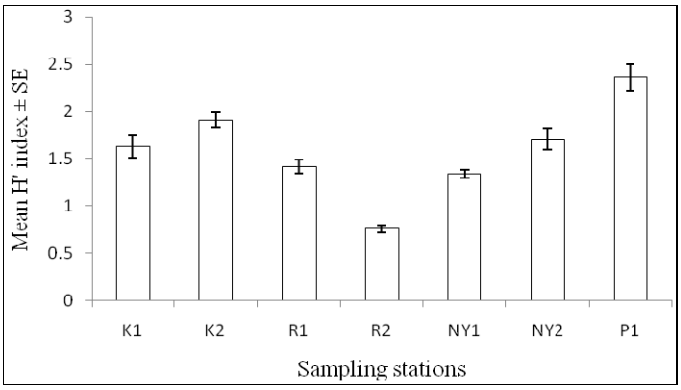
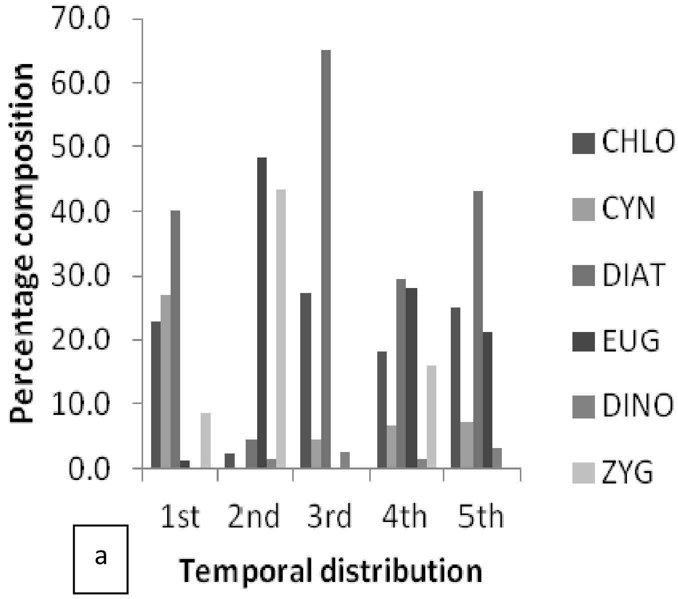
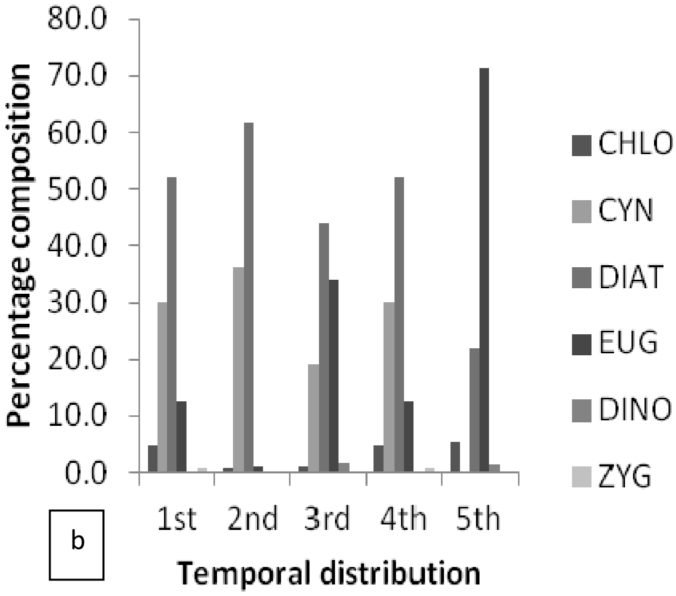



**Fig 4.6:** Shanon-Weiner diversity index per sampling station

Considering the temporal distribution of phytoplankton per sampling site, K2 had a large proportion of diatoms in all the sampling occasions except during 2nd month of sampling in which Euglenophytes were dominant. Diatoms were also dominant temporally at K1 except in the 5th month where Euglenophytes were dominant (Figure 4.7a & b).




### 3.3.2 Principal Component Analysis

Principal component analysis (Fig 4.11) showed that station K1 was mainly influenced by chlorophyll a, TN, and SRP. Station K2 associated mostly with Euglenaphyceae, chlorophyceae where as P1 associated with dinoflagellates and Zygnematophyceae R1 and NY2 associated with cynophyceae and was influenced mainly by pH. Station R2 associated more with diatoms and was influenced mainly with turbidity. Turbidity together with silicates influenced station NY2.

**Table 4.2:** Correlation Coeficient (r) and p values for phytoplankton-water quality parameter relationship (* shows significant relationship)

| | Cynophyceae | Euglenaceae | Chlorophyceae | Zygophyceae | Dinoflagellates |
|---|---|---|---|---|---|
| SRP | -0.306 | 0.623 | 0.163 | 0.098 | -0.145 |
| | 0.505 | 0.135 | 0.727 | 0.834 | 0.756 |
| TN | -0.347 | 0.614 | 0.229 | 0.029 | -0.106 |
| | 0.446 | 0.143 | 0.621 | 0.951 | 0.821 |
| TP | -0.394 | 0.453 | 0.236 | 0.240 | -0.207 |
| | 0.382 | 0.308 | 0.611 | 0.604 | 0.656 |
| Ammonia | -0.210 | 0.582 | 0.004 | 0.002 | -0.234 |
| | 0.651 | 0.170 | 0.993 | 0.996 | 0.613 |
| Silicates | -0.075 | -0.607 | 0.625 | 0.232 | -0.312 |
| | 0.873 | 0.148 | 0.133 | 0.617 | 0.496 |
| Chloro a | -0.235 | 0.759* | 0.025 | -0.216 | -0.298 |
| | 0.612 | 0.048 | 0.957 | 0.642 | 0.516 |
| Alkalinity | -0.276 | 0.869* | 0.209 | 0.039 | -0.003 |
| | 0.550 | 0.011 | 0.652 | 0.934 | 0.995 |
| Hardness | 0.261 | 0.660 | -0.466 | -0.512 | -0.132 |
| | 0.571 | 0.106 | 0.292 | 0.240 | 0.778 |
| Ph | 0.617 | -0.503 | 0.162 | 0.085 | -0.058 |
| | 0.140 | 0.250 | 0.729 | 0.856 | 0.902 |
| DO | 0.109 | 0.556 | -0.013 | 0.243 | 0.180 |
| | 0.817 | 0.195 | 0.978 | 0.599 | 0.700 |
| Temp | 0.214 | -0.442 | -0.138 | -0.003 | 0.118 |
| | 0.645 | 0.321 | 0.768 | 0.995 | 0.800 |
| Turb | -0.250 | -0.592 | -0.163 | -0.578 | -0.517 |
| | 0.589 | 0.162 | 0.727 | 0.174 | 0.234 |
| Cond | 0.213 | -0.714 | -0.169 | -0.031 | 0.005 |
| | 0.647 | 0.072 | 0.717 | 0.948 | 0.992 |

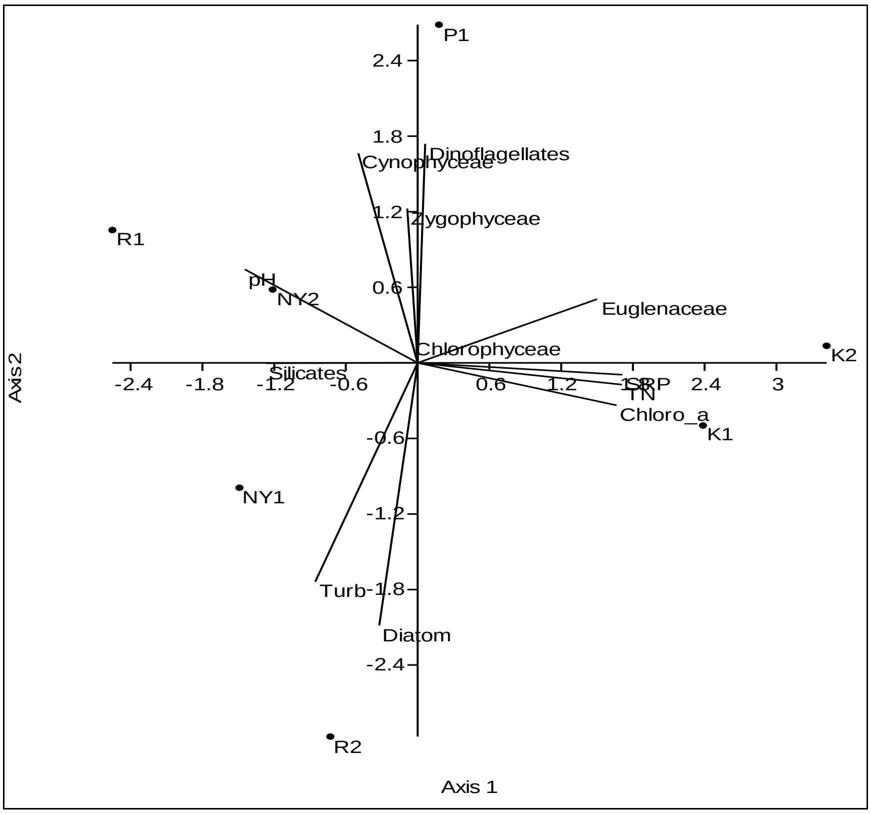



**Fig 4.11:** Principal Component Analysis between phytoplankton families and water quality parameters

## 4. Discussions

### 4.1 Physico-chemical parameters

The low dissolved oxygen levels were particularly recorded at the stations below the point source pollution and could have been as a result of nutrient and organic matter loading into the system from the point sources of pollution. Increased load of organic matter could result in eutrophication and hence trigger microbial activities which may lead to low oxygen levels or oxygen depletion Hecky and Bugenyi (1993) [5, 18]. The relatively high oxygen level recorded at stations above these areas such as NY1, K1, R1 could have been as a result of mixing of water and atmospheric oxygen as the water flows down the river and absence of major point of influx. In addition, proliferation of algae, can also lead to oxygen depletion incase of algal blooms. Studies by Gichuki *et al.* (2006) [14] also showed that water temperatures can influence the amount of oxygen dissolved in water with warmer waters likely to have relatively low dissolved oxygen levels.

Conductivity levels were found higher in stations K1 and K2. A study by Sitoki *et al*. (2010) reported conductivity levels of between 157.72-195.17 µs/cm within Kisumu Bay, which were slightly lower than those recorded at the K stations in the current study. The reason for this could be that the current study sampled water at the upper part of the river as well as at the river mouth, while the study by Sitoki *et al* (2010) measured water sampled from the open lake where aspects of dilution could have been higher hence the slightly lower levels. Nevertheless, the relatively high levels of conductivity at certain sampling points like Kisat river for instance was expected since the river serves as a conduit of urban effluents, fertilizers, sewage and other wastes (Lung'aiya *et al*., 2000; Gikuma-Njuru and Hecky, 2005) [26].

Elevated temperatures at K2 could be explained by the high turbidity of the waters at this point that could then absorb solar radiation and subsequently causes an increase in water temperatures. These findings are consistent with KEMFRI (2005) which showed that in areas with high suspended and dissolved solids, high temperatures are persistent and are often associated with a high concentration of total suspended solids most of which are discharged from influent rivers which then trap the solar energy, thus increasing the water temperatures (Gikuma-Njuru and Hecky, 2005). Studies show that high water temperatures can accelerates oxygen consuming reactions and result in oxygen depletion at particular times of the day, hence affect some aquatic organisms (Hecky, 1993; Lehman *et al.*, 1998) [5, 18].

Turbidity which is closely related to the suspended solids in water was also highest (401.2±343.6 NTU) at K2. This was partly due to the fact that waste from some sections of Kisumu city were discharging into the river directly, The relatively high pH was indicative of pollution status at the sampling points has also reported by (Gale *et al.,* 1993). Untreated sewage waste and livestock activities observed at R2 and K2 could have contributed to the low pH levels recorded at these points.

**4.2. Dissolved Nutrients**

Studies by Scheffer *et al.* (1997) showed that the high phosphorus loads can result to a domain shift in aquatic vegetation which can promote the establishment of turbid waters, phytoplankton dominated, and changes in ecological state and thus a change in the structure of aquatic biota. What of Gichuki *et al.* and Sitoki *et al.* Muggide *et al.* and Hecky-with DMRT further showing that total phosphorus levels were significantly higher. The elevated TP levels at K2 sampling station could have been as a result of the influence from sewage and municipal effluents, Most urban watersheds, produce considerable pollutant loads into the rivers originating from a wide range of sources including municipal wastewater discharges, failing septic systems and sewage overflows Schueler and Simpson (2001).

The high nutrient values experienced particularly at K2 was a clear indication of elevated nutrient input from this particular river. This is due to the fact that the river flows through sections of the town as well as through high populated informal settlements of Obunga slums and also receives wastewater and raw sewage from the faulty municipal sewerage plant. The impact of sewage and waste waters on river Kisat as well as agricultural effect on other rivers such as Kisian have also been reported by Gichuki (2000, 1995) [26]; Hecky, (1993) [5, 18]; Mugidde, (1993) [32]; and Lungaiya *et al*. (2000). The current study showed relatively low silica levels which could probably be associated with the presence of Bacillariophyceae (diatoms) at some of the sampling sites.

Total nitrogen concentrations could be attributed to nutrient loadings manure, fertilizers present in storm water run offs, sewage and municipal effluents, all of which find their way into rivers and subsequently into the lake waters. Previous studies of the Lake Victoria ecosystem showed that the lake has been under the threat of eutrophication since the early 1990s when changes oxygen regimes and phytoplankton composition were noted (Sitoki *et al*., 2010). Comparisons with even earlier data collected by Talling (1965) [45] and (1966) [43] demonstrated serious eutrophication trends in the inshore waters of Napoleon Gulf, Winam Gulf and the open waters near Bugaia Island. Studies show that nutrient enrichment especially of TP comes from human activities such as bush burning: sewage disposal from domestic, municipal and industrial sources and poor land use practices in the catchment areas. Management of human activities in the current areas is therefore key to the management of algal diversity in the LVB.

The high chrolorophyl a concentration level can be attributed to high photosynthesis due to eutrophication at those stations. Tallying, (1966) [43] reported that elevated values in chlorophyll a were attributed to higher photosynthetic activities due to increased eutrophication

**4.3 Phytoplankton species diversity and abundance**

Phytoplankton (algae) is the most important primary producer in Lake Victoria and a vital basis for higher production, including that of fish. Phytoplankton is, therefore, one of the basic water resources in the Lake Victoria ecosystem. Blue-green algae) dominance over Chlorophyceae could be attributed to increased nutrient loading in the study sites from the surrounding areas

Contrary to the current study findings in which diatoms were recorded as the dominant phytoplankton species, previous studies on phytoplankton diversity and productivity demonstrated the dominance of blue green algae (Cyanophyceae) over diatoms (Bacillariophyceae) and green algae chlorophyceae. such studies as those conducted by talling are limited to the near shore zones (Talling, 1966; Ochumba and Kibaara, 1989; Lehman and Brandstator, 1993; Mugidde, 1993; Hecky, 1993; Lehman *et al*., 1998; Lungaiya *et al*., 2002) [43, 39, 32, 5, 18] and most not did not focus on the upstream sections of some of the rivers draining into Lake Victoria, as was done in this study.

The composition and relative abundance of algae are determined by environmental factors especially nutrients and light conditions. The growth of algae depends on dissolved nutrients, trace elements, light and temperature. Algal diversity often changes with depth and the conditions in the surrounding environment.

Equatorial lakes, like those in the LVB exhibit considerable seasonality in algal species composition related to alterations of temperature, stratification and mixing Lungaiya *et al*., 2002)

Over the past decades, changes in nutrient status in Lake Victoria between 1960s and 1990 shows that there have been changes in nutrient, thermal and oxygen regimes.

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