A STUDY ON THE EFFICIENCY OF NAKURU TOWN SEWAGE TREATMENT WORKS

BY

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A thesis submitted in partial fulfillment of the requirement for the award of a degree of master of Science in Microbiology in the School of Pure and Applied Sciences of Kenyatta University.

March 2007

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A study on the efficiency of Nakuru

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DECLARATION

Student declaration
I declare that this is my original work and has not been presented for a degree or any other award in any other university.

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Signature: .................................................. Date: 21-03-07

Supervisors declaration
We confirm that the student carried out the work reported in this thesis under our supervision.

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DEDICATION

I dedicate this work to my husband Daniel Ngari and children Evelyn, Branham and Titus for their encouragement during the research work period.
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I wish to express my gratitude to my supervisors for their guidance and willingness to assist me in this study. I also appreciate the financial support offered by Mr. George Okore (SPHRO) on behalf of the Permanent Secretary, Ministry of Water and Irrigation to facilitate this study. I acknowledge the technical assistance offered by Daniel Mwangi and Mwaura Murigi. A lot of thanks go to my husband and children for being supportive and understanding throughout my study period. I also appreciate the assistance offered by my colleagues at Nakuru, Ministry of Water and Irrigation, Provincial Water Office especially during the sampling days. Finally, my thanks go to all other friends who assisted me during this study. My prayer is, God bless you all.
Abstract

A study to evaluate the efficiency of Nakuru town sewage treatment plant (TSTP) was carried out between August 2003 and March 2004. TSTP is located within Nakuru National Park and discharges its final effluent into Lake Nakuru via a small stream or a trickle. The treatment plant has two wastewater treatment systems. One system is a combination of both conventional treatment units and wastewater stabilization ponds in series (trickling filter line). The second system comprises wastewater stabilization ponds only (anaerobic pond line). This study aimed at establishing the efficiency of the various treatment units within the plant and whether the final effluent meets the minimum requirement for discharge to the environment. This was achieved by determining changes in the levels of selected physico-chemical and biological properties of the wastewater at different stages along the two treatment systems. There were significant differences ($P < 0.05$) in the concentration of most of the selected physico-chemical and biological properties between equivalent treatment units and of the two treatment lines. All the treatment units did not meet the designer's specification in terms of BOD reduction. For example, a mean BOD of $299.4 \text{ mg O}_2 \text{ L}^{-1}$ observed at the anaerobic pond treatment unit was much higher than the expected value of $206 \text{ mg O}_2 \text{ L}^{-1}$. The nutrient levels in the final effluent from the anaerobic pond line (APL) were within the set limits while those from trickling filter line (TFL) were above the set limits for discharge to the environment. There were significant reductions in microbial numbers by most treatment units along both APL and TFL. Mean densities of faecal coliforms (261 coliforms/100ml) observed in the final maturation pond effluent of APL was significantly lower ($P = 0.007, n = 22$) than that recorded in the same pond along TFL (6500 coliforms/100ml). A mean of 261 faecal coliforms/100 ml recorded in the final effluent from APL was lower than the set limit of 1000 faecal coliforms/100 ml recommended by the designers of the plant for discharge to the lake. Generally APL achieved higher organic and microbial load reductions. Identification of algal species in the facultative and maturation ponds revealed presence of potentially toxic algae such as Microcystis in the ponds along both APL and TFL. The results generated from this study indicate that; various treatment units do not meet their expected performance standards, there is a danger of algal toxin poisoning to wildlife, and that there is a danger of polluting the environment with human pathogens. Corrective measures should be taken to improve the performance of the treatment units and the quality of the final effluent.
ACRONYMS AND ABBREVIATIONS

AP1, AP2 & AP3 - Anaerobic Pond 1, Anaerobic Pond 2 & Anaerobic Pond 3

APHA - American Public Health Association

APL - Anaerobic Pond Line

AWWA - American Water Works Association

BOD - Biological Oxygen Demand

CIDA - Canadian International Development Agency

COD - Chemical Oxygen Demand

D - Digester

DO - Dissolved Oxygen

DWAF - Department of Water Affairs and Forestry

EDTA - Ethylenediamine Tetracetic Acid

FP1 & FP2 - Facultative Pond 1 & Facultative Pond 2

FC - Faecal coliforms

GP - Grass Plot

IDCR - International Development Research Center

IN - Inlet

KEMRI - Kenya Medical Research Institute

MP1, MP2 & MP3 - Maturation Pond 1, Maturation Pond 2 & Maturation Pond 3

MIU - Motility Indole Urease Test medium

MPN - Most Probable Number

NSTP - Njoro Sewage Treatment Plant

RF - Rock Filter

SP - Sampling Point

SS agar - Salmonella-Shigella agar
TCBS - Thiosulphate-Citrate-Bile salt- Sucrose agar
TFL - Trickling Filter Line
TF - Trickling Filter
TSTP - Town Sewage Treatment Plant
WWEE – World Water and Environmental Engineering
WEF - Water Environmental Federation
WHO - World Health Organization
WSP - Wastewater Stabilization Ponds
WSP-AF - Water Sanitation Programme - Africa
1C - 1st Primary Sedimentation Tank
2C - 2nd Secondary Sedimentation Tank
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CHAPTER 1

1 INTRODUCTION

1.1 Background to the study

Sewage is the wastewater released from domestic, agricultural, and industrial activities of a community. Sewage comprises 99.94 % water with only 0.06 % dissolved and suspended solids (Karen, 1996). The aim of wastewater treatment is to reduce, as much as possible, sanitary and ecological risks associated with raw or inadequately treated sewage disposed into aquatic systems (Arana et al., 2000). Every ecosystem has a limited biodegradative capacity, which enables it to cope with a limited amount of inorganic and organic wastes through use of inorganic nutrients and biodegradation of the organic matter by microorganisms in the ecosystem. Discharge of raw sewage into a water body overwhelms the homeostasis of aquatic microbial communities and causes unacceptable deterioration of water quality (Dart et al., 1977; La Riviere, 1977).

Many waterborne diseases are as a result of faecal contamination of drinking water supplies and remain a major hazard in many parts of the world (Cairncross & Faecham, 1993). Each year more than 500 million people are afflicted with waterborne diseases and more than 10 million die as a result (Atlas, 1989). In Kenya diarrhoea is largely attributed to faecal contamination of water and poor sanitation (CARE, 2003). It is rated as the fourth cause of death in children less than five years and accounts for 40 % of all outpatient cases. At Kenyatta National Hospital, which is the biggest referral and teaching hospital in Kenya, 60 % of all admission cases are due to diarrhoea (CARE, 2003). Kenya’s Rapid Food and Livelihood Assessment of 1996 and the subsequent baseline survey of 1999 revealed that 66 % of the population in western Kenya lacked access to clean water and 47 % of the children under five years had diarrhoea in the
preceding two weeks (CARE, 2003). One way this problem can be avoided is by proper
treatment of domestic wastewater.

Nowadays, most urban centers worldwide have domestic wastewater treatment plants
to handle sewage. The commonly used methods of sewage treatment are wastewater
stabilization ponds (WSP) and the conventional system. The conventional treatment
system which comprises primary and secondary sedimentation tanks, trickling filter and
sludge digester (aerobic or anaerobic) require large capital investment and high
maintenance cost and is therefore not a feasible solution for developing countries
(Cairncross & Feacham, 1983; Niemczynowicz, 1996; Edwards, 1996). The
conventional system is efficient in reducing BOD and nutrients but not pathogens
(Bartone, 1991). Its dependence on chemical disinfection for the final effluent
complicates effluent reuse in non-restricted irrigation schemes (Mara & Pearson, 1998;

WSP are large, man-made basins into which wastewater flows and from which high
quality treated effluent can be produced after retention time of days as opposed to hours
in a conventional treatment process (Mara et al., 1998). They are simple to operate,
cheaper in terms of maintenance, do not require highly trained personnel, have high
efficiency and are therefore technologies of choice for developing countries (Yanez &

Current interest in the reuse of biologically treated effluents (Gleick, 1998) has led
some developed countries to come up with sophisticated wastewater treatment plants
which are capable of producing high quality effluents. Good examples are Honolulu
water reclamation plant in Hawaiian Island (WWEE, 1999b) and Gut Marienhof
wastewater treatment plant located to the north of Munich, Germany (WWEE, 2004d).
These modern plants employ the new technologies of ultra membrane filtration and reverse osmosis. The effluents produced are of very high quality and can be used for unrestricted irrigation and in industries for non-potable purposes. In other countries e.g. Singapore where water is scarce, a new technology of Multiple Water Reuse is embraced (Chemosensory Research Center, 2003). This technology purifies wastewater from sewers to standards suitable for many households, agricultural and industrial uses.

In developing countries, a wastewater treatment plant is one of the basic infrastructures many poor communities cannot afford as the financial demand for sewage treatment greatly exceeds the resources available to provide it (Reed, 1995). There is also a serious problem in maintaining the few that are in existence. In India, most existing conventional treatment plants needs repair (Chawathe & Kantawala, 1987). A report by Canadian International Development Agency (CIDA, 1998) indicates that in Malawi, 30% of established wastewater facilities are not functional as a result of poor operation and maintenance caused by insufficient funds for the sanitation sector, unskilled workers, vandalism and over prioritization of potable water sector over sanitation sector. A report by Institute of Marine Science (1998) indicates that in Tanzania, Dar-es-salaam has no operative sewage collection infrastructure. The existing sewerage system is old and dilapidated. This has led to frequent outbreaks of water-borne diseases especially during the rainy seasons in the city due to seepage of sewage into stormwater. It was built in the late 1950’s and its attempted rehabilitation between 1980 and 1988 was unsuccessful. Sewage from the areas served by this sewage system is discharged into the Indian ocean untreated.

Most major towns in Kenya have conventional (mechanized) wastewater treatment plants, the majority of which are either non-functional or malfunctioning. Such plants are found in Nairobi, Thika, Nakuru, Eldoret and Kisumu. Most of them are dogged by
operational and maintenance problems resulting from their complexity, lack of spare parts and poor maintenance by the councils concerned. For example, a visit to the Nairobi Kariobangi wastewater treatment plant (conventional) in 1996 revealed that the plant urgently required a major rehabilitation of its treatment units. At the time of a visit to the plant, all the clarifiers, trickling filters and anaerobic digesters were out of order and all the domestic wastewater to the plant was being diverted to Ruai wastewater treatment plant that has WSP. The situation has not changed to date. By the year 2003, Thika town’s conventional wastewater treatment plant was not operational and all the influent to the plant was being pumped to a WSP plant in the town (Muiruri, Pers. Comm., 2004). Other towns like Nyahururu have wastewater treatment plants usually with a biological oxygen demand (BOD) well above 100 mg O$_2$ L$^{-1}$ (Yegon, Pers. Comm., 2004). A study carried out by Gatundu (1991) on a number of wastewater treatment plants in Kenya (including Thika and Nairobi) revealed that the effluent from all the treatment plants studied showed BOD levels consistently higher than the standard set by World Health Organization (WHO, 1989) for any effluent discharged into the natural water body (20 mg O$_2$ L$^{-1}$).

Due to the high cost of maintenance and complexity of conventional wastewater treatment plants, most towns have constructed WSP in addition to the conventional wastewater treatment plants. Examples of such towns are Nairobi, Thika, Nakuru, Eldoret and Kisumu. However, the conventional treatment plants of Kisumu and Eldoret wastewater treatment plants are currently under rehabilitation so as to improve the quality of the effluent released to the environment (Gor, Pers. Comm., 2004).

Nakuru town sewage treatment plant was rehabilitated in 1996 and no detailed investigation has been carried out to assess if the various treatment units meet design specifications. This treatment plant is situated in a conservation site (Nakuru National
Park) where serious ecological problems can occur if efficient treatment of sewage is not observed (Mara, 1976). Sewage effluents contain impurities that can move up the food chain and their long-term effect on the ecosystem are not well understood (Phuong, 2002). This puts more emphasis on the importance of efficient treatment of sewage before it is discharged to the environment.

1.1.1 The process of domestic sewage treatment

The purpose of domestic sewage treatment is to render it harmless to the environment mainly by reducing its BOD, number of pathogenic microorganisms and the concentration of toxic elements such as heavy metals (Atlas, 1989). BOD is the measure of oxygen consumption required for the microbial oxidation of readily degradable organics and ammonia. In general there are four stages in the treatment of sewage. However, depending on the quality of the effluent required, not all the stages may be used.

1.1.1.1 Preliminary treatment

As the sewage enters the treatment plant, it is passed through a grid which traps all large objects such as pieces of wood, plastics, metals etc. The objects trapped by the grid are removed manually and allowed to dry. This solid waste is sorted out, either for recycling, incineration or dumping in waste landfills. After the grid, the sewage enters a grit removal chamber where all grit settles and is removed using a draw pipe located at the bottom of the chamber. All sewage entering a sewage treatment plant must pass through the preliminary treatment stage in order to prevent blockages of effluent channels, fast wearing out or damage to machinery in the treatment works and accumulation of grit in the WSP that could reduce their retention time. These would lead to production of poor quality final effluent.
1.1.1.2 Primary treatment (sedimentation)

In a conventional wastewater treatment plant, wastewater from the grit removal chamber is passed to a primary sedimentation tank whose role is to remove settleable suspended organic matter. The tank has a conical shaped bottom to facilitate effective sedimentation. The sludge formed is removed using a fixed draw pipe at the bottom of the tank and pumped to a sludge thickener from where it is pumped to a sludge digester for further treatment (Atlas, 1989). Excessive organic loading, blockage of sludge withdrawal pipe, worn-out sludge collectors and irregular removal of sludge can cause problems in the sedimentation tank which are indicated by sludge floating on the surface of the tank, presence of black and odorous septic wastewater. The primary treatment stages lower the BOD of the effluent by a value between 30 and 40% (Schroeder, 1977) while the bacterial load is reduced by a value between 40 and 75% (Atlas, 1989).

1.1.1.3 Secondary (biological) treatment

Various biological treatment units use either aerobic, facultative anaerobes or anaerobic microorganisms for biodegradation of dissolved organic matter. The actual process varies between the conventional (mechanical) and stabilization ponds systems. Sewage treatment plants usually have one or both systems. Nakuru town wastewater treatment plant has both systems.

1.1.1.4 Tertiary treatment

Tertiary treatment is usually carried out when a higher quality effluent is required. It involves the removal of non-biodegradable organic pollutants and mineral nutrients, especially nitrogen and phosphorus salts. Tertiary treatment of final effluent is usually carried out using either of the following approaches; chlorination, passage through grass plots, filtration, constructed wetlands, UV light ozonation and chemical
precipitation of nutrients (Atlas, 1989; Mangat et al., 1996). Nakuru town sewage treatment plant employs rock filters and grass plots for its tertiary treatment. The key factor that can lead to poor performance of these units is poor maintenance. The rock filters should be dislodged at appropriate time and grass in the grass plot kept short as recommended by the plant’s design specifications.

Disinfection (using chlorine, ozone or UV light) is a final step in wastewater treatment carried out to reduce the risk of transmission of waterborne diseases but it is not commonly practiced in developing countries. The disinfectant of choice depends on several different considerations ranging from their bactericidal capacity to the production of toxic by-products when they react with organic compounds in the water (Zanetti et al., 1996). For example, chlorine that is commonly used for water disinfection, combine with organic matter present to form chloramines that are carcinogenic to man. The discharge of disinfected effluent into recipient aquatic systems may have deleterious effects on aquatic organisms (Muela et al., 1998). This may also change the quality and quantity of organic matter in the effluents (Langlais et al., 1992; Lazarova et al., 1998).

1.1.2 The convectional systems

Here the wastewater from the primary sedimentation tank is passed through a trickling filter followed by a secondary sedimentation tank. The sludge formed in the sedimentation tanks is passed through either an aerobic or an anaerobic digester for further treatment. Nakuru town sewage treatment plant has an aerobic sludge digester. The sludge from the sludge digester is pumped to the drying bed from where it is disposed off in landfill or used as farm manure.
1.1.2.1 The trickling filter

This is a simple and relatively cheap, aerobic sewage treatment unit. It comprises porous stones or plastics and receives effluent from primary sedimentation tank that is passed through the filter bed. The trickling filter is designed to allow free passage of air throughout the media so that the microorganisms always have enough dissolved oxygen (Atlas, 1989; Atlas, 1993). As the effluent passes through the filter bed, slime forming microorganisms, such as *Zooglea ramigera* and *Sphaerotilus natans*, form a layer surrounding the filter media. Other microorganisms attach to the slime formed resulting in a complex microbial community (Mack *et al.*, 1975; Hawkes, 1977). As the sewage passes through the filter, the microorganisms mineralize the dissolved organic matter while the rest is converted into an insoluble organic form that is removed as sludge in the secondary sedimentation tank. The trickling filter is the key treatment unit in a conventional system (Federal Water Pollution Control Regulation, 1984) and its efficiency determines the quality of its effluent. Its performance can be affected by failure to supply it with adequate volume of wastewater to set it in motion, allowing the filter bed to dry therefore killing the microorganisms involved in biodegradation of organic matter, blockage of its rotating arm by objects, hydraulic and organic overloading.

There are two types of trickling filters, standard and high rate trickling filter. Nakuru town sewage treatment plant has a high rate trickling filter. This type of filter allows continuous re-circulation of wastewater through the filter bed. High rate trickling filter can reduce the BOD of the wastewater by a range of 65 to 80 % (Water Pollution Control Federation, 1928).
1.1.2.2 Aerobic or activated-sludge digester

Sludge from primary and secondary sedimentation tanks is directed to the aerobic sludge digester where it is mixed with recycled sludge, rich in microorganisms and aerated for many hours. Aeration time depends on the project design and varies from 1 to 30 hours (Mason, 1991). Microbial activities result in the production of a clear effluent. The environment within the digester does not favour growth of intestinal pathogens (e.g. Salmonella and Shigella) and their numbers reduce by between 95 to 99% (Atlas, 1989). Organic over loading, poor aeration and re-circulation of activated sludge meant to boost the number of microorganisms within the digester can have negative effect on its efficiency in that, low number of microorganisms would mean low rate of biodegradation of organic matter.

1.1.2.3 Anaerobic sludge digesters

Anaerobic digesters are usually slower than aerobic ones. However, their advantage is that, they can be used to generate biogas, which can be used as fuel within the plant (Cillie et al., 1969; Toerien & Hatting, 1969; Zinder, 1984). Organic over-loading can cause inhibition of methanogenic bacteria involved in the biodegrading of organic matter in this unit leading to its failure. In general, sludge digesters reduce the bacterial load by between 95 to 99% (Atlas, 1993).

1.1.3 The wastewater stabilization ponds system

Wastewater stabilization ponds use anaerobic, facultative and aerobic bacteria to breakdown organic matter in sewage. This is accomplished in anaerobic, facultative and aerobic (maturation) ponds respectively. The ponds can be set up in parallel or in series (Mara et al., 1992; WPCF, 1928). Ponds set up in parallel have the flow split between the ponds. This means that they can take heavier loads with minimal chances of overloading even though they may not produce as good an effluent as ponds in series.
A parallel pond arrangement allows ponds to be closed for cleaning and maintenance by diverting the flow to another pond. Ponds arranged in series reduce chances of short-circuiting but their main disadvantage is that there is a heavier load in the first pond. Sewage retention time in various ponds depends on their design. Ponds in series have a shorter retention time than ponds in parallel (Mara et al., 1992).

1.1.3.1 Anaerobic ponds

Anaerobic ponds are usually 2.5 to 3.6 m deep (WPCF, 1928; Mara et al., 1992) and receive raw sewage after the preliminary treatment. In these ponds there is no dissolved oxygen in any layer of the pond. Two groups of bacteria, non-methanogenic and methanogenic bacteria carry out the biodegradation of the organic waste. Since the anaerobic process involves the two groups of bacteria, it is highly delicate and is not tolerant to any neglect. The degradation of organic matter occurs in two steps. In the first step, the non-methanogenic bacteria hydrolyse organic matter producing volatile acids, carbon dioxide, hydrogen and other end products that stimulates the methanogenic bacteria. The methanogenic bacteria convert the end products of non-methanogenic bacteria to methane gas. Organic overloading leads to increase in the levels of volatile acids lowering the pH within the pond. The low pH inhibits the growth of methanogenic bacteria that can only grow within a narrow pH (about 7) range, leading to the failure of the treatment processes. Prolonged accumulation of sludge can cause a problem by reducing the depth of the ponds, a factor that can lower the performance of this pond as a result of decreased retention time. Hydraulic overloading can also affect the effectiveness of this pond by reducing the retention time leading to production of poor quality effluent. The supernatant from the anaerobic ponds flow to the facultative ponds, which are about 1.2 to 1.8 m deep (WPCF, 1928, Mangat et al., 1996).
1.1.3.2 Facultative ponds

In the facultative ponds, treatment of sewage is achieved through biodegradation of organic matter by facultative bacteria. These ponds are designed to handle low organic load effluent in order to permit development of an active algal population (Miguel, 2003). The algae play an important role in the facultative and the maturation ponds in that they provide the oxygen required by the bacteria for their metabolism. Excessive growth of algae in the ponds can cause dense algal blooms that would interfere with sunlight penetration, hence affecting the rate of phytoplankton photosynthesis (Kinney & Roman, 1998). This would lead to anaerobic conditions within the pond and, as a result, affect the performance/efficiency of the ponds. Owing to the high turbidity of wastewater in this pond, algal biomass is generally low. In Nakuru town sewage treatment plant each treatment line has two facultative ponds that are 1.8 m deep and in a parallel set up. Each pond has a retention time of 21 days. The efficiency of these ponds can be reduced by prolonged accumulation of sludge at the bottom of the pond, presence of dense algal blooms (Miguel, 2003), organic and hydraulic overloading (Maehlum & Stalnacke, 1999).

1.1.3.3 Maturation ponds

The maturation ponds are purely aerobic and are usually 0.6 to 1.2 m deep (WPCF, 1928). The photosynthetic activity of the algae provides the oxygen needed by the aerobic bacteria and other organisms. The maturation ponds in Nakuru town sewage treatment plant are in series and have a retention time of three days each. The effluent usually produced has a low pathogen count if the treatment plant is operating efficiently (Pescod & Arar, 1985; Miguel, 2003; Curtis, 1994). Inefficiency and poor maintenance of the preceding ponds affect the quality of the effluent produced. The combination of both physical and biological treatment in an efficient plant reduces the influent BOD by a value between 80 and 90 % (Atlas, 1989).
1.2 Statement of the problem

Treated wastewater from Nakuru town sewage treatment plant forms a rivulet that discharges into Lake Nakuru. The lake is within Lake Nakuru National Park, which is the second most visited Park in Kenya after Masai Mara National Park. Although the park has a wide variety of wildlife, the main attraction in the lake is the Lesser Flamingos (*Phoeniconaias minor* Geoffrey).

In recent years, massive flamingo deaths have been reported within the Rift Valley lakes (Bennum & Nasirva, 2000; Wanjiru, 2001). For example, in August-November 1993 and August-September 1995, over 40,000 lesser flamingos died in Lake Nakuru (Yarrow *et al.*, 1998). Various reasons have been advanced for the deaths. Among the reasons advanced are cyanotoxin poisoning (Motelin *et al.*, 1995) heavy metal poisoning (Kairu, 1996; Nelson *et al.*, 1998) and infection by *Mycobacterium* (Kock *et al.*, 1999). Both hepatotoxic and neurotoxic algae have been detected in Lake Nakuru and its occurrence in the lake can be linked to lake eutrophication. Hence there is need to assess the various potential sources of nutrient enrichment of Lake Nakuru. As the sewage treatment plant is one potential source of nutrient enrichment, this study aims at establishing whether each stage of the sewage treatment process meets the design standards and whether the treated wastewater poses any harm or threat to the lake ecosystem.

Recent surveys have revealed that, the exposure of wildlife to human pathogens has resulted an increase in incidences of infectious diseases in wildlife (Peter *et al.*, 2000). Pathogen pollution seems to be taking over as the main threat to biodiversity (Peter *et al.*, 2000). Current studies on the death of wildlife indicate that most deaths are occurring in areas subject to human pollution and that when human pathogens are exposed to wildlife, there is an increased risk that pathogens will mutate leading to emergence of new diseases. Hence, if for any reason the treatment process at Nakuru
Town Sewage Treatment Plant is incomplete, ecological problems may result, for example, death of wild animals drinking water from the rivulet formed by the final effluent as it flows to Lake Nakuru.

1.3 Hypothesis

1.3.1 Null hypotheses

(i) The treatment process has no significant impact on effluent BOD and levels of COD, pH, conductivity, total alkalinity, ammonia nitrogen, nitrates nitrogen, nitrate nitrogen, total nitrogen and total phosphorus of the wastewater at different stages of the treatment processes.

(ii) The treatment process has no significant impact on density of faecal coliforms, Salmonella spp, Shigella spp and Vibrio cholerae of the wastewater at different stages of the treatment processes.

(iii) There is no significant relationship between phytoplankton biomass, species composition and changes in nutrient concentrations in the facultative and maturation ponds.

(iii) The treatment process has no significant impact on the quality of sewage effluents at different stages of the treatment processes and does not meet the plant's design specifications.

1.3.2 Alternative hypotheses

(iv) The treatment process has a significant impact on effluent BOD and levels of COD, pH, conductivity, total alkalinity, ammonia nitrogen, nitrates nitrogen, nitrate nitrogen, total nitrogen and total phosphorus of the wastewater at different stages of the treatment processes.
The treatment process has a significant impact on densities of faecal coliforms, *Salmonella* spp, *Shigella* spp and *Vibrio cholerae* of the wastewater at different stages of the treatment processes.

There is a significant relationship between phytoplankton biomass, species composition and changes in nutrient concentrations in the facultative and maturation ponds.

The treatment process has a significant impact on the quality of sewage effluents at different stages of the treatment processes and does not meet the plant’s design specifications.

### 1.4 Objectives of the study

#### 1.4.1 General objective

The overall objective of the study is to evaluate the performance of various wastewater treatment units of Nakuru Town Sewage Treatment Plant and the quality of the final effluent.

#### 1.4.2 Specific objectives

(i) Establish the changes in BOD and selected physico-chemical properties at different stages of the treatment process.

(ii) Quantify changes in faecal coliforms and selected pathogenic microorganisms at different stages of the treatment process.

(iii) Establish phytoplankton biomass and species composition in the facultative and maturation ponds and relate biomass changes to nutrient concentration.

(iv) Determine whether the quality of sewage effluents at different stages of the treatment processes meet the plant’s design specifications.
1.5 Significance of the study

The data generated from the study will provide a valuable insight into the performance of the various wastewater treatment units and give an indication of whether the expected standards of effluent purification are met. The information generated will also help highlight the likely environmental problems associated with discharge of treated domestic effluents to the environment in general and to domestic animals and wildlife in particular. The study findings will also help in establishing whether the final effluent makes a significant contribution to the nutrient levels in Lake Nakuru and if the performance of the treatment units is below the plant’s specifications and recommend ways of enhancing their performance.
2 LITERATURE REVIEW

2.1 Effects of inadequately treated sewage effluents on the environment

Environmental pollution is presently a major cause for concern among several non-governmental and governmental organizations (Ahmad, 1990; World Resource Institute et al., 1996). World Bank data for the Mediterranean and North Africa region indicates serious nitrogenous aquatic pollution due to failure to treat sewage (Munasinghe, 1992). Nutrients in sewage effluents partly contribute to the current global nutrient pollution. The presence of these nutrients in water bodies combined with global climate change has contributed to harmful algal blooms worldwide (Flo, 2001). Discharge of treated sewage effluent water bodies leads to great loss of nutrients and can have a detrimental impact on coastal ecosystems (Appasamy & Lundqvist, 1993).

A number of studies (Geary 1993; O'Neill et al., 1993; Jelliffe 1995a) have indicated that disposal of raw sewage or poorly treated domestic effluents can have serious environmental effects, which include nutrient-related water quality problems as well as bacterial contamination of surface and ground waters. High nutrient levels in a water body leads to growth of algal blooms, which may produce toxins lethal to aquatic life. Toxin producing algae are usually common in eutrophic waters. These algal species produce a wide variety of toxins that can cause liver damage, affect breathing, attack the nervous system, and cause cancer in addition to milder health effects such as diarrhoea, skin rashes, and eye irritation (Dennis, et al., 1999; Codd et al., 1989). One such an algae *Prymnesium parvum* grows well in nutrient-rich and slightly brackish water and produces toxins that cause fish deaths. *P. parvum* adversely affected commercial fish ponds in Israel and severely damaged a first class recreational fishery in Hickling Broad, eastern England (Holdway et al., 1978).
Microcystis aeruginosa produces a hepatotoxin known as microcystins that causes liver damage at low concentrations (Dennis, et al., 1999). Species such as Anabaena flos aquae produce a neurotoxin called anatoxin-a that attacks the central nervous system (Falconer, 1989; Falconer, 1991). For example, in September 1989, eight dogs and a number of sheep died after drinking water from a reservoir at Rutland Water in Leicestershire, Eastern England that had blooms of Microcystis. Filtrates from the algal scum contained the toxin microcystin LR at concentrations of 0.1-1.7 μg L⁻¹ (Gray, 1994). Presence of Microcystis aeruginosa in a pond in Nishinomiya in Japan caused mass deaths of wild ducks (Matsunaga et al., 1999).

Bacterial contamination of surface and ground water by domestic effluents is also an area of great concern because of its negative health effects. In Blantyre, the capital of Malawi, bacteriological investigations carried out on boreholes, wells and standpipes showed that the water contained high levels of faecal coliforms and other enteric pathogens such as Salmonella (Lobster, 1981; Wycliffe & Blessing, 2000). This has led to frequent outbreak of waterborne diseases especially in the low-income urban areas.

In Zambia, bacteriological studies carried between 1996 and 1997 on effluents from sewage treatment plants revealed that the quality of the effluents did not meet the environmental standards of the country (Taylor et al., 1998). The most serious environmental threats in Zambia are due to inadequacy or absence of facilities for the disposal of solid and domestic sewage. This has contributed to frequent outbreaks of cholera and other waterborne diseases in Lusaka (Taylor et al., 1998).

Eutrophication due to the release of inorganic nutrients from domestic sewage into coastal waters around Zanzibar has been identified as one of the main causes of the decreased cover of corals (Institute of Marine Science, 1998). A two year study of water
and sanitation management in Kisumu, on the Kenyan shore of Lake Victoria, indicates that the discharge of inadequately treated sewage into the lake has caused an increase in nutrient levels in the lake (Gunnel, 2001). Kisumu town has two wastewater treatment plants that discharge their final effluents into Lake Victoria. The plants for sometime have been discharging inadequately treated effluent into the lake but are currently being rehabilitated to improve the quality of the final effluent (Gor, Pers. Comm., 2004).

Dissolved oxygen depletion and eutrophication are the most visible problems associated with discharge of inadequately treated sewage effluents into the environment in developing countries (Kenneth, 1995). This has resulted from failure to strictly enforce effluent quality standards and absence of systematic and reliable monitoring of water quality. In Kenya, there seems to be a bit of laxity in the maintenance of the waste stabilization ponds. For example, in Nyeri town the oxidation ponds are often flooded and overflow spilling untreated domestic sewage into a nearby stream hence causing faecal contamination and nutrient enrichment (Njenga, 1980). The water from these streams is used for various domestic chores hence a danger of an outbreak of waterborne diseases. A clear indication of the levels of neglect for wastewater treatment plants was observed at Ruai wastewater treatment plant in 1996, where the scum on the anaerobic ponds was so thick that there was some vegetation growth. The Nakuru wastewater stabilization ponds were commissioned in 1996 and were due for disludging in the year 2000 (Mangat et al., 1996). To date, this has not been done.

2.2 Health effects of inadequate treatment of sewage effluent

Discharge of raw sewage or inadequately treated effluent into water bodies can lead to an outbreak of waterborne diseases. Sewage effluents have contaminated most drinking water wells in the United States of America exposing 3 million people to waterborne diseases and nitrate poisoning (Cunte, 1997). A disease outbreak in Walkerton, Ontario
in USA in May 2000 that resulted in the hospitalization of dozens of people, seven of whom died has been attributed to faecal contamination of a water supply well with *Escherichia coli* (McKay *et al.*, 2002). Failures of on-site treatments (septic tanks) in New South Wales caused an outbreak of hepatitis in Lake Nallis in 1996 (Brook, 1999). Nearly 1,000 sea otters have been found dead along the California coastline in the past five years (Jessup, 2001). The sea otters mortality is associated with peak river flows and storm events and researchers believe that parasites are entering water bodies through untreated or partially treated sewage and terrestrial waste carried in storm-water runoff (Jessup, 2001).

In Africa, a report by World Bank (2001) on sanitation indicates that about 50 % of the population suffers from waterborne diseases. These illnesses are as a result of faecal contamination of water and inadequate sanitary facilities. This problem is most acute in the sub-Saharan Africa where only 60 % of the people have access to clean water supplies. In Ethiopia, waterborne illnesses resulting from faecal contamination are the most common public health problems (Lobster, 1981). A report by Turekegne (2000) indicates that Addis Ababa has inadequate water and sanitation infrastructure, lacks proper sewerage system and has a shortage of potable water. Due to poor sanitation facilities there is persistence of waterborne diseases in Zambia (CIDA, 1998). Faecal contamination of water wells has led to cholera epidemics and other waterborne diseases in peri-urban areas of Lusaka (Taylor *et al.*, 1998). In Malawi, the poor state of sanitary services makes the city residents vulnerable to outbreaks of diseases such as cholera and dysentery, especially in low-income areas (Wycliffe & Blessing, 2000). The capital city, Blantyre has inadequate waste disposal facilities and only 8.3 % of the urban population lives within sewered area. Between 1998 and 1999, there were 2056 cases of cholera outbreaks, which claimed 54 lives in the city. In Uganda, a report by Ministry of Health (1997) indicates that the Uganda Government marginalizes sanitation in planning,
budgeting and resource allocation. Hence incidences of water related diarrhoea are high in the country. In Kenya, a study carried out by Waithaka (1990) in Nakuru (Kihingo location) concluded that there was a scarcity of bacteriologically safe water in Nakuru, and as a result waterborne diseases were prevalent in the area. A study on the prevalence of waterborne diseases within the health facilities in Nakuru District revealed that 5.6% of the patients attended had waterborne related diseases (Hlupheka & Hailemariam, 2001). A socio-economic survey of Nakuru town has revealed that most illnesses reported are associated with bacteriologically contaminated water (Mangat et al., 1998). Typhoid fever and diarrhoea were identified as the common diseases.

2.3 Socio-economic cost of inadequate sewage treatment

The human socio-economic costs of poorly managed domestic wastewater treatment are very high (Munasinghe, 1992). Disease outbreaks that result from inadequate treatment of domestic sewage are a strain to the economies of both communities and governments worldwide (Gleick, 2000). In Peru, a cholera epidemic caused by sewage contamination resulted in an estimated loss amounting to three times the expenditure on water and sanitation for the entire country over the preceding 10 years. In India, the 1994 plague epidemic linked to faecal contamination of water sources resulted in a loss of tourist revenue estimated at $US 200 million (Gleick, 2000) while in Shanghai China, a major outbreak of hepatitis A was attributed to sewage contamination (Munasinghe, 1992; Giles & Brown, 1997). Worldwide waterborne diseases cost over $US 125 billion per year in direct medical expenses and lost work time (Pearce & Warford, 1993). This cost is exclusive of the social cost to communities affected in terms of lost academic hours and productivity of the sick workers. Residents of Karachi living without sanitation or hygiene knowledge spend six times more on medical bills than people with sanitary facilities (Gleick, 2000) while in Ghana, people without good sanitary facilities spend more on medical bills than people with access to sanitary facilities (Khan, 1997).
Failure to recover organic waste from urban areas causes great loss of nutrients that instead of being used in agriculture for food production ends up in the receiving water bodies (Niemczynowicz, 1996). Recycling of organic wastes and sewage effluents in urban areas for agricultural purposes is not expensive and should be encouraged to prevent loss of nutrients and environmental pollution (Appasamy & Lundqvist, 1993; Sanio et al., 1998).

2.4 Potential benefits of proper treatment of sewage

Proper sewage treatment opens the potential for the utilization of this water in economically viable ventures. During the 1950s and 1960s, a greater concern was on capturing and treating wastewater to reduce human and environmental health problems associated with human domestic sewage and industrial waste (Gleick, 1998). More recently, focus has shifted to capturing and treating domestic sewage and reusing it. As a result, considerable research has been done on the application of treated wastewater (Kirkpatrick & Asano, 1986; Asano et al., 1992; Asano, 1994). Wastewater reuse offers two major advantages: it improves the environment by reducing the amount of wastewater discharged into watercourses, and conserves water resources by lowering fresh water demand for irrigation (Khouri et al., 1994). This is important particularly in the arid and semi-arid parts of the world where high quality water currently being used for agriculture could instead be made available for domestic purposes (Pescod & Arar, 1985; Khouri et al., 1994). Reuse has the potential to reduce the cost of both wastewater disposal, the provision of aquifer recharge and water for industrial purposes (Mara, 1976).

Many countries now consider wastewater re-use as a method for conserving water resources (Shelef & Azov, 1996). Most developed countries use treated sewage effluents for various purposes. The most economical re-use of sewage effluent is in food
production, either as irrigation water or by stimulating the growth of algae for fish in aquaculture (Mara, 1976). For example, the high quality effluent produced by Gut Marienhof wastewater treatment plant in Germany is used for unrestricted irrigation and other non-potable purposes (WWEE, 2004d).

The economic benefits of reusing human wastes in agriculture can also be realized at the farm level through supplementing the use of inorganic chemical fertilizers with the reclaimed organic fertilizer derived from biowaste (Sanio et al., 1998). The recycling of organic waste is one aspect of a multi-dimensional and comprehensive approach to upgrading and protecting urban environment resources and aesthetic amenities of the hinterland surrounding the urban centers. Sustainable recovery and re-use of human wastes in agriculture can be economically justified in that producing food close to urban centers means jobs for people and provide the basis for effective wastewater management through providing a sustainable re-distribution of organic nutrients (Cointreau et al., 1984; United Nations Development Programme, 1996; Gardner, 1998). Zero-discharge of organic waste nutrient provides a viable approach for the management of valuable wastewater resources (Gardner, 1998; Harsch, 1996; Otterpohl et al., 1997; 1998).

Due to shortage of potable water and environmental pollution linked to disposal of inadequately treated wastewater into the environment, many countries e.g. USA, Japan, Hawaiian Island, China, Israel, Romania, India and Kuwait have upgraded their wastewater treatment plants in order to produce high quality effluents which is reused for irrigation, industrial and other non-potable purposes (Mary, 1999; WWEE, 2001; Mohamed et al., 2001; National Research Council, 1998; Asano et al., 1996; Chawathe & Kantawala, 1996; Shelef & Azov, 1996). In Ireland the largest wastewater treatment plants have been upgraded and this has halted the local discharge of untreated and treated
sewage into the sea and off-shore dumping of wastewater sludge (Anon., 2004). This has greatly improved the water quality of the beaches of the Dublin Bay. The new technologies of micro filtration and reverse osmosis have been embraced to improve the quality of biologically treated effluents.

China is one of the developing countries that has optimized approaches to recovering and reusing treated domestic wastewater (Chan, 1993). The country has developed a national policy that promotes the development of efficient water treatment technologies, and encourages the reuse of treated wastewater (Zhongxiang & Yi, 1991). In the Middle East where water is scarce, dual distribution systems will in the near future provide high quality biologically treated effluents for industrial purpose and other uses e.g., flushing of toilets (Shelef et al., 1996). In Windhoek, Namibia, reclaimed water has been used to augment the potable water supply since 1968, and in drought years up to 30 % of the city’s drinking water supply is treated wastewater (Van der Merwe & Menge, 1996).

There are socio-cultural objections to agricultural reuse in many parts of East Africa (Mangat et al., 1996). In Kenya, use of wastewater for irrigation is practiced secretly in some parts of the country. In Nakuru it is used for growing kales in Bahati area (Mangat et al., 1996).

In some countries, treated sewage effluents are used for recharging aquifers especially in dry areas (Pescod & Arar, 1985). The recharging technique known as soil aquifer treatment or geo-purification systems, converts sewage effluent into renovated water as it passes through soil of the unsaturated zone or constructed percolation basins which can be collected from the aquifer by wells or drains for reuse (Pescod & Arar, 1985; Bouwer, 1993b). This technique prevents contamination of ground water by reducing the levels of fluoride, heavy metals, faecal coliforms, viruses, total organic carbon and nutrients
through absorption on soil particles preventing the contaminants from reaching the water aquifer (Pescod & Arar, 1985; Lance et al., 1996; Bouwer, 1993b). The technique is widely used in countries such as USA and Israeli (Muniz & Ziegler, 1994; Kanareks & Michail, 1996). Typical examples include the storage of tertiary-treated wastewater in the shallow Florida limestone aquifer system in the USA (Muniz & Ziegler, 1994) and the infiltration and injection of pretreated surface water into Holocene sand dunes in the Netherlands (Stakelbeek et al., 1996).

Australia is one of the dry continents of the world hence has limited sources of potable water. As a result, storm water and treated sewage effluent, previously regarded as wastewater are now re-used through innovative aquifer storage and re-charge technique in the Southern part of the continent (Barnett, et al., 2000). The aquifer in the Nakuru catchments area is enclosed by basement rocks and would require recharge using direct borehole injection (Mangat et al., 1996). However this would necessitate treating the sewage to very high standards.

2.5 Current efforts towards proper sewage treatment

World governments are reneging on their commitment to meet the United Nations target of halving the percentage of people without access to safe drinking water and basic sanitation by 2015 (WWEE, 2004a). As water costs increase and supplies dwindle communities and businesses alike are turning to high-tech wastewater treatment systems that recycle and reuse water (WWEE, 2004a). In south California, satellite systems, which extract domestic wastewater from the main sewer line and treat it locally by ultra-filtration and reverse osmosis techniques for re-use for non-potable purposes, are being used in residential community developments, where issuance of building permit depends on availability of wastewater treatment facility (WWEE, 2004d).
In Australia, a research supported by Victoria Smart Water Fund on dry composting toilets has concluded that, dry composting toilets can save up to 28% of household indoor water use (WWEE, 2004b). Dry composting can also divert over 40% of most pollutants from the sewerage system leaving grey water that is easier to treat for reuse and therefore extends the life of the existing sewers and treatment plants.

In Germany, in order to improve the bathing water quality in the Middle Isar recreational facility, the Munich City Sewerage Authority is building Germany’s largest UV disinfection unit at the Gut Marienhof wastewater treatment plant located to the north of the city (WWEE, 2004c). In mid 1990s, when the Bavarian government launched a project called ‘improving the water quality of the Upper Isar’, sand filters and UV disinfection systems were installed in the wastewater treatment plants of the six neighbouring municipalities of Lenggries, Bad Tolz, Wolfratshausen, Scharftlarn, Penzberg and Benedictbeuern/bchl in Germany (WWEE, 2004c). These led to a marked improvement in the bathing water quality in Upper Isar.

Singapore has six centralized treatment works and has a policy that all wastewater must be discharged into sewers. This has helped to safeguard water catchments from pollution. Sludge and biogas produced are used for soil conditioning and generation of electricity respectively. The government has set up recycling schemes for domestic and conducted extensive public education programmes to promote recycling (WWEE, 1999a).

In South Australia, artificial wetlands constructed have proved effective in removal of contaminant from storm water and wastewater (Bavor & Mitchel, 1994).

Currently, some countries have embraced low cost technologies for treatment of their wastewater. For example, in India Calcutta, treatment of sewage using wetlands technology has been very successful (Ghosh, 1991). The wetlands are more than 3,000
hectares in size and are the world’s largest traditional system for treating domestic wastewater and fertilizing fish production ponds. In the wetland, water is purified through a variety of natural forces (chemical, physical and solar) which act synergistically to achieve wastewater treatment. A series of shallow ponds acts as stabilization lagoons, while the water hyacinth helps to accumulate heavy metals, and the microbial community act further to purify the water (Furedy & Ghosh, 1984). Wetland technologies for wastewater treatment in developing countries offer a comparative advantage over conventional treatment systems because of their level of self-sufficiency, ecological balance and economic value (Ghosh, 1991).
CHAPTER 3

3. MATERIALS AND METHODS

3.1 The study site

Nakuru town sewage treatment plant (TSTP) is located some 2 km to the south of Nakuru town center, near the main entrance to Nakuru National Park. Nakuru town is located within the Great Rift Valley at 36° 05'E, 00° 24'S, some 160 km northwest of Nairobi City. The town has a population of about 296,085 (Ministry of Finance and Planning - Kenya, 2006) and is served by two sewage treatment plants namely, TSTP and Njoro sewage treatment plants (NSTP). TSTP plant has two lines with capacities of 3400 m³ day⁻¹ and 3200 m³ day⁻¹. The 3400 m³ day⁻¹ line consists of primary sedimentation tank, a trickling filter, aerobic sludge digester, secondary clarifier, two parallel facultative ponds each with three maturation ponds in series, a rock filter and two grass plots in series (Fig. 1). The trickling filter was built in 1956 but the ponds and tertiary treatment units (rock filter and the grass plots) were added in 1996. The 3200 m³ day⁻¹ line was constructed in 1996 and is very similar to the 3400 m³ day⁻¹ line but with an anaerobic pond in place of the trickling filter and sedimentation tanks. For the purpose of this study the 3200 and 3400 m³ day⁻¹ will be referred to as anaerobic pond and trickling filter lines respectively. The plant receives domestic sewage only and discharges its treated effluents to Lake Nakuru. The surrounding area is a gently sloping grassland with scattered trees. The area between the plant and the lake is however covered by well-developed woodland dominated by *Acacia xanthophloea* Benth.
Fig. 1 A schematic flow diagram of Nakuru town sewage treatment plant showing the sampling points and the map of Kenya indicating the location of Nakuru town.
3.2 Sampling design

Samples were collected on weekly basis for a period of eight months. Grab samples for physico-chemical and bacteriological analyses were collected at the inlet and outlet of various wastewater treatment units. Samples for BOD analysis were collected in clean opaque plastic containers and immediately kept in a cool box with ice. For bacteriological analyses, samples were collected in sterile bottles and stored in a cool box with ice. Integrated samples for quantitative phytoplankton analyses were randomly collected from each facultative and maturation ponds and fixed / preserved with lugol’s iodine solution. For qualitative phytoplankton analyses, samples were concentrated with a phytoplankton net with a mesh size of 25 μm and fixed with 1 % formaldehyde. Samples for pH, temperature and conductivity were collected in a clean plastic bucket on site and analysed immediately. Once in the laboratory bacteriological and physico-chemical analyses were carried out immediately. Samples for the remaining physico-chemical analyses were refrigerated and the analyses completed within the recommended storage period (APHA, 1998).

3.3 Field studies

3.3.1 Environmental conditions

Rainfall (mm) and air temperature (°C) data was obtained from of the nearby Nakuru meteorological station number 9036261, at the Ministry of Transport, Meteorological Department, Nairobi.

3.3.2 pH

pH was determined using a pH meter (Model WTW LF - 320, Hach Co., Colorado, U.S.A.). This meter measures hydrogen ion concentration by direct potentiometry. pH readings were taken to the nearest one decimal place.
3.3.3 Electrical conductivity (μS cm⁻¹)
Wastewater conductivity was determined using a conductivity meter (Model WTW LF - 320, Hach Co., Colorado, U.S.A.). The meter has a conductance cell containing platinum electrodes coated with carbon, having a surface area of 1 cm² and placed at a distance of 1 cm.

3.3.4 Temperature (°C)
Temperature was taken using conductivity meter (Model WTW LF - 320, Hach Co., Colorado, U.S.A.). The conductivity meter has an in-built temperature sensor, which gives a direct reading of the water temperature in degrees Celsius to one decimal point.

3.4 Laboratory studies
3.4.1 Total alkalinity (mg CaCO₃ L⁻¹)
Total alkalinity was determined using the potentiometric titration method (APHA, 1998). An amount of 50 mls of the sample was titrated with 0.02 N H₂SO₄ using a pH meter (Model No. CG 712, Schott Gerate GmbH Inc., West Germany) to determine the end-point pH of 4.5. Total alkalinity was calculated based on the volume of sample, normality and the amount of titrant used.

3.4.2 Biological oxygen demand (BOD₅, mg O₂ L⁻¹)
Appropriate dilutions of the samples were prepared using aerated nutrient enriched water (APHA, 1998) and poured into BOD bottles in two sets. BOD₅ was determined as the difference between the oxygen concentration of the appropriately diluted sample before and after incubation for 5 days at 20 °C (APHA, 1998). Dissolved oxygen (DO) concentration was electrochemically determined using DO meter (Ultra-DO meter, Central Kagaku Corp., Japan.). The meter reads DO concentration in mg L⁻¹ to one or two decimal points.
3.4.3 Chemical oxygen demand (COD, mg O\textsubscript{2} L\textsuperscript{-1})

This was determined using the closed reflux method (APHA, 1998). Water samples were refluxed for 2 hours with potassium dichromate as the oxidizing agent and sulfuric acid in the presence of mercuric sulphate to neutralize the effect of chlorides (APHA, 1998). Silver sulphate was used as a catalyst. The excess potassium dichromate was titrated against ferrous ammonium sulphate using ferroin as an indicator. The amount of potassium dichromate used is directly proportional to the oxidizable organic and inorganic matter present in the sample.

3.4.4 Total phosphorus (mg L\textsuperscript{-1})

Glassware for total phosphorus analysis were washed with 10 % hydrochloric acid, then rinsed with tap water and finally washed with copious amounts of freshly distilled water. The method involved oxidation of all forms of phosphorus to phosphates by boiling unfiltered sample on a preheated hot plate for 30 to 40 minutes using potassium persulphate as an oxidizing agent. The total phosphorus concentration in samples was determined using the stannous chloride method (APHA, 1998). Under acidic conditions the molybdate ions react with phosphate ions to form molybdophosphoric acid, which is reduced by stannous chloride to intensely colored molybdenum blue. A reagent blank and standards in the range of 0.2 to 1.0 mg L\textsuperscript{-1} were prepared from a stock phosphate solution (APHA, 1998). Colour intensity was measured spectrophotometrically at a wavelength of 690 nm and the phosphates concentrations determined based on the standards curve of known phosphate phosphorus concentrations.

3.4.5 Ammonia nitrogen (mg L\textsuperscript{-1})

The concentration of ammonia nitrogen was determined using diazotization method (APHA, 1992). In this method, ammonium ions react with Nessler’s reagent (K\textsubscript{2}Hgl\textsubscript{4}) to
form a brown coloured compound whose absorbance is determined spectrophotometrically at a wavelength of 425 nm. To prevent interferences caused by turbidity, colour and other substances precipitated by hydroxyl ions such as calcium and magnesium ions, a micro-kjeldahl distillation unit was used for distillation of ammonia present in the water samples. The distilled ammonia was collected in 0.4N H$_2$SO$_4$ acid. A distilled water blank was prepared and passed through the micro-kjeldahl distillation unit in a similar manner as the samples. Ammonia standards were prepared for verifying the accuracy of the spectrophotometer. The concentration of ammonia nitrogen in mg L$^{-1}$ was read directly from the Hach spectrophotometer (model No. RD 2000, Hach Co., Colorado, U.S.A.).

3.4.6 Nitrite nitrogen (mg L$^{-1}$)
Nitrite nitrogen concentration was determined by the diazotization method (APHA, 1998). Nitrites react with sulphanilic acid in an acidic medium (pH 2.0 and 2.5) to form a diazonium salt that combines with N-(1-naphthyl)-ethylenediamine dihydrochloride to form a pinkish azo compound. The samples were first filtered through pre-washed Whatman Glassfibre Filters (Whatman GF/C, Whatman International Ltd, Maidstone, England) and 2 ml of colour reagent added to 50 ml of the sample, mixed and allowed to stand for not less than 12 minutes. Standards in the range of 0.2 to 1.0 mg L$^{-1}$ were prepared and treated as the samples. The absorption of both standards and samples were determined at wavelength 543 nm using Hach Spectrophotometer (model No. RD 3000, Hach Co., Colorado, U.S.A.). The concentration of nitrite nitrogen was determined from the standards curve.

3.4.7 Nitrate nitrogen (mg L$^{-1}$)
Nitrate nitrogen concentration was determined by the cadmium reduction method (APHA, 1998). Nitrates are almost reduced quantitatively to nitrites in the presence of
cadmium (Cd) granules treated with copper sulphate. A portion of the sample was filtered through prewashed Whatman Glassfibre Filters (Whatman GF/C, Whatman International Ltd, Maidstone, England) and treated with EDTA solution. The EDTA treated samples, nitrate nitrogen standards and a blank were passed through the cadmium reduction column to reduce nitrates to nitrites. The resultant nitrites were next analysed as described in 3.4.6 above.

3.4.8 Total nitrogen (mg L$^{-1}$)

Unfiltered water samples were autoclaved at 121 °C, 15 psi for 40 minutes using potassium persulphate as an oxidizing agent (Kalff & Bentzen, 1984). The nitrates formed were next determined by the sodium salicylate technique (Muller & Wilderman, 1955). Sodium salicylate reacts with nitrate ions in acidic medium (H$_2$SO$_4$) to form nitrosalicylic acid that turns yellow under alkaline conditions. A blank and nitrogen standards in the range of 0.2 to 1.0 mg L$^{-1}$ were prepared from a stock solution of potassium nitrate (1.0 N) analytical reagent, then subjected to the same treatment as the samples. The absorbencies of the blank, standards and samples were determined using Hach Spectrophotometer (model No. RD 3000, Hach Co, Colorado, U.S.A.) at wavelength 420 nm and the concentration of total nitrogen calculated based on absorbance values of standard nitrate solutions.

3.4.9 Most probable number (MPN) of faecal coliforms 100 ml$^{-1}$

The levels of both general and faecal coliforms were estimated using the MPN technique (APHA, 1998). The technique involves two stages: a presumptive test for general coliforms and a confirmative test for faecal coliforms. For presumptive test, MacConkey broth medium was prepared, dispensed in universal bottles with Durham tubes and autoclaved for 15 minutes at 121 °C. Dilution water (APHA, 1998) was prepared and appropriate dilutions of the sample prepared. The sterile media was aseptically
inoculated with the diluted sample and incubated at 36 °C for 18 to 24 hours. A change of color (from purple to yellow) and gas production indicates presence of coliforms (positive results). The combinations of positive tubes were recorded and the MPN of general coliforms determined with the aid of McCrady’s statistical table (McCrady, 1918). For the confirmative test, the positive tubes (gas and acid present) were sub-cultured in EC broth and incubated at 44.5 ± 2 °C for 18 to 24 hours. The combinations of positive tubes were recorded and the MPN of faecal coliforms determined with the aid of McCrady’s statistical table (McCrady, 1918).

3.4.10 Vibrio cholerae (counts ml⁻¹)

To isolate Vibrio cholerae serial dilutions of the samples were prepared and aseptically cultured on thiosulphate citrate bile-salt sucrose (TCBS) agar (APHA, 1998; Cheesbrough, 1984). The cultured samples were next aerobically incubated at 35 °C for 18 to 24 hours. Colonies characteristic of Vibrio cholerae (yellow colonies, 2-3 mm in diameter, shiny and surrounded by yellow zone in the medium) were counted, sub-cultured on nutrient agar and incubated at 35 °C for 18 to 24 hours for biochemical identification. Colonies formed on nutrient agar were subjected to various biochemical tests used for differentiation of Vibrio cholerae from other Vibrio spp and related microorganisms that may grow on TCBS agar. These included oxidase tests using Kovac’s reagent, motility in both distilled water and physiological saline, growth in sodium chloride free peptone water, 8 % and 10 % peptone water. To rule out the possibility of the suspected colonies being Vibrio fluvialis, the suspected colonies were inoculated into arabinose sugar and incubated at 35 °C overnight. Vibrio fluvialis ferments arabinose sugar unlike Vibrio cholerae.
3.4.11 Isolation of *Salmonella* spp and *Shigella* spp (counts ml⁻¹)

To isolate *Salmonella* spp and *Shigella* spp, serial dilutions of samples were prepared and aseptically plated on selective media, *Salmonella-Shigella* (SS) agar (APHA, 1998; Cheesbrough, 1984). The cultured samples were incubated at 35 °C for 18 to 24 hours. The initial differentiation of the two species was based on motility and the culture characteristics. The colonies formed by *Salmonella* spp are pale cream with brownish to black centers due to production of H₂S while *Shigella* spp form pale cream colonies with no black centers. The *Shigella* colonies are flat and have a lighter appearance as compared to *Salmonella* spp whose colonies are dome-shaped. Suspected colonies were counted and sub-cultured on Motility-Indole-Urease (MIU) medium for motility, Indole and Urease production tests, conventionally used for physical and biochemical differentiation of *Salmonella* spp from *Shigella* spp and other species that may grow on *Salmonella-Shigella* (SS) agar. The inoculated MIU medium was incubated at 35 °C for 18 to 24 hours and growth characteristics observed. *Salmonella* spp and *Shigella* spp are urease and indole negative although some strains of *Shigella* give different reactions for indole production. Defined substrate technique (4-methy umbeliferone-B-D-glucuro-galactocide) was also used for further confirmation of suspicious colonies of *Shigella* spp.

3.4.12 Chlorophyll a for algal biomass (g m⁻³)

Chlorophyll *a* concentration was determined using Trichromatic Method (APHA, 1998). The phytoplankton samples were concentrated by filtering 100 ml of the water sample through Whatman glassfibre filters (GF/C Whatman International Ltd, Maidstone, England) with the aid of a suction pump. The phytoplankton concentrate on the glass fiber filter was placed on a tissue grinder and 1 to 2 ml of 90 % aqueous acetone and a few drops of magnesium carbonate solution added. The sample was macerated and transferred from the tissue grinder into a 10 ml tube, topped to 10 ml mark using 90 %
aqueous acetone and left overnight at 4 °C to allow pigment extraction. The macerated sample was next centrifuged at 2500 revolutions per minute for 15 minutes and the extract decanted into an Erlenmeyer’s tubes and topped to 25 ml using 90 % aqueous acetone. Optical densities (absorbencies) of the extracts were determined at wavelengths 630 nm, 664 nm 647 nm and 750 nm with a Hach spectrophotometer (model RD 3000, Hach Co, Colorado, U.S.A). The concentration of chlorophyll a, was calculated and algal biomass determined as described APHA (1998).

3.4.13 Phytoplankton identification

Wet mounts were prepared by transferring about 0.1 ml of the net concentrated sample to a clean slide using a wide bore pipette and the wet mount examined microscopically. Algal identification was carried out under the oil immersion objective of an Olympus Phase-Contrast Light Microscope (BX50-32000, Olympus Optical Co., Japan). Species identification was carried out with the aid of appropriate reference books (Hindak, 2001; Timothy et al., 1996; APHA 1998; Hilda & John, 1995; John et al., 2002).

3.4.14 Phytoplankton (counts ml⁻¹)

The lugol preserved samples were shaken gently but thoroughly and using a wide bore pipette, 1 ml sub-sample was transferred into Sedgwick-rafter counting chamber. The sample was left to stand for not less than 15 minutes. Appropriate fields were selected and the counts done microscopically under the ×40 objective of an Olympus phase-contrast light microscope (BX50-32000, Olympus Optical Co., Japan).
CHAPTER 4

4. RESULTS

4.1 Rainfall (mm) and air temperature (°C)

During the study period, the highest total monthly rainfall (201.1 mm) was recorded in August 2003 while the lowest (15.2 mm) was observed in February 2003 (Fig. 2). December and February were the driest months while August was the wettest of all the months. The month of March 2004 was the warmest with a mean temperature of 28.4 °C (Fig. 3). The lowest mean temperature was observed in December 2003.

Fig. 2 Monthly total rainfall (mm) recorded at Nakuru meteorological station 9036261 in the year 2003-2004. (Source: Meteorological Department, Ministry of Transport, Nairobi).

Fig. 3 Monthly mean maximum and minimum temperatures (°C) recorded at Nakuru meteorological station 9036261 in the year 2003-2004. (Source: Meteorological Department, Ministry of Transport, Nairobi).
4.2 Physico-chemical properties

4.2.1 Water temperature (°C)

The lowest (20.3 °C) and the highest temperature (27.0 °C) were recorded in August and in September 2003 along the anaerobic pond line in the facultative and maturation ponds respectively (Fig. 4). During the study period, the lowest mean temperature (22.8 °C) was recorded in the effluent from the anaerobic pond while the highest (24.7 °C) was recorded in the effluent from the rock filter of the anaerobic pond line (Table 1). Along the trickling filter line, the lowest mean temperature was observed in the effluent from the grass plot while the highest (24.3 °C) was recorded in the effluent from the rock filter. Apart from a decline in mean temperature noted in the effluent from anaerobic pond effluent, there was a general increase in mean temperatures (°C) up to the rock filter followed by a decline in the grass plot effluent along anaerobic pond line. Along the trickling filter line, mean temperature variations were irregular (Table 1). The temperature of the influent fluctuated within a narrow range (1.8 °C) throughout the study period (Fig. 4).

Table 1. Means/median levels, ranges and percent reduction of selected physico-chemical properties recorded at different sampling points.

<table>
<thead>
<tr>
<th>TL</th>
<th>SP</th>
<th>Temperature (°C)</th>
<th>Cond. (µS cm⁻¹)</th>
<th>pH</th>
<th>TA (mg CaCO₃ L⁻¹)</th>
<th>PRPTU</th>
<th>CPR</th>
</tr>
</thead>
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<td></td>
<td></td>
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<td>Range</td>
<td></td>
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<td>Range</td>
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Key: AP- anaerobic pond, APL - anaerobic pond line, Cond. - conductivity, FP - facultative pond, GP - grass plot, Me. - median, MP - maturation ponds, RF - rock filter, SP - sampling points, TA - total alkalinity, TL - treatment lines, PRPTU. - percent reduction per treatment unit, CPR - cumulative percent reduction.
Fig. 4 Temporal changes in temperature observed at different sampling points along the two treatment lines. The dotted sections indicate times of no effluent at the sampling point.

**Key:** APL - anaerobic pond line, TFL - trickling filter line.

### 4.2.2 Conductivity (µS cm⁻¹)

The highest (1560 µS cm⁻¹) and the lowest (300 µS cm⁻¹) conductivities were recorded along the anaerobic pond line at the anaerobic pond and the rock filter respectively (Fig. 5). There was a general decline in conductivity along both treatment lines up to the rock filter followed by a slight increase in the final effluent from the grass plot (Table 1). Comparatively, the decline in conductivity was greater along the anaerobic pond line than along trickling filter line treatment units where it was more gradual (Table 1). The mean conductivity recorded in the final effluent from the trickling filter line (975.24 µS cm⁻¹) was significantly higher (P < 0.001, n = 13) than the mean value (746.4 µS cm⁻¹) of the effluent from the anaerobic pond line (Table 1). Over time, conductivity of the influent and effluent from different treatment units fluctuated widely with slightly
reduced levels during wet seasons in the months of August, October (2003) and January 2004 (Fig. 5).

Fig. 5 Temporal changes in conductivity (μS cm⁻¹) recorded at each sampling site. The dotted sections indicate times of no effluent at the sampling point. Key: APL - anaerobic pond line, TFL - trickling filter line.

4.2.3 pH

The lowest pH (6.8) and the highest (10.6) pH values were recorded at anaerobic pond and third maturation pond of the anaerobic pond line in December 2003 and in September 2003 respectively (Table 1). The pH of the influent fluctuated within a fairly narrow range (0.9) as compared to pH range observed at different treatment units (Table 1). The highest pH range was observed at the maturation ponds of the two treatment lines. Generally, pH values recorded during rainy periods were slightly lower than those
recorded during dry periods (Fig. 6). There was a general increase in median pH from the inlet (pH 7.3) up to the maturation ponds where pH values of 9.1 and 8.4 were recorded in the anaerobic and trickling filter lines respectively (Table 1). This was followed by a gradual decline in pH of the effluent from the rock filters and the grass plots along the two treatment lines. Generally, pH of the anaerobic pond line treatment units was higher than that of the trickling filter line treatment units (Fig. 6).

Fig. 6 Temporal changes in pH at different treatment units along the two treatment lines. The dotted sections indicate times of no effluent at the sampling point.

Key: APL - anaerobic pond line, TFL - trickling filter line.

4.2.4 Total alkalinity (mg CaCO\(_3\) L\(^{-1}\))

The highest total alkalinity (520 mg CaCO\(_3\) L\(^{-1}\)) level was recorded in the anaerobic pond effluent in September 2003 while the lowest (90 mg CaCO\(_3\) L\(^{-1}\)) was obtained in the rock
filter of anaerobic pond line sometimes in August and September 2003 (Fig. 7). The highest mean (339.6 mg CaCO₃ L⁻¹) was recorded in the effluent from anaerobic pond and the lowest (159.2 mg CaCO₃ L⁻¹) in the effluent from the grass plot of the same line (Table 1). The highest percent reduction per treatment units was realized at the maturation ponds of the two treatment systems (Table 1). Except in the anaerobic pond effluent where an increase was noted, there was a gradual decline of mean total alkalinity in the anaerobic pond line treatment units up to the grass plot where an overall reduction of a 46.7 % was achieved. In the trickling filter line, an increase in total alkalinity was noted in the effluent from the rock filter and the grass plots resulting in an overall total alkalinity reduction of only 17.2 % (Table 1). Comparatively, higher levels of total alkalinity were recorded during the wet season. The mean total alkalinity of the final effluent from the trickling filter line (247.8 mg CaCO₃ L⁻¹) was significantly higher than a mean of 247.8 mg CaCO₃ L⁻¹ observed in the anaerobic pond line final effluent (159.2 mg CaCO₃ L⁻¹) (P < 0.05; n = 13).
Fig. 7 Temporal changes in total alkalinity at the different treatment units. The dotted sections indicate times of no effluent at the sampling point.

Key: APL - anaerobic pond line, TFL - trickling filter line.

4.2.5 Biological oxygen demand (BOD5, mg O2 L⁻¹)

The lowest influent BOD5 (364.0 mg O₂ L⁻¹) was recorded in August 2003 following a heavy downpour. The highest BOD5 (1200.0 mg O₂ L⁻¹) was observed twice at the inlet in December 2003 and January 2004 (Fig. 8). During the rest of the study period the BOD5 of the influent fluctuated within a fairly narrow range (Fig. 8). A sharp decrease in BOD5 was noted across the treatment units of the two treatment systems up to rock filter from where a slight increase in BOD of the grass plot effluent was noted (Table 2). The highest percent reduction per treatment unit was recorded in the facultative ponds of the two treatment systems (Table 2). The anaerobic pond and the trickling filter lines achieved an overall BOD5 reduction of 97.6 and 94.4 % respectively (Table 2). There
was a significant difference ($P = 0.001, n = 14$) between the mean BOD$_3$ recorded in the final effluent from anaerobic pond line (20.5 mg L$^{-1}$) and trickling filter line (48.3 mg O$_2$ L$^{-1}$). The mean BOD$_3$ of the final effluent from both treatment lines was higher than the set standard of 10 mg O$_2$ L$^{-1}$.

Table 2. Variation in means, ranges and % reduction of BOD$_3$ and COD recorded at each sampling site.

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<th>TL</th>
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<th>BOD$_3$ mg O$_2$ L$^{-1}$</th>
<th>COD mg O$_2$ L$^{-1}$</th>
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Key: TL - treatment lines, SP- sampling point, IN - inlet, AP-anaerobic pond, FP - facultative pond, MP - maturation pond, RF - rock filter, GP - grass plot, APL - anaerobic pond line, Recom.- recommended value, PRPTU. - percent reduction per treatment unit, CPR - cumulative percent reduction.

4. 2.6 Chemical oxygen demand (COD, mg L$^{-1}$)

The lowest influent COD of 640.0 mg O$_2$ L$^{-1}$ was recorded in August 2003 after a heavy downpour whiles the highest BOD values of 2300.0 mg O$_2$ L$^{-1}$ was observed in January 2004 at the inlet. Changes in COD followed a similar trend to that of BOD (Fig. 8 & Fig. 9). The highest COD reduction per treatment unit (63.3 %) was observed at the anaerobic pond and at the facultative pond (49.9 %) of anaerobic pond and the trickling filter lines respectively (Table 2). Overall percent reduction of COD achieved by anaerobic pond and trickling filter lines were 92.6 and 83.6 % respectively (Table 2). Mean COD of the final effluent from the trickling filter line (210.9 mg O$_2$ L$^{-1}$) was significantly higher ($P = 0.001; n = 13$) than that (95.4 mg L$^{-1}$) recorded in the anaerobic pond line. There was a positive correlation ($r = 0.65; P = 0.054$) between the influent mean BOD and COD. The ratio of influent BOD$_3$ to COD was 0.68.
Fig. 8 Changes in BOD at different stages of the treatment process along the two lines. The dotted sections in the line graph indicate times of no effluent at the sampling point.

Key: APL - anaerobic pond line, TFL - trickling filter line.
Fig. 9 Temporal changes in COD recorded at different stages of the treatment process. The dotted sections in the line graph indicate times of no effluent at the sampling point.
Key: APL - anaerobic pond line, TFL - trickling filter line.

4.2.7 Total phosphorus (mg L\(^{-1}\))

The highest concentration of total phosphorus (29.10 mg L\(^{-1}\)) was recorded in the influent in August 2003 and the lowest (1.07 mg L\(^{-1}\)) in the grass plot effluent along the anaerobic pond line sometimes in November 2003 (Fig. 10). Anaerobic pond and trickling filter line maturation ponds achieved the highest total phosphorus removal of 44.8 % and 27.3 % respectively (Table 3). Lowest total phosphorus removal (11.0 %) was noted in the trickling filter treatment unit. There was a general decline in the levels of total phosphorus along the two treatment lines up to the rock filter followed by a slight increase in the grass plot effluent (Table 3). Anaerobic pond and trickling filter lines achieved an overall total phosphorus reduction of 71.5 and 43.1% respectively. There
was a slight decrease in the concentration of total phosphorus in the influent during wet months of August, October (2003) and January 2004 (Fig. 2; Fig. 10). Mean concentration of total phosphorus (9.6 mg L\(^{-1}\)) in the final effluent from the trickling filter line was significantly higher (P = 0.001, n = 13) than 4.0 mg L\(^{-1}\) recorded in the final effluent of the anaerobic pond line and much higher than the set standard of 5 mg L\(^{-1}\).

**Table 3.** Means, ranges and percent reductions of total phosphorus and ammonia nitrogen along the anaerobic and the trickling filter lines.

<table>
<thead>
<tr>
<th>TL</th>
<th>SP</th>
<th>Total phosphorus (mg L(^{-1}))</th>
<th>Ammonia nitrogen (mg L(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Mean</td>
<td>Range</td>
</tr>
<tr>
<td>APL</td>
<td>IN</td>
<td>18.2</td>
<td>12.5 - 29.1</td>
</tr>
<tr>
<td></td>
<td>AP</td>
<td>11.2</td>
<td>5.5 - 20.4</td>
</tr>
<tr>
<td></td>
<td>FP</td>
<td>7.8</td>
<td>3.2 - 16.9</td>
</tr>
<tr>
<td></td>
<td>MP</td>
<td>4.3</td>
<td>1.4 - 8.3</td>
</tr>
<tr>
<td></td>
<td>RF</td>
<td>2.9</td>
<td>1.1 - 5.2</td>
</tr>
<tr>
<td></td>
<td>GP</td>
<td>4.0</td>
<td>1.1 - 11.1</td>
</tr>
<tr>
<td>TFL</td>
<td>TF</td>
<td>16.2</td>
<td>10.4 - 23.8</td>
</tr>
<tr>
<td></td>
<td>FP</td>
<td>14.1</td>
<td>8.5 - 22.5</td>
</tr>
<tr>
<td></td>
<td>MP</td>
<td>10.2</td>
<td>5.3 - 18.3</td>
</tr>
<tr>
<td></td>
<td>RF</td>
<td>8.1</td>
<td>4.2 - 13.3</td>
</tr>
<tr>
<td></td>
<td>GP</td>
<td>9.6</td>
<td>4.3 - 15.5</td>
</tr>
</tbody>
</table>

**Key:** TL - treatment lines, SP - sampling point, IN - inlet, AP - anaerobic pond, FP - facultative pond, MP - maturation pond, RF - rock filter, GP - grass plot, APL - anaerobic pond line, PRPTU - percent reduction per treatment unit, CPR - cumulative percent reduction.
4.2.8 Ammonia nitrogen (mg L⁻¹)

The highest concentration of ammonia nitrogen (108.5 mg L⁻¹) was recorded in the effluent from the anaerobic pond late in November 2003 while the lowest (0.04 mg L⁻¹) was measured in the effluent from the final maturation pond along the anaerobic pond line in September 2003 (Fig. 11). Apart from an increase of 19.8 % observed in the anaerobic pond effluent, there was a sharp decline in the concentration of ammonia nitrogen down the treatment stages up to the maturation ponds from where a progressive increase was noted in the effluent from the rock filters and the grass plots (Table 3). The highest ammonia nitrogen removal was achieved at the maturation ponds of the two treatment systems (Table 3). The anaerobic pond line achieved an overall reduction of
93.55 % that was much higher than 83.70 % achieved by the trickling filter line (Table 3). Mean ammonia nitrogen concentration of 4.11 mg L\(^{-1}\) recorded in the final effluent from anaerobic pond line was significantly lower (\(P = 0.01, n = 13\)) than a mean of 10.36 mg L\(^{-1}\) observed in the final effluent from the trickling filter line. Both treatment lines had much higher values (Table 3) than the set limit of 1.0 mg L\(^{-1}\) for the plant.

![Graph](image)

**Fig. 11** Temporal changes in ammonia nitrogen at different sampling points of the two treatment lines. The dotted sections in the line graph indicate times of no effluent at the sampling point. **Key:** APL - anaerobic pond line, TFL - trickling filter line.

### 4.2.9 Nitrite nitrogen (mg L\(^{-1}\))

The highest concentration of nitrite nitrogen (0.7 mg L\(^{-1}\)) was recorded along the anaerobic pond line in the facultative pond in January 2004 and the lowest (0.01 mg L\(^{-1}\)) was observed a number of times in different treatment units along both lines (Fig. 12). Along the anaerobic pond line, the highest mean concentration of nitrite nitrogen was
recorded in the facultative pond from where there was a decline in nitrite nitrogen up to
the final maturation pond, followed by an increase in the rock filter and a decline in the
grass plot (Table 4). Along the trickling filter line nitrite nitrogen values fluctuated along
the treatment units. With exception of the facultative pond of the anaerobic pond line, the
nitrite nitrogen levels in the treatment units along the anaerobic pond line were lower
than those recorded for similar units of the trickling filter line (Fig. 12). The changes in
nitrite nitrogen did not appear to follow any distinct trend throughout the study period.
There was a significant difference ($P = 0.003, n = 14$) in the mean levels of nitrite
nitrogen noted in the final effluent from the two treatment lines.

Table 4 Means and the ranges of nitrite nitrogen, nitrates nitrogen and total nitrogen at different
sampling point along the two treatment lines.

<table>
<thead>
<tr>
<th>TL</th>
<th>SP</th>
<th>NO$_3$N (mg L$^{-1}$)</th>
<th>NO$_2$N (mg L$^{-1}$)</th>
<th>Total nitrogen (mg L$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Mean</td>
<td>Range</td>
<td>Mean</td>
</tr>
<tr>
<td>APL</td>
<td>IN</td>
<td>0.04</td>
<td>0.02 - 0.08</td>
<td>0.39</td>
</tr>
<tr>
<td></td>
<td>AP</td>
<td>0.06</td>
<td>0.01 - 0.20</td>
<td>0.12</td>
</tr>
<tr>
<td></td>
<td>FP</td>
<td>0.35</td>
<td>0.11 - 0.70</td>
<td>0.53</td>
</tr>
<tr>
<td></td>
<td>MP</td>
<td>0.06</td>
<td>0.01 - 0.15</td>
<td>0.57</td>
</tr>
<tr>
<td></td>
<td>RF</td>
<td>0.07</td>
<td>0.01 - 0.20</td>
<td>0.18</td>
</tr>
<tr>
<td></td>
<td>GP</td>
<td>0.05</td>
<td>0.01 - 0.30</td>
<td>0.26</td>
</tr>
<tr>
<td>TFL</td>
<td>TF</td>
<td>0.12</td>
<td>0.02 - 0.43</td>
<td>0.43</td>
</tr>
<tr>
<td></td>
<td>FP</td>
<td>0.08</td>
<td>0.02 - 0.43</td>
<td>0.46</td>
</tr>
<tr>
<td></td>
<td>MP</td>
<td>0.20</td>
<td>0.31 - 0.53</td>
<td>0.54</td>
</tr>
<tr>
<td></td>
<td>RF</td>
<td>0.10</td>
<td>0.01 - 0.37</td>
<td>0.37</td>
</tr>
<tr>
<td></td>
<td>GP</td>
<td>0.19</td>
<td>0.01 - 0.60</td>
<td>0.46</td>
</tr>
</tbody>
</table>

Key: TL - treatment lines, SP - sampling point, IN - inlet, AP - anaerobic pond, FP - facultative
pond, MP - maturation pond, RF - rock filter, GP - grass plot, APL - anaerobic pond line,
PRPTU - percent reduction per treatment unit, CPR - cumulative percent reduction.
Fig. 12 Temporal changes in nitrite nitrogen recorded at each of the two treatment lines sampling point. The dotted sections in the line graph indicate times of no effluent at the sampling point. Key: APL - anaerobic pond line, TFL - trickling filter line.

4.2.10 Nitrate nitrogen (mg L\(^{-1}\))

The highest concentration of nitrate nitrogen along the anaerobic pond line (2.1 mg L\(^{-1}\)) in facultative pond effluent in March 2004 and the lowest (0.01 mg L\(^{-1}\)) determined in the effluent from grass plot of the same line sometimes in January 2004 (Fig. 12). Along the anaerobic pond line, low levels of nitrate nitrogen were recorded in the anaerobic pond effluent while higher levels were measured at the maturation ponds (Table 4). Nitrate nitrogen levels along the trickling filter line were generally higher than those observed in the anaerobic pond line (Table 4). Changes in nitrate nitrogen concentration did not appear to follow any distinct seasonal pattern during the study period. Mean
concentration of nitrate nitrogen in the final effluent of trickling filter line was significantly higher (0.46 NO₃⁻N mg L⁻¹) than that recorded in the anaerobic pond line (0.26 NO₃⁻N mg L⁻¹) at P = 0.023 and n = 13. The concentration of nitrate nitrogen in the final effluent from both treatment lines was lower than the set limit of 5.0 mg L⁻¹.

Fig. 13 The temporal changes in nitrate nitrogen recorded at each sampling point. The dotted sections in the line graph indicate times of no effluent at the sampling point.

Key: APL - anaerobic pond line, TFL - trickling filter line.

4.2.11 Total nitrogen (mg L⁻¹)

The highest concentration of total nitrogen (120.2 mg L⁻¹) was recorded in the effluent from the anaerobic pond in November 2003 while the lowest (0.08 mg L⁻¹) was observed in the effluent from the maturation pond of the same line (Table 4). There was an increase in the concentration of total nitrogen noted in the effluents from anaerobic pond (Table 4) followed by a sharp decline up to the maturation pond from where a
progressive increase was noted in the effluent from the rock filter and the grass plot along the trickling filter line (Table 4). There was a slight decrease in the levels of total nitrogen during the wet season months of August 2003, January and March of 2004 (Fig. 14). Anaerobic pond and the trickling filter lines achieved 93.44 and 82.02 % reductions in the levels of total nitrogen (Table 4). Generally, higher levels of total nitrogen were recorded in the treatment units of trickling filter line. The concentration of total nitrogen recorded in the final effluent from the trickling-filter line (12.23 mgL⁻¹) was significantly higher (P < 0.05, n = 13) than that recorded in the final effluent from the anaerobic pond line (4.46 mg L⁻¹). Mean levels of total nitrogen (Table 4) recorded in the final effluent from trickling filter line were much higher than the set limit of 5.0 mg L⁻¹.

Fig. 14 Temporal changes in total nitrogen (TN) at various sampling points of the two treatment lines. Dotted sections indicate time of no effluent flow at the sampling point. Key: APL - anaerobic pond line, TFL - trickling filter line.
4.3 Biological properties

4.3.1 MPN of faecal coliforms 100 ml⁻¹

There was a general decrease in densities of faecal coliforms up to the third maturation pond followed by an increase in the effluent from the rock filter and the grass plot along the anaerobic pond line. Along the trickling filter line, the decline in counts of faecal coliforms densities continued up to the rock filter from where an increase was noted in the grass plot effluent (Table 5). The highest removal of faecal coliforms was observed in the facultative pond (99.99 %) and final maturation pond (99.98 %) of anaerobic pond and trickling filter lines respectively (Table 5). The trickling filter treatment unit achieved a low faecal coliforms removal (12.92 %) as compared to 67.09 % achieved by the anaerobic pond treatment line (Table 5). Overall faecal coliforms reduction achieved by anaerobic pond and trickling filter lines were 99.999 % and 99.719 % respectively (Table 5). Mean levels of faecal coliforms observed in the final maturation pond effluent of the anaerobic pond line (261 / 100 ml) was significantly lower (P = 0.007, n = 22) than that observed at the trickling filter line final maturation pond (6500 / 100 ml) and lower than the set limit of 1000 / 100 ml for the treatment plant (Mangat et al., 1996). An increase in mean levels of faecal coliforms was noted in the rock filter (454 / 100 ml) and the grass plot (1077 / 100 ml) effluents of the anaerobic pond line, and the grass plot (14870 / 100 ml) of the trickling filter line (Table 5). Mean counts of faecal coliforms (1077 / 100 ml) in the final effluent of anaerobic pond line was significantly lower (P = 0.003, n = 14) than 14871 / 100 ml obtained in the final effluent of the trickling filter line. Occasionally, the anaerobic pond line's final effluent could stagnate at the outlet but at other times there used to be no effluent at all (Fig. 15).
Table 5 Means, ranges and % reduction in faecal coliforms and *Salmonella* spp at each sampling point along anaerobic and trickling filter treatment lines

<table>
<thead>
<tr>
<th>SP</th>
<th>TL MPN of faecal coliforms 100 ml⁻¹</th>
<th><em>Salmonella</em> spp counts ml⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean (×10⁴)</td>
<td>Range (×10⁴)</td>
</tr>
<tr>
<td>IN</td>
<td>13218</td>
<td>9200 - 16000</td>
</tr>
<tr>
<td>APL</td>
<td>4350</td>
<td>1800 - 9200</td>
</tr>
<tr>
<td>AP</td>
<td>0.1909</td>
<td>0.009 - 0.79</td>
</tr>
<tr>
<td>FP</td>
<td>0.0261</td>
<td>0.004 - 0.09</td>
</tr>
<tr>
<td>MP</td>
<td>0.0454</td>
<td>0.008 - 0.23</td>
</tr>
<tr>
<td>RF</td>
<td>0.1077</td>
<td>0.01 - 0.26</td>
</tr>
<tr>
<td>GP</td>
<td>1.4871</td>
<td>0.06 - 4.6</td>
</tr>
<tr>
<td>PRPTU CPR Mean</td>
<td>Range</td>
<td>PRPTU CPR</td>
</tr>
<tr>
<td>TFL</td>
<td>11510</td>
<td>5400 - 16000</td>
</tr>
<tr>
<td>AP</td>
<td>3602</td>
<td>220 - 9200</td>
</tr>
<tr>
<td>MP</td>
<td>0.6500</td>
<td>0.02 - 2.2</td>
</tr>
<tr>
<td>RF</td>
<td>0.2994</td>
<td>0.05 - 0.8</td>
</tr>
<tr>
<td>GP</td>
<td>1.8471</td>
<td>0.06 - 4.6</td>
</tr>
</tbody>
</table>

Key: TL - treatment lines, SP - sampling point, IN - inlet, AP - anaerobic pond, FP - facultative pond, MP - maturation pond, RF - rock filter, GP - grass plot, APL - anaerobic pond line, PRPTU percent reduction per treatment unit, CPR - cumulative percent reduction.

**Fig. 15** Temporal changes in faecal coliforms (FC) at different treatment units along the two treatment systems. Dotted sections indicate time of no effluent flow at the sampling point.

**Key:** APL - anaerobic pond line, TFL - trickling filter line, FC - faecal coliforms.
4.3.2 Salmonella spp counts ml\(^{-1}\)

Highest counts of *Salmonella* spp (30000 ml\(^{-1}\)) were recorded in the influent on a number of occasions in 2004 while the lowest (10 ml\(^{-1}\)) was recorded a number of times in the third maturation pond and rock filter of anaerobic pond line (Fig 16). The lowest mean counts of *Salmonella* spp along both treatment lines were achieved at final maturation ponds (Table 5). There was a general decrease in the number of *Salmonella* spp up to the maturation pond followed by a progressive increase in the effluent from the rock filters and grass plots of both treatment lines (Table 5). An increase in *Salmonella* spp counts of the influent was noted during low rain months of October, December (2003) and February 2004 (Fig. 16). Anaerobic pond and trickling filter lines achieved an overall *Salmonella* spp reduction of 99.98 and 98.55 % respectively at the third maturation pond. The maturation ponds of the two treatment systems achieved the highest reduction of *Salmonella* spp (Table 5). An increase in *Salmonella* spp counts in the effluent from the rock filter and the grass plot reduced the overall reduction of *Salmonella* spp counts in the anaerobic pond and trickling filter lines to 98.89 and 92.37 % respectively (Table 5). Mean counts of *Salmonella* spp at maturation ponds were significantly different (P = 0.0003, n = 15) with a higher count being realized along trickling filter line. Mean counts of *Salmonella* (1683 counts ml\(^{-1}\)) recorded in the final effluent from trickling filter was significantly higher (P = 0.006, n = 11) than the mean count (244 counts ml\(^{-1}\)) recorded in the anaerobic pond line (Table 5).
Fig. 16 Temporal changes in *Salmonella* spp counts at different sampling points. Dotted sections indicate time of no effluent flow at the sampling point.

**Key:** APL – anaerobic pond line, TFL – trickling filter line

### 4.3.3 *Shigella* spp counts ml⁻¹

The highest counts of *Shigella* spp (9000 counts ml⁻¹) was recorded in the influent in February 2004. On a number of occasions, no *Shigella* spp was detected at third maturation pond of anaerobic pond line (Fig 17). There was a general decrease in *Shigella* spp counts up to the maturation pond followed by a progressive increase in the effluent from rock filters and grass plots of both treatment lines (Table 6). An increase in *Shigella* spp counts ml⁻¹ was noted during drier month of November and December of 2003 and February 2004 (Fig. 17). The facultative pond of the anaerobic pond line and the final maturation pond of the trickling filter line achieved the highest percent removal of 89.59 % and 95.89 % respectively (Table 6). The anaerobic pond and trickling filter
lines achieved an overall percent reduction of 99.87 and 98.81 respectively, at the third maturation. An increase in *Shigella* spp counts in the effluent from the rock filter and the grass plot reduced the percentage reduction of the anaerobic pond and trickling filter lines to 98.35 and 90.81 % respectively (Table 6). Mean counts of *Shigella* spp (400 counts ml⁻¹) recorded in the final effluent from trickling filter line were significantly higher (P < 0.001, n = 11) than that recorded in the final effluent of anaerobic pond line.

**Table 6** Means, ranges and % reductions of *Shigella* spp and *Vibrio cholerae* counts ml⁻¹ recorded at each sampling point along the anaerobic pond line and the trickling filter lines.

<table>
<thead>
<tr>
<th>TL</th>
<th>SP</th>
<th><em>Shigella</em> spp counts ml⁻¹</th>
<th><em>Vibrio cholerae</em> counts ml⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
<td>Range</td>
<td>PRPTU</td>
</tr>
<tr>
<td>IN</td>
<td>4353.33</td>
<td>1100 - 9000</td>
<td>-</td>
</tr>
<tr>
<td>APL</td>
<td>460.00</td>
<td>100 - 900</td>
<td>89.43</td>
</tr>
<tr>
<td>FP</td>
<td>47.86</td>
<td>10 - 90</td>
<td>89.59</td>
</tr>
<tr>
<td>MP</td>
<td>5.80</td>
<td>0 - 11</td>
<td>87.86</td>
</tr>
<tr>
<td>RF</td>
<td>13.18</td>
<td>5 - 30</td>
<td>-127.24</td>
</tr>
<tr>
<td>GP</td>
<td>63.58</td>
<td>15 - 170</td>
<td>-382.40</td>
</tr>
<tr>
<td>TFL</td>
<td>3593.33</td>
<td>1500 - 8000</td>
<td>17.45</td>
</tr>
<tr>
<td>FP</td>
<td>1253.33</td>
<td>200 - 3000</td>
<td>65.12</td>
</tr>
<tr>
<td>MP</td>
<td>51.80</td>
<td>15 - 90</td>
<td>95.89</td>
</tr>
<tr>
<td>RF</td>
<td>142.00</td>
<td>60 - 300</td>
<td>-174.15</td>
</tr>
<tr>
<td>GP</td>
<td>400.13</td>
<td>12 - 800</td>
<td>-181.78</td>
</tr>
</tbody>
</table>

**Key:** TL - treatment lines, SP - sampling point, IN - inlet, AP - anaerobic pond, FP - facultative pond, MP - maturation pond, RF - rock filter, GP - grass plot, APL - anaerobic pond line, PRPTU - percent reduction per treatment unit, CPR - cumulative percent reduction.
Fig. 17 Temporal changes of *Shigella* spp counts observed at each sampling point of the two treatment lines. The dotted sections in the line graph indicate times of no effluent at the sampling point.

**Key:** APL – anaerobic pond line, TFL – trickling filter line.

### 4.3.4 *Vibrio cholerae* counts ml⁻¹

The highest counts of *Vibrio cholerae* (6000 counts ml⁻¹) was recorded in the inlet on sometimes in January and in February 2004 while the lowest was observed a number of times in the effluent from third maturation and the rock filter along the anaerobic pond line (Fig. 18). The facultative pond of the anaerobic pond line and the final maturation pond of the trickling filter line achieved the highest percent removal 98.26 % and 94.03 % respectively (Table 6). The lowest mean levels were achieved at maturation ponds along both lines (Table 6). A mean *Vibrio cholerae* value of 2 counts ml⁻¹ realized at third maturation pond of anaerobic ponds line was significantly lower (P = 0.003, n = 15) than 23 counts ml⁻¹ recorded along the trickling filter line. The greatest reduction (90.9
% in *Vibrio cholerae* counts was noted in the anaerobic ponds effluent (Table 6). There was an increase in *Vibrio cholerae* counts in the rock filter and grass plot effluents of the two treatment lines (Table 6). The mean level of *Vibrio cholerae* (65 counts ml\(^{-1}\)) in the final effluent from the trickling filter line was significantly higher (P = 0.023, n = 10) than 15 counts ml\(^{-1}\) recorded in the anaerobic pond line.

Fig. 18 Temporal changes in the levels of *Vibrio cholerae* observed at different sampling points along the two treatment lines. The dotted sections in the lines indicate times of no effluent at the sampling point.

Key: APL – anaerobic pond line, TFL - trickling filter line.

4.3.5 Chlorophyll *a* (mg m\(^{-3}\))

There was an increase in the mean levels of chlorophyll *a* up to first maturation pond followed by a decline along the anaerobic pond line (Fig. 19). In the trickling filter line, an increase in the levels of chlorophyll *a* in successive ponds was noted (Table 7). There
was no significant difference in the mean concentration of chlorophyll $a$ in the facultative and first maturation ponds of the trickling filter and anaerobic pond lines ($P = 0.582, n = 20$; $P = 0.488, n = 20$ respectively). Mean concentration of chlorophyll $a$ in the second and third maturation ponds of the trickling filter line were significantly higher ($P = 0.008, n = 20$; $P = 0.001, n = 20$ respectively) than those recorded in the same ponds along the anaerobic pond line (Table 7).

![Fig. 19 Temporal changes in chlorophyll $a$ concentration at facultative and maturation ponds of the anaerobic pond and trickling filter treatment lines. The dotted sections indicate times of no effluent at the sampling point. Key: APL - anaerobic pond line, TFL - trickling filter line, Chl. $a$ – chlorophyll $a$.](image)

### 4.3.6 Algal counts (single cells or colonies) ml$^{-1}$

Ponds along the anaerobic pond line were dominated by members of the Euglenophyta while ponds in the trickling filter line, the Cyanophyta (mostly *Arthrophira*) were dominant. Along the anaerobic pond line, a gradual increase in the phytoplankton counts up to second maturation pond followed by a decrease in the third maturation was noted. However, in the trickling filter line there was a progressive increase in phytoplankton density up to the third maturation pond. Although the species identified along the
anaerobic pond line were common in all the ponds, in the third maturation pond the species were more abundant and diverse.

Table 7 Means, ranges of phytoplankton (counts ml⁻¹) and chlorophyll a (g m⁻³) observed in the facultative and maturation pond of the two treatment lines.

<table>
<thead>
<tr>
<th>TL</th>
<th>SP</th>
<th>Chlorophyll a (g m⁻³)</th>
<th>Phytoplankton counts (cells / colonies) ml⁻¹ (10⁶)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Mean</td>
<td>Range</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mean</td>
<td>Range</td>
</tr>
<tr>
<td>APL</td>
<td>FP</td>
<td>1.10</td>
<td>0.13 – 3.24</td>
</tr>
<tr>
<td>1MP</td>
<td></td>
<td>1.47</td>
<td>0.38 – 2.40</td>
</tr>
<tr>
<td>2MP</td>
<td></td>
<td>1.34</td>
<td>0.20 – 2.14</td>
</tr>
<tr>
<td>3MP</td>
<td></td>
<td>0.54</td>
<td>0.02 – 1.75</td>
</tr>
<tr>
<td>TFL</td>
<td>FP</td>
<td>0.88</td>
<td>0.08 – 5.26</td>
</tr>
<tr>
<td>1MP</td>
<td></td>
<td>1.66</td>
<td>0.20 – 3.62</td>
</tr>
<tr>
<td>2MP</td>
<td></td>
<td>2.71</td>
<td>0.20 – 6.35</td>
</tr>
<tr>
<td>3MP</td>
<td></td>
<td>3.60</td>
<td>0.35 – 9.63</td>
</tr>
</tbody>
</table>

Key: TL - treatment line, SP - sampling point, APL - anaerobic pond treatment line, FP - facultative pond, 1MP - 1st maturation pond, 2MP - 2nd maturation pond, 3MP - 3rd maturation pond, TFL - trickling filter treatment line.

Fig. 20 Phytoplankton density changes in the facultative and maturation ponds of the two treatment lines. The dotted sections in the line graph indicate times of no effluent at the sampling point.

Key: APL - anaerobic pond line, TFL - trickling filter line.

4.3.7 Algal species in maturation and facultative ponds

In terms of numbers, species of *Euglena, Phacus* (Euglenophyta) and *Scenedesmus* (Chlorophyta) dominated the algal community of the facultative and maturation ponds of
the anaerobic pond line (Table 8). The alga *Arthrospira* was abundant in the ponds along the trickling filter line but absent in ponds along the anaerobic pond line except for the facultative pond where it was occasionally present in very low numbers. Most algal species identified in different ponds were present throughout the study period.

### Table 8 The algal species in the facultative and maturation ponds of the two treatment lines

<table>
<thead>
<tr>
<th>Phylum</th>
<th>Species</th>
<th>APL</th>
<th>TFL</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>FP</td>
<td>1MP</td>
</tr>
<tr>
<td><strong>Euglenophyta</strong></td>
<td><em>Euglena</em></td>
<td>p****</td>
<td>p****</td>
</tr>
<tr>
<td></td>
<td><em>Phacus</em></td>
<td>p***</td>
<td>p***</td>
</tr>
<tr>
<td></td>
<td><em>Lepocinclis</em></td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td><strong>Chlorophyta</strong></td>
<td><em>Scenedesmus</em></td>
<td>p**</td>
<td>p**</td>
</tr>
<tr>
<td></td>
<td><em>Sphaerocystis</em></td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td></td>
<td><em>Dictyosphaerium</em></td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td></td>
<td><em>Micractinium</em></td>
<td>A</td>
<td>P</td>
</tr>
<tr>
<td></td>
<td><em>Actinastrum</em></td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td></td>
<td><em>Volvox</em></td>
<td>P</td>
<td>P</td>
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<tr>
<td></td>
<td><em>Botryococcus</em></td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td></td>
<td><em>Rivularia</em></td>
<td>P</td>
<td>P</td>
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<tr>
<td></td>
<td><em>Coelastrum</em></td>
<td>P</td>
<td>P</td>
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<tr>
<td></td>
<td><em>Chroococcus</em></td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td></td>
<td><em>Pediasastrum</em></td>
<td>A</td>
<td>P</td>
</tr>
<tr>
<td></td>
<td><em>Crucigenia</em></td>
<td>P</td>
<td>P</td>
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<tr>
<td></td>
<td><em>Staurastrum</em></td>
<td>P</td>
<td>P</td>
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<tr>
<td></td>
<td><em>Ankistrodesmus</em></td>
<td>P</td>
<td>P</td>
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<tr>
<td></td>
<td><em>Tetrastrum</em></td>
<td>P</td>
<td>P</td>
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<tr>
<td></td>
<td><em>Tetraspora</em></td>
<td>P</td>
<td>P</td>
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<td></td>
<td><em>Chlorella</em></td>
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<td>P</td>
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<td><strong>Dinophyta</strong></td>
<td><em>Peridinium</em></td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td><strong>Cyanophyta</strong></td>
<td><em>Arthrospira</em></td>
<td>P</td>
<td>A</td>
</tr>
<tr>
<td></td>
<td><em>Aphanizomenon</em></td>
<td>P</td>
<td>P</td>
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<tr>
<td></td>
<td><em>Gleotrichia</em></td>
<td>P</td>
<td>P</td>
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<tr>
<td></td>
<td><em>Microcystis</em></td>
<td>P</td>
<td>P</td>
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<tr>
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<td><em>Oscillatoria</em></td>
<td>P</td>
<td>P</td>
</tr>
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<td><strong>Bacillariophyta</strong></td>
<td><em>Fragilaria</em></td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td></td>
<td><em>Nitzschia</em></td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td></td>
<td><em>Asteroionella</em></td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td></td>
<td><em>Navicula</em></td>
<td>P</td>
<td>P</td>
</tr>
</tbody>
</table>

Key. APL - anaerobic pond line, TFL - trickling filter line, FP - facultative pond, 1MP - 1st maturation pond, 2MP - 2nd maturation pond, 3MP - 3rd maturation pond, P - present, A - absent, P**** - most abundant spp, P*** - second abundant, PP** - third abundant, P* - fourth abundant.
CHAPTER 5

5. DISCUSSION

5.1 Physico-chemical properties

5.1.1 Temperature (°C)

In wastewater biological treatment processes, temperature affects the metabolic rate of microorganisms making the overall influence of temperature on the treatment process important (Pfaefflin & Ziegler, 1992). As temperature rises, the rate of reaction increases (Droste, 1997). Temperature has been regarded as one of the most important factors influencing the efficiency of nitrogen removal because it directly affects the metabolic rate of microorganisms (Gray, 1992; Lai & Lam, 1995). The temperature recorded at various sampling points fell within a suitable range for mesophilic microorganisms (between 20 °C and 40 °C) involved in biological treatment of wastewater. This is important for the efficient performance of a treatment plant and a clear indication that any poor performance of the treatment units within Nakuru town sewage treatment plant cannot be as a result of unfavorable temperatures. The mean temperature at the inlet (23.9 °C) compares well with the mean temperature (23 °C) recorded in 1996 at the same plant (Mangat et al., 1996). Temperature changes seem to have been largely influenced the nature of treatment unit because the lowest mean temperature (22.8 °C) was recorded in the anaerobic pond which is the deepest of all the ponds (3.6 m) and the highest (24.7 °C) in the rock filter where the sewage flows through stones well heated up by the sun, raising its temperature (Table 1).

5.1.2 Conductivity (μS cm⁻¹)

A narrow range influent of conductivity for most of the study period (Fig. 5) suggests that the sewage was mostly domestic (Guo, 1995). Support for this position is provided by a near neutral median pH (7.3) recorded at the inlet that is characteristic of typical domestic
sewage. It is suspected that the slight reduction in conductivities recorded during wet season months of August 2003 and January 2004 was due to infiltration of storm water into sewer lines and increased water supply in Nakuru town. A gradual decline, low overall percent reduction and relatively high conductivities recorded in treatment units along the trickling filter line as compared to anaerobic pond line (Fig. 5) is an indication of delayed mineralization of organic matter along the treatment system. This is a sign of a lower efficiency, a fact supported by the high BOD values recorded along the trickling filter line treatment units (Table 2). This raises the possibility of organic overloading of the treatment units hence lowering the quality of the effluent produced. Organic overloading is known to lower the quality of sewage effluent (Miguel, 2003). The significant difference in the levels of conductivity in the final effluent from the two treatment lines supports existence of a real difference in the efficiency of the two treatment lines.

5.1.3. pH

The median pH (7.3) of the influent recorded in this study compares well with a median pH of 7.29 recorded at Honolulu East wastewater treatment plant during a study on the efficiency of the plant (Guo, 1995). A median pH of 7.4 recorded in the anaerobic pond fell within the recommended value of above 7 for effective performance of the pond (Miguel, 2003). The pH of above 7 is important in order to permit growth of methanogenic bacteria involved in biodegradation of organic matter (Zendher et al., 1982). The growth of methanogenic bacteria is inhibited by an acidic environment (Trivedy & Goel, 1989).

A neutral influent further supports the observation that the sewage treatment plant only receives domestic sewage only. The neutral nature of the raw domestic wastewaters can be attributed to the presence of neutral components mainly fats, proteins, carbohydrates,
amino sugars, amides, and detergents, which are common in domestic sewage (Guo, 1995). A near neutral pH facilitates efficient biological treatment of wastewater (Metcalf et al., 1995). The pH of the effluent recorded at the facultative and the maturation ponds fell within the range of pH 6.0 – 9.0 that is suitable for algal growth (Trivedy & Goel, 1989). Algal growth is important because during their photosynthesis, they provide the oxygen required by the facultative anaerobes and aerobic microorganisms in the facultative and maturation ponds. The algae for their photosynthesis utilize carbon dioxide produced by the bacteria. The high median pH of between 7.9 and 9.3 recorded in the facultative and maturation ponds along both treatment lines (Table 1) is an indication of high algal activity within the ponds which is important for the aeration of the ponds (Kinney & Roman, 1998; Isaac, 2000). High pH minimizes chances of microbial toxicity from gases and metallic salts in the wastewater (Mervin, 2001). The slightly lower pH readings made during the wet season could have been as a result of reduced algal activity due to cloudiness that characterizes the wet season leading to reduced solar irradiation.

5.1.4 Total alkalinity (mg CaCO₃ L⁻¹)

Total alkalinity reduction of 43 % and 18 % achieved at the third maturation pond in the anaerobic pond and the trickling filter lines respectively were lower than an overall reduction of 48.4 % achieved at Honolulu East wastewater treatment plant during a study on the plant’s efficiency (Guo, 1995). Total alkalinity of the Honolulu East wastewater treatment plant influent was 258 mg CaCO₃ L⁻¹ (Guo, 1995) while that of Nakuru town sewage treatment was 299 mg CaCO₃ L⁻¹ (Table 1). Although the difference in percent reduction can be partly attributed to different levels of total alkalinity in the influent of the two treatment plants Nakuru town sewage treatment plant appears to be less efficient in alkalinity reduction as compared to Honolulu east wastewater treatment plant. The lower efficiency is supported by the high mean values of total alkalinity recorded in the
final effluent from the anaerobic pond line (159.2 mg CaCO₃ L⁻¹) and the trickling filter line (242.5 mg CaCO₃ L⁻¹) as compared to 133 mg CaCO₃ L⁻¹ recorded in the final effluent from the Honolulu east wastewater treatment plant (Guo, 1995). The low total alkalinity percent reduction achieved at Nakuru town sewage treatment plant can be partly attributed to denser algal population in the Nakuru town sewage stabilization pond reflected in high algal biomass (0.54 and 3.6 g m⁻³ in the final maturation pond of anaerobic pond line and trickling filter line respectively) recorded in these ponds as compared to 0.1 g m⁻³ observed at Akuse wastewater treatment plant (Guo, 1995). Carbon dioxide produced by the algae during respiration especially at night could be responsible for the low percent reduction in that carbon dioxide dissolution in water forms carbonate ions that are partly responsible for alkalinity in water (Trivedy & Goel, 1989). The high reduction of total alkalinity achieved by the maturation ponds along the two treatment systems can be greatly attributed to the utilization of carbon dioxide and other compounds that contribute to total alkalinity by phytoplankton and other organisms in the water for their metabolic processes. Mean total alkalinity of 339.6 mg CaCO₃ L⁻¹ recorded in anaerobic ponds effluent during the study indicated a well-buffered anaerobic pond evident from a fairy narrow range of pH (1.3) observed during the study period (Table 1). This is an important factor in wastewater treatment and especially in the anaerobic ponds where the degradation of organic matter involves two groups of bacteria, the non-methanogenic and methanogenic bacteria which co-exist. The methanogenic bacteria can only operate within a narrow range of pH and a sudden increase or decrease in pH would cause a failure of the anaerobic ponds by inhibiting the growth of the methanogenic bacteria (Trivedy & Goel, 1989; Miguel, 2003). The significant difference (P = 0.001, n = 13) in the mean total alkalinity in the final effluent is a clear indication that the anaerobic pond line was more efficient than the trickling filter line.
5.1.5 Biological oxygen demand (BOD$_s$, mg O$_2$ L$^{-1}$)

BOD of wastewater is the key parameter used by engineers to design wastewater treatment plants (Karel, 1983) and it is for this reason that BOD was used as one of the key physico-chemical properties for evaluating the efficiency of Nakuru town sewage treatment plant. BOD$_s$ test is conventionally used in the BOD determination (Sawyer et al., 1994). The lowest BOD recorded in August 2003 followed heavy rainfall possibly resulting from storm water entering the sewer line through manholes not well raised above the ground level hence diluting the sewage. The high BOD recorded in December 2003 and January 2004 can be attributed to an upsurge of organic loading as this coincided with the period when many residents empty their septic tanks. The high BOD removal achieved by the facultative ponds along the anaerobic pond and the trickling filter lines systems is largely due to their long retention period (21 days) as compared to other treatment units, for example nine days for maturation ponds (Mangat et al., 1996).

An influent mean BOD of 857 mg O$_2$ L$^{-1}$ recorded at Nakuru town sewage treatment plant was close to 800 mg O$_2$ L$^{-1}$ recorded between January and May 1996 at the same plant (Mangat et al., 1996). The slight increase in BOD could have been as a result of a combination of an increase in human population within Nakuru town and water shortage that was characteristic of Nakuru town. The influent BOD of 857 mg O$_2$ L$^{-1}$ recorded in this study is much lower than < 100 mg O$_2$ L$^{-1}$ recorded at Akuse wastewater treatment plant in Ghana (Isaac, 2000) and 63 mg O$_2$ L$^{-1}$ recorded at East Honolulu (Guo, 1995).

The difference in BOD$_s$ could be due to different life styles and water availability in the different localities, population, design, weather, etc. The influent BOD$_s$ value is also higher than the expected range of between 200 and 600 mg O$_2$ L$^{-1}$ according to Atlas (1989). Domestic water shortage is common in Nakuru and is most likely responsible for the high BOD recorded at Nakuru town sewage treatment plant.
The results obtained from this study indicate that the treatment units within the treatment plant do not meet the design standards with regard to BOD reduction (Mangat et al., 1996). The mean BOD of the effluent from anaerobic pond was higher (299.38 mg O$_2$ L$^{-1}$, Table 2) than the expected BOD (206.0 mg O$_2$ L$^{-1}$). In terms of percent reduction of BOD, the anaerobic ponds BOD reduction of the influent by 65.1 % closely agrees with Van Eck's (1965) observation that the anaerobic ponds are able to reduce the BOD of domestic sewage by 40 to 60 % at 20 °C, Cairncross & Feacham (1983) report of BOD reduction of 70 to 85 % by anaerobic pond and Miguel’s (2003) report of a 60 % BOD reduction at 20 °C for a well designed anaerobic pond. The mean temperature of the influent observed at Nakuru town wastewater treatment plant was 23.86 °C (Table 1) and this could have contributed to the 65 % reduction of BOD observed. The overall influent BOD reduction 94.4 % achieved by the trickling filter line was slightly lower than that recorded at Honolulu east wastewater treatment plants of 98.40 % (Guo, 1995) while the 98.0 % reduction achieved by the anaerobic line (Table 2) compares well with 98.40 % achieved at the Honolulu east wastewater treatment plant. The overall BOD reduction achieved by the anaerobic pond line (97 %) and the trickling filter line (79.0 %) at final maturation ponds were much higher than the 64 % reduction recorded at Akuse wastewater treatment plant in Ghana (Isaac, 2000). The difference in overall BOD percent reductions may have been due to difference wastewater stabilization pond design, prevailing weather conditions at the various localities and maintenance state of the stabilization ponds used for various studies.

The results of this study agrees with Atlas (1989) that a well designed and operated biological treatment units can reduce the BOD of wastewater by 80 to 90 % producing an effluent with a BOD of less than 20 mg O$_2$ L$^{-1}$. However, the effluent BOD was higher than the standard set by the plant designers of 10.0 mg O$_2$ L$^{-1}$. The 97 and 79 % overall BOD reductions achieved by anaerobic and trickling filter lines at maturation ponds
agrees closely with Arceivala (1981) who found out that wastewater stabilization ponds can give BOD removal efficiencies greater than 70\%.

The failure or poor performance of a treatment unit can affect the performance of the next unit (WPCF, 1928; Mara & Pearson, 1998). This was evident in Nakuru town sewage treatment plant where the mean BOD of the anaerobic pond effluent (299 mg O$_2$ L$^{-1}$) was higher than the expected value of 206 mg O$_2$ L$^{-1}$ with a similar trend of higher BOD values than the expected being recorded at all the subsequent treatment units. This stresses the importance of ensuring that each treatment unit is functioning efficiently for effective treatment of the sewage.

The trickling filter reduced the BOD of the influent by 36\%. Compared to the documented 65 to 80\% BOD reductions, the performance of the trickling filter was poor (Atlas, 1989; WPCF, 1928). The poor performance could have been as a result of a lack of a continuous flow of wastewater through the filter bed, allowing the filter bed to dry out. This results in the death of the film-forming microorganisms around the filter media, which are responsible for the biodegradation of the organic matter in the wastewater. Once the microorganisms die, the filter bed must be re-seeded with microorganisms for the filter to start functioning again. During the study period, the trickling filter would remain dry for many hours and sometimes days and no re-seeding was ever done. The effluent from the trickling filter went through a non-functional secondary sedimentation tank (no scooping out of sludge due to lack of an electrical motor) to the facultative pond. Mean BOD of the effluent from facultative, maturation and rock filter along the trickling filter line (206.89, 105.25 and 44.53 mg O$_2$ L$^{-1}$) were all significantly higher (P < 0.05) than those obtained in respective units of the anaerobic pond line (Table 2) further confirming the poor performance of the trickling filter line. High BOD value of
the final effluent would favour the survival of microorganisms, both non-pathogenic and potentially pathogenic (McGarry & Bouthiller, 1965).

5.1.6 Chemical oxygen demand (COD, mg O₂ L⁻¹)

This test evaluates the effects of organic and inorganic waste materials on dissolved oxygen in receiving waters. The COD value does not give an indication of whether the wastewater is degradable biologically and the rate at which biological processes would proceed, hence the rate at which the oxygen would be required in the environment (Trivedy & Goel, 1989). However, it is important to analyze both BOD and COD of a water sample in order to obtain complete information on organic waste pollution. The ratio of BOD to COD of between 0.4 and 0.6 is considered normal in terms of waste composition in most cases (Guo, 1995). A BOD to COD ratio outside this range (0.4 – 0.6) indicates unfavorable growth conditions for the microorganisms involved in biodegradation of organic matter (Guo, 1995). This could be as a result of the presence of toxic compounds inhibitory to microbial growth or a high concentration of organic matter. The influent BOD to COD ratio of 0.68 obtained during this study was very close to the upper limit (0.6) of the accepted normal range, an indication of good environmental condition for microbial growth important for efficient treatment of domestic sewage. The positive correlation between BOD and the COD (r = 0.65; P = 0.054, n = 22) and BOD : COD ratio of 0.68 further confirms that the treatment plant receives domestic sewage only.

A sharp increase in influent COD that is not reflected in the influent’s BOD can indicate the introduction of non-domestic sewage with high content of inorganic matter into the system. The mean COD of the influent recorded in this study (1287.03 mg O₂ L⁻¹) is higher than that observed at East Honolulu wastewater treatment plant (183 mg O² L⁻¹, Guo, 1995). The high COD observed at Nakuru wastewater treatment plants could have
resulted from water scarcity experienced in Nakuru town leading to high concentration of oxidizable organic and inorganic matter. The 92.59 % and 82.90 % reductions achieved at Nakuru town sewage treatment plant by the anaerobic pond and trickling filter lines respectively, were higher than 78.10 % recorded at Honolulu East wastewater treatment plants (Isaac, 2000). A high percent reduction achieved at Nakuru town wastewater treatment plant can partly be attributed to the high mean influent COD (1287.27 mg L⁻¹) as compared to 183 mg L⁻¹ observed at East Honolulu wastewater treatment plant. A high overall percent reduction in COD (92.59 %) achieved by the anaerobic pond line as compared to of the trickling filter line (82.90 %, Table 2) is an indication of poorer performance of the trickling filter line. Mean COD of the final effluents from both anaerobic and trickling filter treatment lines (76.6 and 210.8 mg O₂ L⁻¹ respectively) were much higher than the set limit of 30 mg O₂ L⁻¹, a sign of poor performance of the treatment units in general. The results obtained in this study indicate that Nakuru town sewage treatment plant is safe from industrial effluent poisoning due to the fact that the fluctuations in COD were reflected in BOD. This is because an introduction of industrial effluents would lead to an increase in COD but not in BOD.

5.1.7 Total phosphorus (mg L⁻¹)

The source of phosphorus in domestic sewage is mainly detergents and organic matter and its removal by wastewater stabilization ponds is usually low (Mara & Pearson, 1986; Mara et al., 1992). Mean total phosphorus recorded at the inlet (16.2 mg L⁻¹) was closer to the documented total phosphorus of 15.0 mg L⁻¹ in raw sewage (Metcalf et al., 1991; Mangat et al., 1996) but higher than 1.52 mg L⁻¹ recorded at Honolulu East wastewater treatment plant (Guo, 1995). High levels of phosphorus concentration in the raw sewage at Nakuru town sewage treatment plant must be as a result of high organic load in the influent as evidenced by mean BOD value observed (Table 2) which was higher than a mean BOD of 63 mg O₂ L⁻¹ recorded at Honolulu East wastewater treatment plant (Guo,
1995). The overall percent reduction achieved by the anaerobic pond line (71.49%) was much higher than the reported 50% reduction of the initial phosphorus load in the influent (Mara & Pearson, 1986; Mara et al., 1992). A 43% reduction achieved by the trickling filter line was lower than the documented 50% reduction (Mara & Pearson, 1986; Mara et al., 1992) and 23.7% reduction achieved at Honolulu East wastewater treatment plant (Guo, 1995). The difference in percent reduction can be attributed to different influent organic load received at the different treatment plants, hydraulic loading and plant designs. High percent reduction observed at the maturation ponds of the two treatment lines can be attributed to utilization of phosphorus (phosphates) by the phytoplankton and other organisms e.g. bacteria in the wastewater for their metabolic processes. Low percent reduction of total phosphorus by the trickling filter is indicative of the poor performance of the treatment unit. Poor performance of the trickling filter unit means only a small portion of the organic matter in the wastewater is utilized by the microbial community in the filter bed. An increase in total phosphorus observed in the final effluent from the trickling filter line must have been due to poor maintenance of the grass plot or introduction of phosphorus by animals grazing on the grassplot through their wastes. For optimum performance of the grassplot, the grass should be kept short and once cut, the cuttings should not be allowed to decompose on the grass plot. Keeping the grass short would also keep off the grazing animals from the grass plots. This would prevent destruction of the structural components of the grass plot and other treatment units. It is suspected that the slight decline noted during wet season (Fig. 9 & 1) was caused by increased water supply within Nakuru town and infiltration of storm water leading to dilution of the sewage.

The mean concentration of total phosphorus in the final effluent from the anaerobic pond line (4.65 mg L\(^{-1}\)) met the set limit (5 mg L\(^{-1}\)) for discharge to the environment. However, final effluent from the anaerobic pond was in most cases so little such that it
Microorganisms require phosphorus (and other nutrients) for their growth and therefore high phosphorus levels in the final effluent may promote survival of pathogenic microorganisms in the rivulet formed by the sewage effluent as it flows to Lake Nakuru (Tom & Owen, 1973). This may predispose the wildlife to infection by pathogenic microorganisms on drinking water in the wastewater. An efficient treatment plant should be capable of reducing phosphorus to levels (5 mg L\(^{-1}\) for Nakuru town sewage treatment plant) that cannot cause ecological problems in the receiving water bodies. In Western Australia, sewage effluent with phosphorus concentration between 8 to 12 mg L\(^{-1}\) has been considered safe for discharge to the environment. However, this might not be allowed in the next 5 to 10 years because this has resulted in the development of algal blooms that have affected many rivers in the country (Western Australian Department of Environmental Protection, 1998).

5.1.8. **Ammonia nitrogen (mg L\(^{-1}\))**

Urea is the principal form in which man excretes nitrogen and it is rapidly hydrolyzed to ammonia nitrogen (Mara, 1976). The mean ammonia nitrogen recorded in the influent (64.83 mg L\(^{-1}\)) was higher than a reported range of 12 to 50 mg L\(^{-1}\) (Karen, 2003); 5.79 mg L\(^{-1}\) recorded at Honolulu East wastewater treatment plant (Guo, 1995) and 43.0 mg L\(^{-1}\) observed at Delhi advanced integrated wastewater stabilization ponds (Atsushi, 2000).
The percent reduction achieved by anaerobic pond line (93.55 %) and trickling filter line (83.70 %) were lower than the recommended 95 % (Mara et al., 1992) and 100 % reduction achieved at Honolulu East wastewater treatment plant (Guo, 1995) but higher than 78 % reduction achieved by the advanced integrated wastewater pond systems at Delhi (Atsushi, 2001). Differences in design standards, environmental conditions and influent organic load are known to affect the efficiency of a wastewater treatment plant (Cromar et al., 1996; Mara et al., 1992) and could be responsible for the different percent reductions at the two plants. For example, the Honolulu East wastewater treatment plant has conventional treatment units unlike Nakuru Town Sewage Treatment Plant where there is a combination of both conventional treatment system and wastewater stabilization ponds. Conventional treatment systems usually have a shorter retention time as compared to wastewater stabilization ponds. A high level of ammonia nitrogen recorded in the anaerobic pond effluent is indicative of effective hydrolysis of nitrogenous matter (Trivedy & Goel, 1989). A high percent removal of ammonia nitrogen by the maturation ponds along the two treatment lines can be linked to the photosynthetic activity in these ponds. During photosynthesis, algae utilize ammonia and also provide oxygen needed for the oxidation of ammonia to nitrates. Levels of ammonia nitrogen in the final effluent from anaerobic pond (4.18 mg L\(^{-1}\)) and trickling filter lines (10.57 mg L\(^{-1}\)) were much higher than 1.0 mg L\(^{-1}\) recommended by the designers of the sewage plant (Mangat et al., 1996), an indication that there would be an increase in the levels of nitrates in the sewage effluent resulting from oxidation of the ammonia. These would contribute to nutrient enrichment of Lake Nakuru. Although the 10.57 mg L\(^{-1}\) observed in the final effluent from the trickling filter line compares well with 11.0 mg L\(^{-1}\) recorded at Delhi advanced integrated wastewater pond systems (Atsushi, 2001), the effluent from Delhi may not be entering an ecologically sensitive environment as is the case with Nakuru wastewater treatment plant whose final effluent is discharged into the Nakuru National Park. Generally, high levels of ammonia nitrogen recorded in the
treatment units along the trickling filter is another indication of delayed mineralization of the organic matter in the wastewater, a sign of inefficiency.

5.1.9. Nitrite nitrogen (mg L\(^{-1}\))

High levels of nitrite nitrogen recorded in the effluent from the facultative pond along anaerobic pond line is an indication of high rates of nitrification of ammonia in the effluent from anaerobic ponds that also reflect a well-aerated facultative pond, an important factor for efficient performance of the pond. High pH values (Table 1) recorded in the facultative and maturation ponds along the anaerobic line (Table 1) creates an optimal growth condition for nitrifying bacteria and is partly responsible for the high rate of nitrification along this line (Lai & Lam, 1995). High nitrite nitrogen concentration in the maturation pond along the trickling filter line is a sign of poor aeration, a factor that reflects inefficiency (Fig. 12). The high turbidity and undesirable algal population [evident from algal biomass above the range of 0.5 and 2 g m\(^{-3}\) for an optimally operating pond (Miguel, 2003)] that characterized the wastewater along the trickling filter line treatment units must have contributed to the poor aeration state of the ponds.

A sharp decrease in nitrite levels between facultative and third maturation pond of the anaerobic pond line is an indication of good pond aeration a factor that reflects high algal activity necessary for effective treatment of sewage in these ponds. More gradual changes in mean nitrite nitrogen along the trickling filter line is a sign of poor performance by the treatment units along the line. This can largely be attributed to organic overloading as reflected by the high BOD values observed along this line (Table 2). Facultative ponds are usually designed to handle sewage of low organic load (Miguel, 2003). The facultative ponds of Nakuru town sewage treatment plant are meant to handle sewage with an average BOD of 206.0 mg O\(_2\) L\(^{-1}\). An increase in the concentration of
nitrite nitrogen in the effluent from the grass plot of the trickling filter line may have been caused by poor maintenance of the grass plots and introduction of organic matter through animal droppings. High levels of nitrite nitrogen in the maturation pond effluent are indicative of inefficiency in wastewater treatment. In a well-maintained maturation pond, all processes are aerobic (Mara et al., 1992) and most forms of nitrogen are converted to nitrates. Although the algal biomass in the second and third maturation ponds of the trickling filter line was high (Table 7), the dense algal blooms in these ponds must have greatly contributed to the poor aeration status (by limiting the penetration of sun light) evident from the high level of nitrites (Table 4) in the third maturation pond effluent.

5.1.10 Nitrates nitrogen (mg L⁻¹)

Although the levels of nitrates nitrogen in the final effluent from both treatment lines (Table 4) were much lower than 5.0 mg L⁻¹ set standard for discharge to the environment, the mean levels of total nitrogen indicates that there is substantial amount of nitrogenous compounds that are yet to be oxidized (Table 4). An efficiently functioning plant should convert most forms of nitrogen to nitrate. The relatively high levels of nitrates recorded in the trickling filter effluent (Fig. 12) must have been due to the oxidation of ammonia and nitrites under the aerobic condition in the filter. The combination of high turbidity and algal scums which characterized the wastewater along the trickling filter line must have led to poor penetration of sunlight, thus limiting the photosynthetic activities of the algae to the top layer, hence minimizing the rate of oxidation of the reduced forms of nitrogenous compounds. It is likely that the high nitrate levels (and other nutrients analysed in this study) recorded along the trickling filter line were caused by the poor performance of the trickling filter unit which must have caused organic over-loading in facultative and maturation ponds, a fact evident from the high BOD recorded along this line as compared to anaerobic pond line (Table
2). The facultative ponds of the trickling filter line have hippopotamuses, whose wastes and mixing-up of the wastewater could also be partly responsible for the high nitrate nitrogen levels along the line (Table 4). Organic over-loading in facultative and maturation ponds results in a lot of mineralization within the ponds leading to high nutrient levels. Evidence for increased mineralization can be drawn from the high conductivity recorded in the trickling filter line compared to anaerobic pond line (Fig. 4). High nitrate nitrogen levels in the facultative and maturation ponds of trickling filter line must have contributed to the high algal biomass (chlorophyll $a$) levels recorded in this line as compared to anaerobic pond line (Fig. 18). Although nitrate nitrogen is required by algae and bacteria for their diverse metabolic process, excess nitrates in waste stabilization ponds can cause undesirable algal blooms that are capable of interfering with the efficiency of the ponds (Miguel, 2003). A healthy algal bloom is characterized by a dark green colour (Atsushi, 2000) although occasionally they can turn red or pink due to presence of purple sulphide-oxidizing photosynthetic bacteria (Mara & Pearson, 1986).

5.1.1 Total nitrogen (mg L$^{-1}$)

The mean concentration of total nitrogen recorded in the influent (68.03 mg L$^{-1}$) was higher than 53.9 mg L$^{-1}$ observed at Delhi Advanced Integrated Wastewater Stabilization Ponds (Atsushi, 2001) an indication of high organic load in the influent of Nakuru town sewage treatment plant as compared to Delhi treatment plant. Overall nitrogen reduction achieved by anaerobic pond (93.4 %) and trickling filter lines (82.0 %) were higher than the recommended removal of 80 % in the Wastewater Design Manual for East Africa (Mara et al., 1992). Although the 82.3 % reduction achieved by the trickling filter line was higher than the 80 % removal, the fact that the effluent from Nakuru town sewage treatment plant is discharged into a sensitive environment (Mara et al., 1976), makes it important to improve the performance of the treatment units so as to achieve a higher
reduction. Increase in the levels of total nitrogen in the anaerobic pond can be as a result of accumulation of easily hydrolyzable nitrogenous wastes as is evident from the high concentration of ammonia observed in this pond.

Total nitrogen gives the amount of nitrogenous organic and inorganic matter in the wastewater and its potential to support microbial growth. Although the Nakuru town sewage treatment plant does not have a set standard for total nitrogen discharge to the environment, it is important for a treatment plant to reduce the levels of total nitrogen in the final effluent to minimize any chances of eutrophication or creating conducive environment for survival of microorganisms in the receiving water body. In a wastewater treatment plant, the levels of various forms of nitrogen at different treatment stages can reflect the efficiency of the treatment plant. An efficiently functioning plant should convert most of all the forms of nitrogen to nitrate nitrogen. Going by the levels of nitrate nitrogen recorded in the final effluent from both treatment lines (Table 4), the anaerobic pond line was more successful in the conversion of nitrogenous organic compounds to nitrates.

5.2 Bacteriological quality

5.2.1 Faecal coliforms

The quality of water is commonly established by determining microbial presence, especially faecal coliforms (Gray, 1994; DWAF, 1996; USA-EPA, 1999). The mean level of faecal coliforms recorded in the influent \((1.32 \times 10^8 / 100 \text{ ml})\) was within the documented range of \(10^8\) to \(10^9\) which is typical for raw domestic sewage (Yate & Gerba, 1998) but lower than \(9.04 \times 10^6 100 / \text{ ml}\) recorded in raw sewage at Akuse wastewater treatment plant in Ghana (Isaac, 2000). The difference in the levels of faecal coliforms at Nakuru town sewage treatment plant and Akuse sewage treatment plants is possibly as a result of different influent organic load caused by different life-styles and
water availability in the communities served by the two treatment plants. The influent BOD of 100 mg O₂ L⁻¹ observed at Akuse (Isaac, 2000) was much lower than a mean of 857 mg O₂ L⁻¹ recorded in Nakuru sewage treatment plant (Table 2). Water scarcity in Nakuru town could be responsible for the high levels of faecal coliforms as compared to that recorded at Akuse wastewater treatment plant. A comparatively lower density of faecal coliforms recorded along anaerobic pond line as compared to the trickling filter line can be attributed to the low flow (evidenced by little or no final effluent) that characterized the anaerobic pond line (Table 5). Low flow is known to increase the exposure of faecal coliforms and other microorganisms to ultraviolet light increasing their mortality rate (Craig et al., 2002). A high reduction of fecal coliforms (99.99 %) achieved by facultative pond the anaerobic pond line and the maturation pond of the trickling filter line could have resulted from a combination of high nitrite nitrogen levels observed in these ponds, high pH and UV light. Nitrites are toxic to microorganisms (Amdur et al., 1991; Dubusk et al., 1998). A higher percent reduction (99.99 %) of faecal coliforms in the anaerobic pond as compared to 12.92 % achieved by the trickling filter unit could have resulted from the strict anaerobic condition in the anaerobic pond, which is unfavorable for faecal coliforms (facultative anaerobes). The low percent reduction of faecal coliforms by the trickling filter unit can be explained by the aerobic conditions in the trickling filter unit which favours the growth of faecal coliforms involved in the breaking down of organic waste in sewage.

High mean levels of faecal coliforms in the effluent from the third maturation pond of the trickling filter line (6500 / 100 ml) compared to 261 coliforms / 100 ml recorded in the third maturation pond of the anaerobic pond line is an indication of poor performance and is likely to have been due to poor performance of the trickling filter unit (evident from the high BOD value of its effluent), non-functioning secondary clarifier, low rate of sludge removal at the primary clarifier (evident from the bulging sludge) and high flows
that characterized this line during most of the study period. One of the main purposes of a maturation pond is to reduce the number of microorganisms in the sewage effluent (Miguel, 2003). The third maturation pond of the trickling filter line did not effectively achieve their role of reducing the levels of faecal coliforms to the set standards of 1000 / 100 ml for the Nakuru wastewater treatment plant.

Although the 99.99 % reduction of faecal coliforms achieved by the trickling filter line compares well with a report by the London Metropolitan Water Board (Windle, 1965) that indicates that tertiary treatment of sewage effluent from a conventional treatment plant can reduce \textit{E. coli} by 99.50 %, a higher percent reduction of faecal coliforms by Nakuru sewage plant is necessary so as to achieve the recommended level of 1000 faecal coliforms / 100 ml in the final effluent. The results in this study are closely similar to 99.00 % coliforms reduction reported by New Zealand's Auckland Metropolitan Drainage Board (Ernest, 1971) and 99 % coliforms removal rates reported in two ponds connected in series in Israel (Wachs, 1961). The increase in the levels of faecal coliforms and other enteric pathogens isolated (\textit{Salmonella}, \textit{Shigella} spp and \textit{Vibrio cholera}) in the effluent from the rock filter and the grass plot of both treatment lines from 261 to 1077 / 100 ml along anaerobic pond line and from 6500 to 14871 / 100 ml along trickling filter line (Table 5) was possibly due to introduction of organic matter into the rock filters and grass plots in the form of wildlife droppings, decomposing algae and grass that boosts the survival of the microorganisms by providing the nutrients the microorganisms need for their growth (Mervin, 2001). During the study period the rock filter and the grass plot were usually littered with animal wastes and cut grass. In tropical developing countries, strict microbial effluent standards do not exist but there is a general guideline followed which gives a range of < 5000 / 100 ml as acceptable for release to the environment (Mara, 1976). The levels of faecal coliforms in the final effluent of the trickling filter line was much higher than the < 5000 / 100 ml acceptable for the release to the environment.
(Mara, 1976). This general guideline (< 5000 / 100 ml) may have been safe in 1970s, however with the current trend of emergence of new pathogens and resistant strains of known enteric pathogens, more strict standards are required. A standard of 1000 faecal coliforms / 100 ml based on WHO minimum requirement for unrestricted irrigation (WHO, 1989) was proposed by the designers of Nakuru town sewage treatment plant but none of the two treatment lines achieved 1000 faecal coliforms / 100 ml in the final effluent (Table 5).

5.2.2 Salmonella spp

The range of Salmonella spp in the raw sewage (11000 - 30000 counts ml\(^{-1}\)) were far higher than a range of 5 to 80 counts ml\(^{-1}\) typical for raw domestic sewage (Yate \& Gerba, 1998). High values observed at Nakuru town wastewater treatment plant can be attributed to high prevalence of Salmonella infections in the communities served by the wastewater treatment plant. A study on the prevalence of waterborne diseases within health facilities in Nakuru district indicated that typhoid fever caused by Salmonella typhi is the leading waterborne disease in the area (Hlupheka \& Hailemariam, 2000). Hence it is possible that high counts of Salmonella spp in the raw sewage can largely be attributed to the high number of typhoid fever cases that account for 49 % of all waterborne diseases (Hlupheka \& Hailemariam, 2000) within Nakuru district. A high Salmonella spp removal observed in the maturation ponds of both anaerobic pond and trickling filter lines can be largely attributed to high mortality rate due to high pH (8.4) observed in these ponds (Table 1) and increased exposure to UV light (Dubusk \textit{et al.}, 1998). Increase in Salmonella spp counts noted in the effluent from the rock filter and the grass plot along both treatment lines (Table 5) could have resulted from poor rock filter and grass plot management as explained in earlier (5.2.1). A 99.98 % reduction achieved by the anaerobic pond line was slightly higher than 99.60 % reported by Pescod \& Arar (1985); 99.5 % reported by Coetzee (1965); the 99.6 % reduction achieved by
sewage oxidation ponds in Central and Southern Africa (McGarry & Bouthiller, 1965). However, it is slightly lower than 99.99% reported by Yate & Gerba (1998). The difference in the percent reduction of *Salmonella* spp can be as a result of many factors, e.g. plant’s designs and effluent organic load. A treatment plant with longer retention time and sewage effluent of low organic load would achieve a higher percent reduction of *Salmonella* spp (and other bacteria) due to their prolonged exposure to UV light, predators such as protozoa and limited nutrients that characterize effluent of low organic load influent. Presence of *Salmonella* spp in the final maturation pond effluent after a retention period of over four weeks confirms an observation by McGarry & Bouthiller (1966) in Central and South Africa sewage oxidation ponds which indicates that the survival of *Salmonella* spp in the environment depended on presence of nutrients. This observation supports the findings of Jimenez *et al.* (1989) and Reilly & Twiddy (1992) which demonstrated that *Salmonella* can survive in the environment for weeks. A study by Windle (1965) regularly isolated both *Salmonella* spp and enteroviruses in sewage effluent that had passed through wastewater stabilization ponds. Therefore, presence of *Salmonella* spp in the final maturation ponds effluents indicates a high likelihood of enteric viruses being present in the final effluent. This poses a danger to wildlife in the park as enteric viruses are characterized by low infectivity dose, longer survival period in the environment and ability to adapt to new hosts (Asano, 2001). Their ability to adapt to new hosts means there is a likelihood of them causing new diseases in the new hosts.

Although the trickling filter line is a combination of both mechanical system and wastewater stabilization ponds, it only achieved a 98.5% reduction of *Salmonella* spp up to the third maturation pond that reduced to 92.5% at the outlet. This is an indication of poor performance of the treatment units along this line because wastewater stabilization ponds in series with a conventional treatment system are usually referred to as polishing ponds and their main function is to reduce the pathogenic microorganisms in the effluent.
by exposing them to UV light from the sun. This was not the case with Nakuru town sewage treatment plant trickling filter line as the ponds produced a poorer quality effluent compared to anaerobic pond line.

5.2.3 *Shigella* spp

The range of *Shigella* spp in the influent (1100-9000 counts ml⁻¹) was higher than the documented range of 0.01 to 10 counts ml⁻¹ (Yate & Gerba, 1998). Although a study on the prevalence of waterborne diseases within the health facilities in Nakuru district indicated the presence of few cases of bacillary dysentery (4 %) caused by *Shigella* spp (Hlupheka & Hailemariam, 2000), the area of study by Yate & Gerba (1998) may have been having a lower prevalence of bacillary dysentery than Nakuru area. The low counts of *Shigella* in the influent (Table 6) as compared to other pathogens (*Salmonella* and *Vibrio cholerae*) isolated in this study can be attributed to its low infection prevalence in Nakuru area (Hlupheka & Hailemariam, 2000). Increase in the density of *Shigella* in the effluent from the rock filter and the grassplot (Table 6) is an indication that the rock filter and the grass plot contribute negatively to microbial reduction. Sewage effluents or faecal contamination is the main source of *Shigella* in the environment. Presence of *Shigella* in the environment is a health hazard since the pathogen is highly virulent, hence its discharge to the environment should be avoided. In tropical countries Shigellosis is endemic due to pollution of water sources with sewage effluents and it has been estimated that some 5 million people die every year (David, 1997).

5.2.4 *Vibrio cholerae*

Although no cholera outbreak was reported in Nakuru area during this study, high counts of *Vibrio cholerae* was noted in the influent. It is likely that a number of people in Nakuru town are carriers of *Vibrio cholerae*. *Vibrio cholerae* grows best aerobically (Mackie & McCartney, 1997). The high reduction of *Vibrio cholerae* achieved by the
anaerobic pond (90.90 %) as compared to the trickling filter treatment unit (45.52 %) could be attributed to the anaerobic condition in this pond. The ability of *Vibrio cholerae* to survive for long periods in the environment was evident in this study. *Vibrio cholerae* was present in the final effluents after a retention period of more than four weeks (Table 6). Earlier studies suggested that *Vibrio cholerae* can only survive in the environment for a short period (Bryan, 1980; Lima dos santos *et al.*, 1993). However, recent investigations have reported that *Vibrio cholerae* is commonly found as a natural resident of aquatic environments in areas free of cholera epidemics and that its presence is not necessarily associated with recent faecal contamination (Mackie & McCartney, 1997). The ability of *Vibrio cholerae* to survive in the environment for long periods poses a risk of cholera outbreaks in case inadequately treated sewage effluent is discharged into water bodies. This is because sewage effluent would boost the number of *Vibrio cholerae* in the water to levels that can easily cause an outbreak of cholera, hence the need to ensure that there is a high pathogen reduction in the final sewage effluent. Most cholera outbreaks are linked with faecal contamination of drinking water (Mara, 1976).

### 5.2.5 Likely effects of human pathogens presence in the final effluent

Inadequately treated effluents contain pathogens that are currently wrecking havoc to both man and wildlife (Peter, 2000). The presence of high levels of faecal coliforms (above the set limit of 1000 faecal coliforms 100 ml⁻¹) and the enteric pathogens (*Salmonella* spp, *Vibrio cholerae*, and *Shigella* spp) isolated in the final effluent of Nakuru town wastewater treatment plant (Table 5 & 6) indicates that other potentially pathogenic microorganisms (e.g. enteric viruses and nematodes) commonly isolated from sewage (Windle, 1965; Klutse & Baleux, 1995; Mara, 1976) could be present. It has been shown that the risk associated with enteric viruses in biologically treated effluents and the probability of getting an infection by coming into contact with such waters are
high (Asano et al., 1992; Tanaka et al., 1998). Recent studies have revealed that the exposure of wildlife to human pathogens can lead to an increase in the emergence of new wildlife diseases, especially in areas subjected to wastewater effluent pollution (Peter, 2000). For example, marine turtles are experiencing increased rates of microbial infections and tumors (Alonso, 2001). Field observations indicate a high prevalence of tumors in turtles associated with heavily polluted areas and regions of high human population (Alonso, 2001). Coral reefs are facing large-scale destruction as a result of coral bleaching and infectious diseases (Raymond, 2001). Nearly 30% of the world's reefs have been lost since 1980 and some of the microorganisms affecting the coral reefs are human pathogens from sewage effluents (Raymond, 2001).

In the past, habitat destruction and chemical pollution were thought to be the main threat to biodiversity. However, presently pathogen pollution seems to be taking over as the main threat to biodiversity (Peter, 2000). In Nakuru town sewage treatment plant, wildlife graze on the grass plots meant for tertiary treatment of sewage. This exposes them to pathogen infection. According to Peter (2000), exposure of human pathogens to wildlife increases the risks of pathogens mutating leading to emergence of new diseases. Hence, animals should not be allowed to graze on an area irrigated with biologically treated effluent for at least two days after application of the treated effluent (WHO, 1989).

### 5.2.6 Phytoplankton

Chlorophyll $a$ is commonly used as an indirect measure of phytoplankton biomass because its concentration is closely correlated to phytoplankton biomass (Voros & Padisak, 1991). The mean algal biomass recorded in the effluent from the third maturation pond of the anaerobic pond line ($0.54 \text{ g m}^{-3}$) and trickling filter line ($3.6 \text{ g m}^{-3}$) was higher than $0.1 \text{ g m}^{-3}$ recorded in the final effluent at Akuse wastewater treatment...
plant (Isaac, 2000). The difference in algal biomass can mainly be attributed to difference in nutrient concentration in the ponds of the two wastewater treatment plants. For example, the mean levels of total phosphorus in the final effluent of Akuse wastewater treatment plant (has no rock filter and grass plot treatment units) was 0.2 g m$^{-3}$ (Isaac, 2000) while the levels of total phosphorus recorded at Nakuru town sewage treatment plant's final maturation pond were 6.34 and 10.23 g m$^{-3}$ along anaerobic and trickling filter line respectively.

Levels of algal biomass in the facultative and maturation ponds along anaerobic pond line (Fig. 18) for most of the time fell within the acceptable range of 0.5 to 2 g m$^{-3}$ of optimally performing ponds (Miguel, 2003). In the trickling filter line, algal biomass in the second and third maturation ponds was well above the upper limit (2 g m$^{-3}$) of an optimally performing pond. This explains why the wastewater in facultative and maturation ponds along anaerobic pond line had a dark green appearance, which is characteristic of a healthy algae population (Mara & Pearson, 1986). Occasionally, the colour of a health algal population can turn pink or red due (especially when slightly overloaded) to the presence of anaerobic purple-sulphur oxidizing photosynthetic bacteria (Mara & Pearson, 1986). High algal biomass recorded in the facultative and maturation ponds of the trickling filter line can be attributed to dominance by vacuolated cyanophytes such as Arthrosira and Microcystis that are relatively large in size. Ponds along the anaerobic pond line were dominated by other species e.g. Euglena, Phacus and Scenedesmus species. It is possible that, the high nutrient concentration in the trickling filter line (Table 3 & 4) was responsible for the bloom development of cyanophyte species. The cyanophyte species were also responsible for the surface algal scums owing to their tendency to float on water surface. During a period of calm weather, the vacuolated cyanophytes are known to rise to the surface of the water, forming a surface scum (Hilda & John, 1995).
Presence of dense algal blooms in wastewater stabilization ponds of the trickling filter line is a sign of inefficiency of the sewage treatment which must have been caused by the poor performance of the trickling filter line. This allows a high organic load effluent to reach the facultative ponds, which are normally designed to handle sewage of low organic load (Miguel, 2003). For example, the facultative ponds at Nakuru wastewater treatment plant are made to handle sewage effluent with a BOD of 206 mg O$_2$ L$^{-1}$. The high BOD observed along the trickling filter line confirms organic overloading of the same as compared to the anaerobic pond line (Fig. 7). Other reasons which could be responsible for the undesirable algal blooms is the presence of hippopotamus in the facultative ponds whose wastes and stirring up of the bottom sludge could be greatly contributing to the high nutrient levels in these ponds, creating highly eutrophic conditions.

Most algal blooms are known to contain algal species that have the potential of producing toxins capable of causing illnesses or death in man, domestic animals and wildlife. A bloom of *Anabaena flo-aquae*, *Anabaena discoidea* and *Microcystis aeruginosa* at Nyanza Gulf of lake Victoria was reported to contain cyanotoxins, Microcystins and anatoxin-a (Krienitz *et al.*, 2002). Toxin production by *Microcystis* and other strains of toxic alga have been linked to animal deaths and human health problems worldwide (Falconer, 1989). Toxins produced by blooms of *Microcystis aeruginosa* were responsible for the death of a number of dogs and sheep after drinking water from a reservoir contaminated with the algae at Rutland water supply in Western England (Mason, 1991; Gray, 1994). This therefore means that the presence of *Microcystis aeruginosa* in the wastewater stabilization ponds in Nakuru town sewage treatment plant increases the likelihood of wildlife within the park being exposed to cyanotoxin poisoning. The deaths of the birds that frequent the wastewater stabilization ponds (Personal observation, 2003-2004) may possibly be linked to the toxic algae present in the ponds.
CHAPTER 6

6 CONCLUSION AND RECOMMENDATIONS

6.1 Conclusion

The results obtained in this study indicate that:

(i) The anaerobic pond line is more efficient than the trickling filter line.

(ii) The BOD of effluents from various treatment units were generally much higher than the values recommended by the designers of the Nakuru town sewage treatment plant.

(iii) The mean concentrations of ammonia nitrogen and total phosphorus in the final effluent of the trickling filter line were higher than the set limits by the designer for discharge to the environment.

(iv) The levels of nitrates in the final effluent from the two treatment lines were lower than the set limit for discharge to the environment.

(v) The grass plot contributed negatively to the process of sewage purification by increasing the levels of effluent BOD, nutrients and microbial removal.

(vi) The anaerobic pond was well buffered as evidenced by high total alkalinity observed in this pond, an indication of no danger of treatment failure as a result of sudden pH change.

(vii) The maturation ponds as compared to other treatment units were relatively effective in microbial and nutrient removal.

(viii) The levels of faecal coliforms in the final effluent were much higher than the set limit of 1000 coliforms 100 ml⁻¹. Similarly, the levels of other enteric pathogens isolated were well above 1000 counts 100 ml⁻¹.

(ix) The presence of levels above 1000 faecal coliforms 100 ml⁻¹ (indicator organisms) and enteric pathogens (Salmonella spp, Shigella spp and
*Vibrio cholerae* isolated in the final effluent from both treatment lines is an indication that there is a likelihood of other potentially pathogenic microorganisms commonly isolated from sewage effluents being present.

(x) The trickling filter line had high nutrient levels and its algal population was dominated by cyanophytes that led to formation of algal scums on the surface of the sewage ponds.

(xi) Potentially toxic algae were present in the facultative and maturation ponds of the two treatment lines that are a threat to the wildlife.

### 6.2 Recommendations

The following recommendations could help in improving the situation:

(i) The treatment plant should be fenced off with an electric fence to prevent wildlife from coming into contact with partially treated wastewater in the waste stabilization ponds and grazing on grassplots made for tertiary treatment of the biologically treated sewage effluents. This would minimize direct exposure of the wildlife to microbial human pathogens and the potentially toxic algae in the facultative and maturation ponds. It would also prevent destruction of structural components of the wastewater treatment units.

(ii) Regular maintenance of the wastewater treatment units to ensure a high performance is necessary. This should include disludging of all the ponds at the scheduled time, an action that is long overdue according to the design specifications. Installation of an electrically operated machine at the inlet for removing objects trapped at the grid could be of great value as it would prevent over-flowing of influent at the inlet at night that would allow large objects to enter the treatment system. This is
important especially in the trickling filter whose rotation is determined by the volume of wastewater flowing through it. Presence of large objects within the rotating arm would automatically affect the uniform distribution of the sewage within the rotating arm and over the filter bed.

(iii) Replacement of the whole mechanical system with anaerobic ponds which are cheaper in terms of maintenance.

(iv) It is necessary to protect the wildlife within Nakuru National Park from exposure to human pathogens. This can be achieved through restriction of access of wildlife to the sewage ponds.

(v) With the current trend of emergence of new diseases in wildlife in areas subject to human pollution, it is important that a domestic wastewater treatment plants located within wildlife habitat be made very efficient as a way of protecting the animals from exposure to human pathogens and other pollutants.

(vi) Since exposure of microbial human pathogens to wildlife within Nakuru National Park could be one of the causes of wildlife deaths within the park, wildlife deaths within the park should be closely monitored and the actual causes established.


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