ASSESSING THE PERFORMANCE OF DOMPOASE WASTEWATER TREATMENT PLANT AND ITS EFFECT ON WATER QUALITY OF THE ODA RIVER IN KUMASI

by

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DECLARATION

I hereby declare that this submission is my own work towards the MSc and that, to the best of my knowledge, it contains no material previously published by another person nor material which has been accepted for the award of any other degree of the University, except where due acknowledgement has been made in the text.

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ABSTRACT

Treatment performance of the Dompoase wastewater treatment plant and the effect of final effluent on the Oda River were evaluated. Influent and effluent wastewaters as well as receiving water qualities were monitored for a period of three months within 2 weeks intervals. The study showed mean percentage removal of 51.23 (± 15.34), 89.18 (± 8.43), 36.11 (± 34.65), 80.80 (± 17.85), 58.02 (± 41.05), 22.51 (± 15.53), 22.23 (± 18.93), 60.94 (± 42.79), 68.52 (± 26) and 92.20 (± 3.82) % for TDS, TSS, Fe, COD, BOD, N, P, Pb, total coliforms and faecal coliforms respectively. In contrast, K and pH revealed higher mean effluent than influent values, hence percentage removals of -27.59 (± 34.40) and 10.24 (± 1.03) were obtained for K and pH. Reduction from influent to effluent values showed statistical significant differences among mean values for TDS, TSS, COD, BOD, N, K, pH, and Pb ($P<0.05$). However, there was no significant difference observed for mean values of P, Fe, total and faecal coliforms ($P>0.05$). Total dissolved solids (TDS), TSS, COD, BOD, N, P, Pb, total coliforms and faecal coliforms in effluent wastewater were above the recommended EPA guidelines. But pH was in the acceptable range of 69. It was concluded that effluents fell short of standard requirement before discharge into surface waters. Even though, concentrations of all parameters decreased with distance from the discharge point in River Oda, downstream values of most parameters were higher than upstream values. Water quality parameters of the Oda River were affected as rainfall increases from May through to July.

DEDICATION

This thesis is dedicated to all members of the Abuenyi family and Cynthia Ama Obo for their love, generous support and prayers.

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LIST OF ABBREVIATIONS

BOD	Biological Oxygen Demand
COD	Chemical Oxygen Demand
DDT	Dichoro-diphenyl-trichloroethane
EPA	Environmental Protection Agency
FC	Faecal Coliforms
IAEA	International Atomic Energy Agency
K	Potassium
KMA	Kumasi Metropolitan Assembly
N	Nitrogen
P	Phosphorus
Pb	Lead
PCBs	Polychlorinated biphenyls
TC	Total Coliforms
TDS	Total Dissolved Solids
TSS	Total Suspended Solids

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CHAPTER ONE

1.0 INTRODUCTION

1.1 Background

Adequate potable water supply remains a major challenge for most developing countries, despite its importance in primary health care (Osode, 2007). With increasing population, the demand for quality water has become even more critical (UNESCO/WHO/UNEP, 1996). The United Nations Centre for Human Settlements noted that populations in peri-urban areas in developing countries are growing twice as fast as in the formal cities (Rasula and Rasula, 2001). Such increases have threatened water quality due to domestic and industrial wastewater discharges and by certain agricultural activities. The problem is particularly acute in the densely populated periurban areas and rural areas where the large majority of the dwellers are typically lowincome people. It is estimated worldwide that over half a billion urban people and over 2 billion rural people lack sanitation services (Osode, 2007). Despite efforts by most developing countries in the last two decades, investment in the sanitation sector has remained inadequate while the needs have continued to grow especially with regard to wastewater treatment (Osode, 2007).

Wastewaters show different degrees of environmental nuisance and contamination hazard due to their chemical and microbiological characteristics (Bohdziewicz and Sroka, 2005). Wastewater effluents are responsible for the degradation of several ecosystems (Steven *et al*., 2008). Impacts may arise from an increase in nutrient loads leading to eutrophication, decreased levels of dissolved oxygen and releases of toxic substances, many of which can bioaccumulate and biomagnify in aquatic wildlife (Morrison *et al*., 2001). Physical changes to the environment can also occur, including thermal enhancement, increased

water flow, leading to potential flooding and erosion, increase in suspended solids, and the release of floating debris to the country's waters (Steven *et al*., 2008). The problem is pronounced in areas where wastewater treatment systems are simple and not efficient (Igbinosa and Okoh, 2009).

While the impact of untreated wastewater on local rivers and streams is clear, proper wastewater treatment is also fundamental to maintaining people's health, protecting the quality of drinking water and ultimately promoting economic development (WVRC, 2005). Wastewater streams running directly into the aquatic environment have both an acute and chronic impact on the environment which may be very severe and can diminish biodiversity and greatly reduce populations of sensitive species. Toxic metals and organics, where present, can lead to chronic toxin accumulation in both local and downstream populations (Meena *et al*., 2010). Quality assessment of water and wastewater is, therefore, crucial to safeguarding public health and the environment (Okoh *et al*., 2005; 2007). According to the World Bank, the greatest challenge in the water and sanitation sector over the next two decades will be the implementation of low cost sewage treatment that will at the same time permit selective reuse of treated effluents for agricultural and industrial purposes (Looker, 1998). It is crucial that sanitation systems have high levels of hygienic standards to prevent the spread of disease. Other treatment goals include the recovery of nutrient and water resources for reuse in agricultural production and to reduce the overall user-demand for water resources (Rose, 1999).

Problems concerning water and sanitation in Ghana stem from the rise in urban migration and the practice of discharging untreated wastewater. The uncontrolled growth in urban areas has made planning and expansion of water and sewage systems very difficult and

expensive to carry out. In addition, many people moving to the city have low incomes, making it difficult to pay for any water system upgrades as reported by Looker, (1998).

Agodzo *et al.*, (2003) reported that the total amount of grey and black wastewater currently produced annually in urban Ghana is estimated at 280 million m^3. This wastewater is derived mainly from domestic sources as Ghana's industrial development is concentrated along the coastline where wastewater, treated or untreated, is disposed off into the ocean. But collection and disposal of domestic wastewater is done using underground tanks such as septic tanks and aqua-privies, either at industrial facilities or at the community level and then transported by desludging tankers to treatment works or dumping sites. However, the cost of putting in place the required infrastructure to effectively collect and dispose of all urban wastewater is excessive and this denies majority of urban population in Ghana the appropriate means to manage wastewater (Agodzo *et al.*, 2003). For the country to continue to develop economically, while meeting the wide-ranging needs for water, urgent steps must be taken to protect the quality of the resource. In this regard, wastewater treatment becomes critical.

To help prevent the harmful effect of wastewater on the environment and human health, the local authority in Kumasi started operating a wastewater treatment plant at Dompoase in 2004. The treatment plant was designed to treat 300 m^3 per day of faecal sludge and 300 m^3 per day of leachate from the nearby landfill. About 6,275 m^3 of faecal sludge discharged monthly at Dompoase is treated in the pond system in combination with the leachate from landfilled solid waste. Unfortunately, the quality of the effluent ejected into the Oda River, is not desirable. This effluent is black in colour and foamy, showing that

environmental protection is still questionable (IRC- International Water and Sanitation Centre, 2006).

Effluents may contain organic and inorganic toxic pollutants which might flow laterally or percolate through permeable soil strata and pollute surface or ground water. The effect of such uncontrolled effluent disposal system renders surface waters and the underground water systems unsafe for human, agricultural and recreational use; destroys biotic life, poisons the natural ecosystems, poses a threat to human life and is therefore against the principles of sustainable development (Benka-Coker and Bafor, 1999).

Lack of technical knowledge and failure to consider all relevant local factors at the predesign stage, are likely to contribute to wastewater treatment plant failure in Kumasi. As a result, wrong decisions are often made and inappropriate unsustainable treatment processes are selected and implemented. This is then exacerbated by the absence of any real incentive to operate the wastewater treatment plant correctly once it has been commissioned (Parr and Horan, 1994).

In advanced countries, environmental monitoring agencies are more effective and environmental laws are strictly followed. General environmental quality monitoring is compulsory and the monitoring of the quality of water resources is done on a regular basis (Robson and Neal, 1997; Neal and Robson 2000). As a result, any abnormal changes in the water quality can easily be detected and appropriate action taken before the outbreak of epidemics. The direct opposite is observed in Kumasi where monitoring by local operators is questionable. Again, the Environmental Protection Agency in Ghana lacks the needed logistics to continually monitor and assess the impact of wastewater effluent on

receiving waters. The outcome is weak enforcement of environmental regulations which allow local authorities to flout environmental regulations without any sanctions.

Ahn *et al.*, (2004) reported that it was a common practice to treat leachate together with municipal sewage in the municipal sewage treatment plant. This was because of its easy maintenance and low operating costs. However, this option has been increasingly questioned due to the presence in the leachate of organic inhibitory compounds with low biodegradability and heavy metals that may reduce treatment efficiency and increase the effluent concentrations (Cecen and Aktas, 2004). Again, wastewater treatment plants are usually sited near rivers and streams. Therefore, effluent quality that meets standard requirements is of great importance.

Furthermore, the application of a technology is dependent on local physical factors of land availability, its topography, climate, soil, availability of energy and existing land uses. Sound practices are therefore practices which fit into the environmental, economic, social, cultural and institutional setting of the community. Long term sustainability however, is a function of community resources (funds, skills) to afford the technology and willingness to pay for the technology and its operation. The study therefore seeks to answer questions concerning the efficiency of the existing design, management and the availability of funds for the operation of the treatment process.

1.2 General objective

To investigate the efficiency of the wastewater treatment plant and the level of pollution of the Oda River due to the wastewater effluent discharge.

1.3 Specific Objectives

- To assess the efficiency of the Dompoase wastewater treatment plant by comparing the composition of the influent and effluent wastewater.
- To determine the effect of the wastewater effluent on the quality of the Oda River through the comparison of some physical, chemical and biological indicators obtained from downstream and upstream.
- To check the appropriateness of the Dompoase treatment plant in treating both faecal sludge and landfill leachate.

1.4 Significance of the study

The supply of freshwater is limited and threatened by indiscriminate discharge of untreated wastewater effluents. In developed countries, municipal wastewater systems are well organized and cover most parts of the regions but this is not the case in developing countries like Ghana. Water is a scarce commodity and there is the need to protect the available water resources from discharges of untreated wastewater. Various forms of wastewater treatment exist in Ghana; however this study provides valuable information on waste stabilization ponds as a means of ensuring a cost effective treatment system that meets standard requirements before discharge into surface waters.

Furthermore, the study was planned to generate information that could be used by wastewater treatment plant managers and the Environmental Protection Agency of Ghana in order to develop or review an effective policy for wastewater treatment plants in meeting standard requirements for discharge of effluents into water sources.

CHAPTER TWO

2.0 LITERATURE REVIEW

2.1 Waste

Waste can be loosely defined as any material that is considered to be of no further use to the owner and is, hence, discarded (Allen, 2001). However, most discarded waste can be reused or recycled, one of the principles of most waste management philosophies. Waste is generated universally and is a direct consequence of all human activities. It is generally classified into solid, liquid and gaseous forms. Gaseous waste is normally vented to the atmosphere, either with or without treatment depending on composition and the specific regulations of the country involved. Liquid wastes are commonly discharged into sewers or rivers, which in many countries is subject to legislation governing treatment before discharge. In many parts of the world such legislation either does not exist or is not sufficiently implemented, and liquid wastes are discharged into water bodies or allowed to infiltrate into the ground. Indiscriminate disposal of liquid wastes pose a major pollution threat to both surface and groundwater (Taylor and Allen, 2000).

2.2 Waste Management

The need for appropriate waste management has been regularly voiced out in most countries (Mwesigye *et al*., 2009). With growing concerns over the large quantities of both solid and liquid waste being produced waste management has become an important focal area for sustainable development (Mwesigye *et al*., 2009). Waste management is the

collection, transport, processing, recycling or disposal of waste materials, usually produced by human activity, in an effort to reduce their effect on human health or local aesthetics or amenity (Mwesigye *et al*., 2009).

The safety and acceptability of many widely used solid waste management practices are of serious concern from the public health point of view. Such concern stems from both distrust of policies and solutions proposed by all tiers of government for the management of solid waste and a perception that many solid waste management facilities use poor operating procedures (Hamer, 2003). Landfills are accepted worldwide for the disposal of solid waste. But this technology is subject to criticism either by environmentalists on the grounds of possible hazardous emissions, failure to eliminate pathogenic agents or failure to immobilize heavy metals. Again, key questions concerning the effects of the various practices on public health and environmental safety remain unanswered (Hamer, 2003).

Securing safe water and reducing the unregulated discharge of wastewater are among the underlying concept of wastewater management (WHO, 2008). Unmanaged wastewater has far reaching implications for the health of all aquatic ecosystems, which threatens to demine the resilience of biodiversity and ecosystem services on which human wellbeing depends (Corcoran *et al*., 2010). However, wastewater treatment receives a low or poor target share of development aid and investment developing countries (WHO, 2008).

2.2.1 Major trends and emerging issues on waste management in developing countries

Waste management problems in developing countries are varied and complex with infrastructure, political, technical, social, organizational, regulatory as well as legal issues

and challenges to be addressed (Mwesigye *et al*., 2009). Waste is typically disposed off without consideration for environmental and human health impacts, leading to its accumulation in cities, towns and uncontrolled dumpsites. Co-disposal of nonhazardous and hazardous waste without segregation is a common practice (Mwesigye *et al*., 2009).

Waste management in these countries suffer from limited technological and economic resources as well as poor funding which collectively result in the prevalent low standards of waste management. This is exacerbated by public perception of waste disposal as a welfare service issue and hence the reluctance to pay for waste disposal especially among the poor (Mwesigye *et al*., 2009).

Across Africa, improper waste disposal has resulted in poor hygiene, lack of access to clean water and sanitation by the urban poor. Consequently most of the countries in the region may not be able to meet the Millennium Development Goal target of reducing by half the proportion of people without sustainable access to safe drinking water and basic sanitation by 2015 (Mwesigye *et al*., 2009).

2.2.2 Waste management in Ghana

Urban centers in Ghana are experiencing a complex waste management crisis. An assessment of past and present waste management policy plans has revealed several structural weaknesses accounting for this crisis, with indirect causes being lack of wellthought-out management and financial sustainability plans that ensure enduring financing for waste management activities. Therefore, the waste management systems have never run efficiently leading to frequent breakdowns, the consequence of which is the worsening environmental quality in the country (Julius *et al*., 2010).

The current state of waste management leaves much to be desired. Less than 40% of urban residents are served with solid waste collection services and less than 30% by an acceptable household toilet facility. The traditionally applied methods of dealing with wastes have been unsuccessful, and the resulting contamination of water and land has led to growing concern over the absence of an integrated approach to waste management in the country (UN, 2004).

Waste management practices in Ghana Solid waste

Solid waste is collected and disposed of at designated landfill and waste dump sites by public and private waste management firms. The issue of landfill site location has been a matter of strenuous negotiations with rising population pressure continuing to impact on waste generation and management. Coastal and marine-based industries tend to pollute coastal areas through the discharge of untreated wastes into the marine environment (Mwesigye *et al*., 2009).

Hazardous Wastes

Biomedical and other hazardous waste are currently being managed through land filling. In response to the global mandate for environmentally sound management of hazardous, solid and radioactive waste, Ghana has, among other things, embarked on a life cycle approach to address chemicals and other hazardous wastes management in an integrated manner. This involves a broad range of stakeholder institutions and organizations including non-governmental organizations. With respect to Hazardous Waste Management, there are currently no clearly distinguishable methods for the disposal of

hazardous waste. However, the Environmental Protection Agency (EPA) is responsible for the provision of guidelines for such wastes (Mwesigye *et al*., 2009).

Radioactive Wastes

The waste management system consists of a decontaminated unit intended for low and intermediate level waste storage and concrete wells for interim storage of spent fuel. The suitability of these facilities has been assessed for waste storage and processing and their contamination units and wells found to be in good condition for refurbishment for use as waste processing and storage facilities. A new storage facility with a capacity of 100 litre drums has been constructed to complement the existing structure. The new facility is consistent with current trends in waste management technological development and IAEA standards (Mwesigye *et al*., 2009).

Liquid waste

In Ghana the excreta disposal problems have become serious: thousands of tons of sludge from on-site sanitation installations are disposed untreated and indiscriminately into lanes, drainage ditches, onto open urban spaces, into inland waters, estuaries, and the sea. Wastewater treatment and disposal, therefore becomes a matter of great concern that needs to be addressed.

In order to design an adequate sewer system for wastewater treatment, cities need to be planned according to a development strategy which formulates a holistic vision for the city (LaGro, 1996). Unfortunately, this difficult task cannot be accomplished in a

developing country like Ghana. The sewer systems built are generally not technically suitable and economically much more expensive (Looker, 1998; Agodzo *et al.*, 2003).

Lack of proper sewer system makes it very difficult to treat wastewater with modern wastewater treatment technologies. Stored wastewater needs to be carried to a suitable receiving medium at regular intervals. These collected wastewaters are generally denser than ordinary wastewater and therefore when the wastewater is disposed; it causes serious environmental and ecological problems in the receiving medium, especially when sewage is discharged uncontrolled. Wastewater disposal must be managed effectively to safeguard public health, and protect freshwaters from pollution. They must be reintegrated safely in the water cycle and accounted for in the water budget of the household, community, industry, and the agriculture (Looker, 1998).

2.3 Wastewater treatment facilities in Ghana

A monitoring survey conducted by Ghana Environmental Protection Agency (EPA, 2001) on the number, status, treatment methods and distribution of both faecal sludge and sewage treatment plants in Ghana, found out that more than half of all treatment plants in Ghana are in the Greater Accra region. Two regions (Brong Ahafo and Upper West) have no treatment plants at all (Adu-Ahyiah and Anku, 2003). The stabilization pond method is the most extensively used with almost all faecal sludge and largecapacity sewage treatment plants using the method. Most trickling filters and activated sludge plants recorded have a low capacity and belong to private enterprises like larger hotels. Less than a quarter of the treatment plants are operational. No precise figure can be given on the

percentage that meets the EPA effluent guidelines and the capacity of these, but indications show that hardly any of the plants is meeting them (Akuffo, 1998).

2.3.1 Volumes and sources of wastewater in Kumasi

Based on an estimated faecal sludge production of 1l/ca/day for septic tank and 0.2l/ca/day for heavy sludge (Heinss *et al.,* 1998), the total faecal sludge production of Kumasi has been estimated at 23,127 m^3 per month of which 18,323 m^3 is in toilets that can be emptied. The remaining 4,447 and 356 m^3, go respectively into the sewerage system and into the bush (IRC-International Water and Sanitation Centre, 2006).

The principal generators of industrial wastewater in Kumasi are the two breweries, a soft drink bottling plant and an Abattoir. Together, they generate about 1,000 m^3 of effluent daily, all of which end up in the city's drains without treatment. Light industrial activities from "Suame Magazine" and sawdust from the saw mills also generate significant amounts of waste oil and leachate respectively, which add to environmental pollution (Adu-Ahyiah and Anku, 2003).

2.3.2 Disposal and treatment of domestic wastewater in Kumasi

Five separate small-scale sewerage systems are currently available in Kumasi. There are two conventional systems at Kwame Nkrumah University of Science and Technology (KNUST) and one connecting the Komfo Anokye Teaching Hospital (KATH), Golden Tulip Hotel and the central parts of the 4BN Army barracks (Dahlman, 2009). There are

two satellite systems at Ahinsan and Chirapatre suburbs and one simplified sewerage system at Asafo. However, both of the conventional systems are not in operation. The KNUST plant was designed as a trickling filter system and had an inflow of about 390 m^3 per day. Even though this facility has been rehabilitated, current student population and other operational difficulties inhibits its proper functioning. Raw sewage from KNUST sometimes, is discharged into a 'wetland' linked to River Wiwi, where urban farmers practice vegetable farming. Grey water mainly from students' hostels and staff quarters (250 m^3 per day) runs in open gutters to nearby streams (Wiwi and Sisa).

Asafo's simplified sewerage network was built in 1994 in a high population density suburb of Asafo. The plant has 4 stabilization ponds and can serve up to 20,000 people but only 60 % of the people are connected (1.2 % of the Kumasi population). Its effluent is discharged into the Subin stream.

The two satellite plants are at two low-cost housing estates of Chirapatre and Ahinsan. They were built in the late 1970s. They were equipped with a sewer network and communal septic tank systems for black water. Chirapatre had six communal septic tanks for a population of 1800 inhabitants and Ahinsan five for about 1500 inhabitants. Sewer lines were blocked and septic tanks were in a bad state of maintenance. Both schemes have been replaced with two sewerage networks with waste stabilization pond treatment methods. Greywater (effluent from bathrooms and kitchens) is discharged into the drainage system (Obuobie *et al.*, 2006).

Until a few years ago, Kumasi has been without any proper treatment plant for faecal sludge. A temporary treatment facility with design capacity 144 m^3/day was built south of Kaase in 1999 (Leitzinger and Adwedaa, 1999). It was soon overloaded with up to 500 m^3 per day and faecal sludge flowed into the Sisa River without any treatment. However, having no alternative, the Kaase plant was used until 2003, when another 200 m^3 per day capacity plant was constructed and used at Buobai. The use of the Buobai plant was stopped due to conflicts with the community. Since March 2004, the local authority has been operating a second faecal sludge treatment plant at Dompoase with a design capacity of 300 m^3 per day of faecal sludge and 300 m^3 per day of leachate from the nearby landfill (Obuobie *et al*., 2006).

On average 1255 trips of faecal sludge are discharged monthly at Dompoase faecal sludge treatment plant, which amount to 6,275 m^3. This represents just over one third (34 %) of the collectable faecal sludge of 18, 323 m^3 monthly in various emptyable toilets (IRC-International Water and Sanitation Centre, 2006)

The 6,275 m^3 of faecal sludge discharged monthly at Dompoase is treated in the pond system in combination with the leachate from landfilled solid waste. Treatment is through a series of 5 anaerobic ponds, 1 facultative pond and 2 maturation ponds.
Unfortunately, the quality of the treated effluent is not known (Buama-Ackon, 2006). The mixed effluent is black in colour and foamy, showing that environmental protection is still questionable (IRC-International Water and Sanitation Centre, 2006).

2.4 Importance of wastewater treatment

Proper wastewater treatment enables ecosystems within water sheds to thrive and deliver services to communities and economies that depend on them (Hernández-Sancho *et al*., 2010). Wastewater treatment and reuse in agriculture can provide benefits to farmers in conserving fresh water resources, improving soil integrity, preventing discharge to surface and ground waters and improving economic efficiency (Corcoran *et al*., 2010).

Treatment methods in a country or region vary with the population density and state of technological development. Sparsely settled rural communities can employ simple treatment processes to reduce the concentrations of BOD, TSS or pathogens in domestic sewage. However, in urban centers as municipal and industrial waste become more complex and the protection of receiving waters more necessary, wastewater treatment methods must become more sophisticated and more efficient (Henry and Heinke, 1989).

2.5 Wastewater treatment by Stabilization Ponds

The most appropriate wastewater treatment is that which will produce an effluent meeting the recommended microbiological and chemical quality guidelines both at low cost and with minimal operational and maintenance requirements (Pereira *et al.*, 2002). Low level treatment is especially desirable in developing countries, not only from the point of view of cost but also in acknowledgement of the difficulty of operating complex systems reliably.

Waste stabilization ponds are now the first choice treatment method for wastewater in many parts of the world (Lukman *et al.*, 2010). In Ghana and other developing African

countries, unlike the developed world, waste stabilization pond is considered the ideal way of using natural processes to improve sewage effluents.

The activity in the waste stabilization ponds is a complex symbiosis of bacteria and algae, which stabilizes the waste and reduces pathogens. The result of this biological process is to convert the organic content of the effluent to more stable and less offensive forms. Through this process, a variety of wastewater from domestic wastewaters to complex industrial waters can be treated (Ramadan and Ponce, 2004a). After treatment, the concentrations of many pollutants that were present in the raw sewage are reduced, but smaller amounts of most of these pollutants still remain in the effluent. In many cases, the concentrations of the remaining pollutants may still be high enough to cause serious environmental damage. Such contaminants include biodegradable oxygenconsuming organic matter, suspended solids, nutrients, microorganisms and sulphides.

2.6 Effect of effluent discharge on receiving water

2.6.1 Nutrient Enrichment

One of the most widely recognized and studied environmental effects of municipal wastewater effluents is nutrient enrichment (Welch, 1992). Some nutrients, particularly phosphorus and nitrogen, are essential for plant production in all aquatic ecosystems. However, increased nutrient loading can lead to eutrophication (Gücker *et al.*, 2006) and temporary oxygen deficits (Rueda *et al*., 2002).

The net effect of eutrophication on an ecosystem is usually an increase in the abundance of a few plant types (to the point where they become the dominant species in the

ecosystem) and a decline in the number and variety of other plant and animal species in the system.

2.6.2 Depletion of dissolved oxygen

Wastewater effluents contain large quantities of organic solids, and the bacterial breakdown of this material and the oxidation of chemicals in it can consume much of the dissolved oxygen in the receiving water. Since dissolved oxygen is essential to most aquatic life, oxygen depletion can have serious effects on aquatic life. These effects may be immediate and short-term or may extend over months or years as a result of the buildup of oxygen-consuming material in the bottom sediments (Hvitved-Jacobsen, 1986).

2.6.3 Direct toxicity to wildlife

The toxic impacts of municipal wastewater on wildlife may be acute and occur within a short period of time, or they may be cumulative and appear only after an extended period of time (Hvitved- Jacobsen, 1986; Harremoes, 1988). Acute impacts from treatment plant effluents are generally caused by high levels of ammonia and chlorine, high loads of oxygen-demanding materials, or toxic concentrations of heavy metals and organic contaminants. Cumulative impacts result from a gradual build-up of pollutants in the receiving water or in its sediments and biota and become apparent only after accumulation exceeds a certain threshold. Because of the complexity and variability of municipal effluents, however, and the variety of environmental factors that affect their biological activity individually and in combination, it is not easy to arrive at broad generalizations about the toxicity of municipal wastewater effluents (Welch 1992; Chambers *et al*., 1997).

Freshwater organisms are most at risk from exposure to ammonia (Environment Canada, 2000). The major impact of ammonia in aquatic ecosystems is likely to occur through chronic toxicity to fish and bottom-dwelling invertebrates, resulting in reduced reproductive capacity and reduced growth in the young. The zone of impact from the toxic components of municipal wastewater effluents varies considerably with discharge conditions, such as river flow rate, temperature, and pH. For example, waters most at risk from municipal wastewater-related ammonia are those that are routinely basic in pH with a relatively warm summer temperature combined with low flows. Under estimated average conditions, some municipal wastewater discharges could be toxic for 10–20 km from their point of release. Severe disruption of bottom flora and fauna has been noted below municipal wastewater discharges, and normal bottom conditions may not resume until as much as 20–100 km from the discharge site.

2.6.4 Bioaccumulation and Biomagnifications of contaminants

Bioaccumulation causes substances that are found only in low or even barely measurable concentrations in water to be found in very high concentrations in the tissues of plants and animals. Bioaccumulative substances tend to be very stable and long-lived chemically and are not easily broken down by digestive processes. Many of them are more soluble in fat than in water and therefore tend to accumulate in fatty tissues rather than being excreted from the body (Morrison *et al.*, 2001). A limited number of these contaminants can undergo further changes through biomagnifications.

Because of these processes, even very low concentrations of certain substances in wastewater are of concern. Persistent, toxic, bioaccumulative substances that have been detected in municipal wastewater include PCBs, dioxins and furans, organochlorine pesticides, and mercury and other heavy metals. Only a few metals and organic chemicals, such as mercury and DDT, are known to biomagnify throughout food webs, even though many substances can bioaccumulate. Although there are several other sources of persistent bioaccumulative toxic substances in the environment, including industrial discharges and deposition of atmospheric contaminants, municipal wastewater remains one of the most significant sources.

2.6.5 Physical changes to receiving waters

Municipal wastewater effluents are sources of thermal enhancement because they are warmer than the water. These changes in temperature affect the variety and abundance of species as well as enhance algal growth (Welch, 1992).

Further, Municipal wastewater effluents are responsible for a long-term continuous input of suspended solids to the environment. Suspended solids released into receiving waters, mainly from wastewater effluent discharges, can cause a number of direct and indirect environmental effects, including reduced sunlight penetration, smothering of spawning grounds, physical harm to fish, and toxic effects from contaminants attached to suspended particles (Horner *et al*., 1994). The growth and survival of some species may also be affected, either through direct effects or through indirect effects caused by changes in the food web or interference with dispersal or migration. Such effects can manifest themselves on various time scales. A single large rainfall or runoff event can cause significant immediate impacts, but generally the long-term effects are more important.

2.7 Water Quality Standards and Monitoring

A common challenge in developing countries is that water quality data are scarce and do not provide adequate information for making decisions or assessing complex situations (Ongley, 2001). The establishment of water quality regulations and monitoring capacity, however, is critical to the implementation of wastewater management programme.

Several types of water quality standards are relevant to wastewater management programmes and are often concerned with the direct disposal or reuse of excreta and grey water and the beneficial use of treated sludges.

For on-site sanitation, design standards should prevent groundwater contamination. In many cases, concentrated wastewater effluents from industries should be pretreated or treated separately from domestic wastewaters. Establishing appropriate standards requires information about the surface and ground waters that receive the wastes, and ongoing monitoring is needed to determine when degradation has occurred. Allowable discharge levels of pollutants should ideally be based on the assimilative capacity of the receiving water body. Approaches that can be used in the development of water quality standards include risk assessment (WHO, 2003), total maximum daily loads and biomonitoring (Resh, 2007).

In view of this the environmental protection agency in Ghana has provided effluent guidelines for both existing and new facilities in an effort to improve effluent quality and prevent pollution of surface waters as well as the natural environment (EPA, 2000). These standards include 1000 mg/L, 25 mg/L, 250 mg/L, 50 mg/L, 75 mg/L, 2 mg/L, 0.1 mg/L,

6-9, 400 MPN/ 100 ml and 400 MPN/ 100 ml for TDS, TSS, COD, BOD, N, P, Pb, pH, total coliforms and faecal coliforms respectively.

CHAPTER THREE

3.0 MATERIALS AND METHODS

3.1 STUDY SITE

The study was carried out in the Kumasi Metropolis, the most populous in the Ashanti Region. The Kumasi Metropolitan Area has a total surface area of 254 sq km with a population density of 5,419 persons per sq. km and a population of 1,170,270 (2000 population census). It has been projected to have a population of 1,625,180 in 2006 based on a growth rate of 5.4% per annum and this accounts for just under a third (32.4%) of the region's population (KMA, 2006).

The city is traversed by major rivers and streams, which include the Oda, Subin, Wiwi, Sisai, Owabi, Aboabo, Nsuben among others. However, encroachment as a result of estate development and indiscriminate waste disposal practices have impacted negatively on the drainage system and have consequently brought these water bodies to the brink of extinction.

The daily generation of solid waste in Kumasi is estimated at 1000 metric tons, about 70% of which is collected. The bulk of the solid waste generated in the Metropolis is collected by the private sector based on a mixture of contract and franchise arrangements. The main collection methods employed are house-to-house and communal container collection systems (Mensah, 2005) and final disposal at the landfill site.

To manage the liquid waste generated in the Metropolis, a faecal sludge treatment plant, consisting of five anaerobic, one facultative and two maturation ponds to treat faecal sludge and landfill leachate is available at Dompoase. It has a design capacity of 300 m^3/day of faecal sludge and 300 m^3/day of leachate. The facility became operational in January 2004. The treated liquid effluent is discharged into the Oda River without further treatment, despite questionable effluent quality (Vodounhessi and Münch, 2006).

3.2 Sampling design and data collection

Four sample sites were selected for the study. Two sites were selected to obtain raw influent and effluent wastewater through the treatment plant. The other two sample sites were selected to obtain water samples before and after the discharge of effluent wastewater into the Oda River. The first sample location was at a point where faecal sludge from trucks was added to the landfill leachate (sample S1– N06˚37'30.4" and W001˚35'28.7"); Second location selected was at the end of the treatment ponds where treated effluent is discharged (sample S2– N06˚37'20.9" and W001˚35'27.2"). The third and fourth sample locations were approximately 100 m upstream and downstream where treated effluent is

discharged into the Oda River. These were represented as sample points S3 (N06˚37’10.9” and W001˚35’17.3") and S4 (N06˚37'06.6" and W001˚35'20.7") respectively. A total of 36 samples were collected over a three-month period within 2 weeks interval. This sampling period was selected to allow the collection of samples throughout the major part of the rainy season. Duplicate samples were collected at each sampling point. The samples were collected in well-labeled clean bottles that were rinsed out thrice with distilled water prior to sample collection.

Parameters selected were specifically for the assessment of the environment. Rainfall data was obtained from the meteorological department in Kumasi, to ascertain the effect of rainfall on wastewater constituents and treatment plant efficiency.

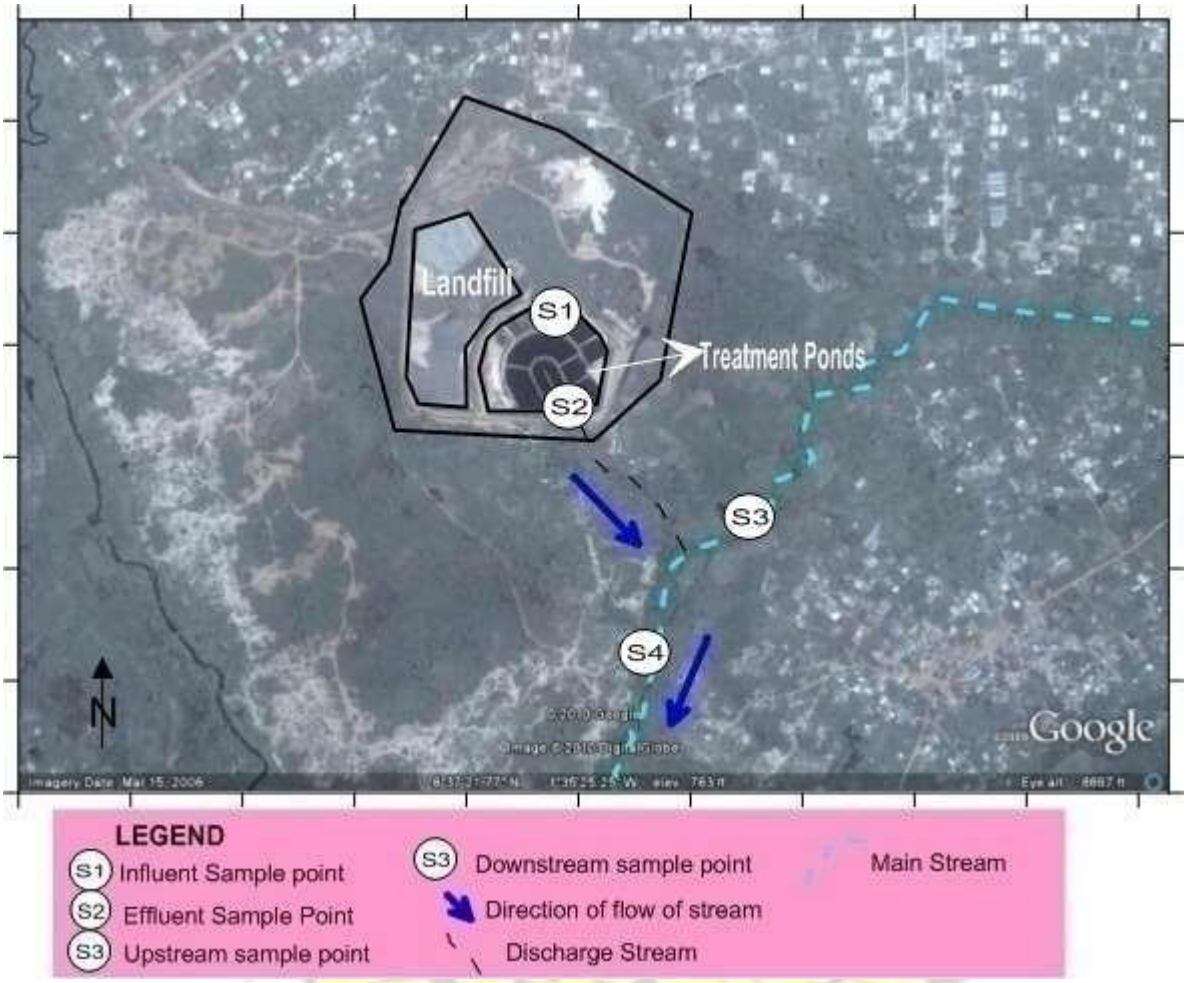

Figure 3.1 Map of study area showing sampling points (Boateng, 2010)

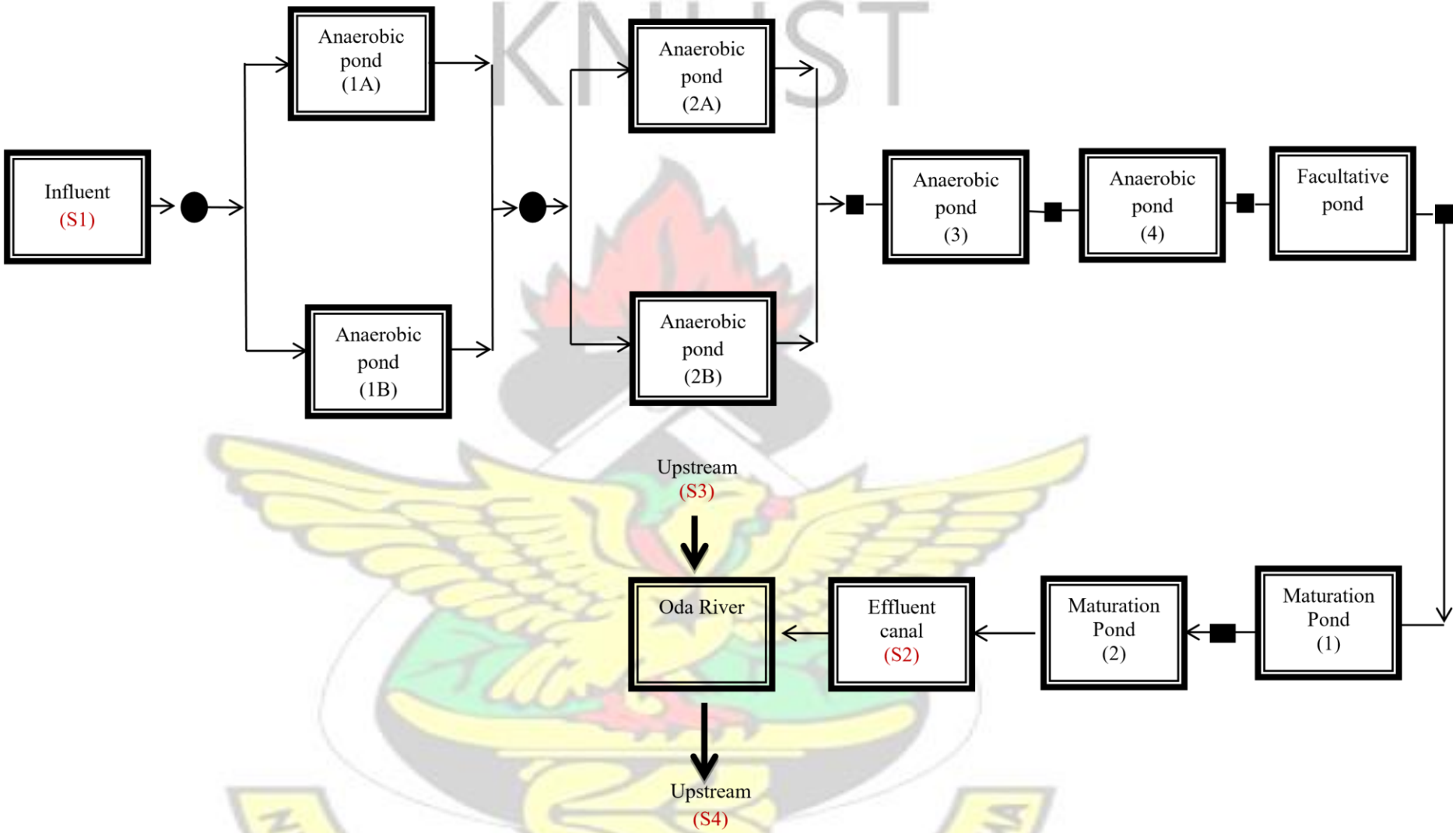

Figure 3.2 Flow chart of the Dompoase wastewater treatment plant and effluent discharge into the Oda River

3.3 Physico-chemical analyses

3.3.1 Apparatus

All glassware and plastic containers used were washed with detergent solution followed by soaking in 10% (v/v) nitric acid overnight. They were rinsed with distilled water followed by 0.5 5% potassium permanganate, rinsed with distilled and dried before use.

3.3.2 Reagents

Analytical reagent (AnalaR) grade chemicals (BDH Chemicals Ltd., Poole. England) were used throughout the study.

3.3.3 Determination of pH

The Mettler Toledo MP220 pH meter was used for the measurements of hydrogen ion concentration. The electrode of the meter was rinsed with distilled water and blotted dry. The sample was swirled and the electrode placed in the sample, ensuring that the entire sensing edge was submerged. The pH values were then recorded when the display on the meter was stable.

3.3.4 Total dissolved solids (TDS)

This was measured using the Hanna instrument HI 9032 microcomputer conductivity meter. The electrode for the measurement of TDS was rinsed with distilled water and blotted dry. The sample was swirled and the electrode placed in the sample, ensuring that the entire sensing edge was submerged. The TDS key selected was then selected.

The value displayed on the screen was recorded in mg/L.

3.3.5 Determination of biochemical oxygen demand (BOD)

Appropriate dilutions of samples were prepared and transferred into two BOD bottles (300 ml). Two other BOD bottles were also filled with dilution water to serve as blank. A stopper was placed on one of the bottles of each dilution and the blank. These were incubated for 5 days at 20 □C in an incubator. To the second set of bottles 1 ml of $MnSO_4$ solution was added, followed by 1 ml alkali- iodide-azide reagent. A stopper was placed carefully on each one of them to exclude air bubbles. The bottles were then inverted several times to obtain a complete mix. After the precipitate has settled sufficiently to leave a clear supernatant above the manganese hydroxide flocs, 1.0 ml of concentrated H_2SO_4 was added. The stopper was replaced and a complete dissolution was achieved by inverting the bottle several times. 200 ml of dissolved precipitate was then transferred into 500 ml beaker. It was titrated with standard $Na_2S_2O_3$ solution to obtain a pale yellow colour. Few drops of starch solution were added and titration continued for the blue colour to disappear. The dissolved oxygen (DO) for the final solution and incubated samples at the end of the fifth day were determined. The BOD was then calculated from the relation:

$BOD_5 = (D1 - D2)/P$ (mg/L)

D_1 = DO of sample immediately after preparation (mg/L) D_2

= DO of sample after 5 days of incubation at 20□C (mg/L)

P = decimal volumetric fraction of sample used.

3.3.6 Determination of chemical oxygen demand (COD)

One gram of mercury (II) sulphate was weighed into a reflux flask. 10 ml of sample was then added to the content of the flask. Again 10 ml of 0.04 M potassium dichromate was added, followed by 20 ml of concentrated H_2SO_4. Another flask was prepared as above using 10 ml of distilled water instead of sample as a blank. The outside of each of the flasks was cooled under running water. One milliliter (1 ml) of silver sulphate solution was added. The content was mixed well and the flask was fitted to the condenser. The heaters were switched on and the flask boiled under reflux for 2 hours. The flasks were removed and 45 ml of distilled water added to each. Again, the flasks were cooled under running water until quite cold and 2 – 3 drops of ferroin indicator was added. Titration was then conducted with standard ferrous ammonium sulphate (FAS) titrant to achieve reddish brown end point. The COD was calculated from the relation:

COD $=((A-B)\times 8000)/V$ mg (O_2)/l

A = volume of FAS used for blank (ml)

B = volume of FAS used for sample (ml)

M = Molarity of FAS

V = volume of sample used (ml)

3.3.7 Determination of total suspended solids (TSS)

A glass-fiber filter was weighed and placed on a filtration apparatus. The sample was mixed thoroughly and filtered to obtain a filtrate of 100 ml. The residue retained on the

filter paper was dried to a constant weight at 103 to 105□C. It was then cooled in a desiccator. The filter paper and dried residue were weighed. Suspended solids were calculated from the relation:

S.S = $((W2 - W1))/V \times 1000$ (mg/L)

W_1 = weight of filter paper (mg)

W_2 = weight of filter paper and dried residue (mg)

V = volume of sample (ml)

3.3.8 Determination of nitrogen (N)

The nitrogen content was quantified using a Kjeltec system 1002 distilling unit (Tecator; Höganäs, Sweden). 10 ml of sample was measured into 500 ml long-necked Kjeldahl flask. One spatula full of Kjeldahl catalyst (mixture of 1 part selenium + 10 parts $CuSO_4$ + 100 parts Na_2SO_4) and 30 ml concentrated H_2SO_4 were added. The mixture was digested for $1\frac{1}{2}$ to 2 hours until a clear and colorless or light greenish colour was obtained. The digest was allowed to cool and the fluid decanted into a 100 ml volumetric flask. The content of the flask was then filled to the mark with distilled water. The flask was then swirled for uniform mixing-10 ml aliquot of fluid was transferred by a pipette into Kjeldahl apparatus. 20 ml of 40% NaOH was then added to the digest mixture to provide the necessary alkaline conditions for the release of organic ammonia. A distillate was collected over 10 ml of 4% Boic acid and 3 drops of mixed indicator was added for

4 minutes. The presence of nitrogen gives a light blue colour. 100 ml of distilled water was then collected and titrated with 0.1N HCl till blue colour changed to grey, then finally flashed to pink. A blank determination was carried out as above using distilled water in place of the sample. The nitrogen content was calculated as follows:

14 g of N contained in one equivalent weight of NH_3

Weight of N in the sample= $(14 \times (A - B) \times N) / 1000$

Where: A = Volume of standard HCl used in the sample titration

B = Volume of standard HCl used in the blank titration

N = Normality of standard HCl

Note: Weight of sample used, considering the dilution and the aliquot taken for distillation:

$$\frac{10\ g \times 10\ ml}{100} = 1\ g$$

Thus, the percentage of total nitrogen in the sample:

$$\frac{14 \times (A - B) \times N \times 100}{1000 \times 1}$$

When N = 0.1 and B = 0

Total percentage Nitrogen = $A \times 0.713$

3.3.9 Determination of phosphorus (P)

The sample was filtered using 0.45-um membrane into 100 ml conical flask. 10 ml of filtrate was then pipetted into a 25 ml volumetric flask. 1.0 ml of molybdate reagent was added followed by 1.0 ml of dilute 1, 2, 4-aminonaptholsulfonic acid to reduce molybdate that is bound with phosphate. A blue solution was developed. The solution was made up with distilled water up to the 25 ml mark. The content was shaken vigorously and allowed to stand for 15 minutes. The percent transmission was then measured at 600 nm on a Hach DR 2010 Spectrophotometer and the percentage transmittance values obtained were recorded. The concentration of phosphorus was calculated as follows: percentage *T* values were converted to 2-log *T*. A graph of *P* standard solutions was plotted and actual concentrations of *P* values were obtained. The concentration of *P* in the extract was obtained by comparing the results with a standard curve plotted.

3.3.10 Determination of potassium (K)

Turbid samples were mixed with distilled water and 50 – 100 ml of each sample was measured into a conical flask. 5 ml of concentrated HNO_3 and few boiling chips were added. The sample was then heated on a hot plate at 70–80 °C until the lowest volume was attained. Heating was continued by adding small volumes of concentrated HNO_3 until a clear solution was obtained. The digested solution was then filtered with 0.45-um membrane and the filtrate diluted to the original volume with distilled water. 10 ml portions are then used for the potassium determination in the flame photometer. However before using a flame photometer (Jenway PFP7, UK) blank potassium calibration standards were prepared. The calibration standards and samples were aspirated over time

to secure a reliable average reading for each standard. Calibration curve for each standard was prepared and potassium concentrations determined using the

curve.

CALCULATIONS: Potassium (mg/L) = mg K/l in portion x D (Dilution factor)

3.3.11 Determination of lead (Pb)

50 ml of sample was measured into a digestion flask. 10 ml of $HClO_4$ and HNO_3 mixture in a ratio of 4: 9 respectively was added to the sample. The content of the flask was digested by heating until a clear mixture was obtained. It was then allowed to cool. The digest was made up to the 50 ml mark with distilled water and a standard curve was prepared. The level of lead was then recorded from an Atomic Adsorption Spectrum using the Buck Scientific model 210 VGP Atomic Absorption Spectrophotometer.

3.3.12 Determination of iron (Fe)

5 ml of concentrated nitric acid was added to l litre of sample. 100 ml of sample was then transferred into a beaker and 5 ml of distilled 1: 1 hydrochloric acid was added. The mixture was then heated on a water bath to a reduced volume of 20 ml. It was then filtered to remove any insoluble material. The pH of the digested sample was increased to 4 by drop-wise addition of 0.5 M sodium hydroxide standard solution. The sample was transferred into 100 ml volumetric flask and distilled water added up to the mark. The iron content of each digested sample was then determined using the Buck Scientific model 210 VGP Atomic Absorption Spectrophotometer.

3.4 Microbiological Analyses

3.4.1 Total coliform determination

Total coliforms were estimated using the most probable number method (MPN) according to Standard Methods (Anon, 1994). The decade dilution with three tubes inoculated at each dilution was used. Serial dilutions of 10^{-1} to 10^{-12} were prepared by filling 12 test tubes with 9 ml of distilled water each, labeled 10^{-1} to 10^{-12}. 1 ml of sample was then pipetted into the first test tube labeled 10^{-1}. The pipette was discarded and using a fresh pipette, the contents in the test tube were mixed thoroughly by pipetting up and down ten times. Using the same pipette 1 ml of diluted sample from the test tube 10^{-1} was pipetted into the test tube labeled 10^{-2}. The pipette was discarded and using a fresh pipette, the contents in the test tube were mixed thoroughly by pipetting up and down ten times. Using the same pipette 1 ml of diluted sample from the test tube labeled 10^{-2} was pipetted into the test tube labeled 10^{-3}. The process was repeated till all the dilutions were obtained. 1 ml of the diluted sample from each test tube labeled 10^{-1} to 10^{-12} was then inoculated into three tubes containing 5 ml of MacConkey Broth (OXOID® Basingstoke, Hampshire, England) with inverted Durham tubes and

incubated at 35 ^{o}C for 24 hours. Tubes showing change in colour and gas formation after 24 hours were considered presumptive positive for coliform bacteria. From the number and distribution of positive and negative reactions, count of the most probable number (MPN) of indicator organisms in the samples were estimated by reference to MPN statistical tables and expressed as MPN 100 ml^{-1} (Anon, 1994).

3.4.2 Faecal coliform determination

Faecal coliforms were estimated following the same procedure for total coliforms as in 3.4.1 above. However, tubes were incubated at 44 ^{o}C for 24 hours. Tubes showing change in color and gas formation after 24 hours were considered presumptive positive for faecal coliform bacteria. From the number and distribution of positive and negative reactions, count of the most probable number (MPN) of indicator organisms in the samples were estimated by reference to MPN statistical tables and expressed as MPN 100 ml^{-1} (Anon, 1994).

3.5 Statistical Analyses

Analytical methods were according to "standard methods for examination of water and wastewater" unless otherwise stated (AHPA, 1998). The data obtained were subjected to statistical analysis using Statistical Package for Social Sciences (SPSS) (Version 16) and sigma plot (Version 11). Holm-sidak test for ANOVA was used to test differences among all possible pairs of treatment. Statistical significance was then assessed at 95 % confidence interval ($P<0.05$).

CHAPTER FOUR

4.0 RESULTS

4.1 Removal efficiencies

Results obtained from the study showed that influent and effluent wastewater as well as water samples presented typical variations in contaminant concentrations with time. However, high contaminant concentrations were obtained for wastewater samples where as water samples showed lower concentrations. Generally, effluent concentrations were lower than those of influents. Mean percentage removal of 51.23 (± 15.34), 89.18 (± 8.43), 36.11 (± 34.65), 80.80 (± 17.85), 58.02 (± 41.05), 22.51 (± 15.53), 22.23 (± 18.93), 60.94 (± 42.79), 68.52 (± 26), 92.20 (± 3.82), -27.59 (± 34.40) and -10.24 (± 1.03) % for TDS, TSS, Fe, COD, BOD, N, P, , Pb, total coliforms, faecal coliforms, K and pH respectively.

Tables 4.1, 4.2 and 4.3 show physico-chemical parameters for wastewater and river water samples in the months of May, June and July. Furthermore, these results depict removal efficiencies of the treatment plant. Again, effluent permissible levels (EPA, 2000) for each parameter except Fe and K are indicated in these tables.

and biological

Table: 4.1 Physico-chemical qualities of influent treated for the month of May

Physico-chemical parameters	Acceptable limit	Influent (mg/L) S1	Treated Final Effluent (mg/L) S2	% Removal Efficiency	100 m Upstream (mg/L) S3	100 m Downstream (mg/L) S4
TDS	1000 mg/L	16680 (± 311.13)	5450 (± 127.28)	67.33	88.2 (± 1.83)	317.5 (± 0.71)
TSS	25 mg/L	35500 (± 282.84)	1120 (± 28.28)	96.85	7.5 (± 0.71)	26.5 (± 0.71)
Fe	*	12 (± 2.83)	9 (± 1.41)	25	1.39 (± 0.01)	1.7 (± 0.42)
COD	250 mg/L	73600 (± 395.98)	3280 (± 141.42)	95.54	68 (± 39.59)	324 (± 62.23)
BOD	50 mg/L	2575 (± 106.06)	210 (± 84.85)	91.84	20 (± 14.14)	77.5 (± 10.61)
N	75 mg/L	37075 (± 176.78)	9985 (± 21.2)	73.07	8245 (± 572.76)	8560(± 0.00)
P	2 mg/L	6765 (± 1958.69)	6605 (± 120.21)	2.37	42 (± 7.07)	115 (± 7.07)
K	*	246000 (± 1414)	234650 (±50133.87)	4.61	1450 (± 212.13)	11350 (±494.97)
Pb	0.1 mg/L	4.615 (± 0.021)	0.36 (± 0.04)	92.2	0.025 (± 0.01)	0.03 (± 0.01)
pH	6-9	5.915 (± 0.01)	6.455 (± 0.01)	***	5.8 (± 0.01)	5.66 (± 0.01)
Biological parameters	**Acceptable limit**	**Influent (MPN/ 100 ml)**	**Treated Final Effluent (MPN/ 100 ml)**	**% Removal Efficiency**	**100 m Upstream (MPN/ 100 ml)**	**100 m Downstream (MPN/ 100 ml)**
TC	400 MPN/ 100 ml	3.45×10^{11} ($\pm 6.3 \times 10^{10}$)	9.0×10^{9} (± 0.00)	97.39	2.35×10^{11} ($\pm 7.07 \times 10^{9}$	4.05×10^{14} ($\pm 4.17 \times 10^{14}$)
FC	400 MPN/ 100 ml	2.35×10^{11} ($\pm 7.07 \times 10^{9}$)	9.0×10^{9} ($\pm 2.12 \times 10^{7}$)	96.61	3.55×10^{9} ($\pm 3.45 \times 10^{9}$)	2.75×10^{10} ($\pm 1.76 \times 10^{10}$)

and biological

***Not available** *** Higher effluent pH compared with influent (9.13)

Table: 4.2 Physico-chemical qualities of influent treated for the month of June

Physico-chemical parameters	**Acceptable limit**	**Influent (mg/L) S1**	**Treated Final Effluent (mg/L) S2**	**% Removal Efficiency**	**100m Upstream (mg/L) S3**	**100m Downstream (mg/L) S4**
TDS	1000 mg/L	10400 (± 141.42)	6575 (± 530.33)	36.78	56.85 (± 3.46)	78.95 (± 3.46)
TSS	25 mg/L	12675 (± 35.36)	1200 (± 70.71)	90.53	15 (± 2.83)	25 (± 19.79)
Fe	*	35 (± 8.48)	8.75 (± 1.06)	75	1.215 (± 0.11)	1.285 (± 0.08)
COD	250 mg/L	29800 (± 282.84)	4200 (± 141.42)	85.91	148 (± 16.97)	148 (± 107.48)
BOD	50 mg/L	1890(± 0.00)	195 (± 63.64)	89.68	4.2 (± 1.70)	13.5 (± 1.27)
N	75 mg/L	32085 (± 7.67)	18540 (± 226.27)	42.22	5705 (± 120.21)	9270(± 0.00)
P	2 mg/L	10000(± 0.00)	6000 (± 707.11)	40	500(± 0.00)	1000(± 0.00)
K	*	136900 (± 14000)	224300 (± 21778.88)	**	1300 (± 141.42)	3045 (± 502.04)
Pb	0.1 mg/L	0.945 (± 0.05)	0.83 (± 0.21)	12.17	0.425 (± 0.11)	0.58 (± 0.11)
pH	6-9	5.785 (± 0.02)	6.475 (± 0.01)	***	5.665 (± 0.33)	5.495 (± 0.04)
Biological parameters	**Acceptable limit**	**Influent (MPN/ 100 ml)**	**Treated Final Effluent (MPN/ 100 ml)**	**% Removal Efficiency**	**100 m Upstream (MPN/ 100 ml)**	**100 m Downstream (MPN/ 100 ml)**
TC	400 MPN/ 100 ml	2.4×10^{14} (± 0.00)	9.3×10^{13} (± 0.00)	61.25	1.6×10^{9} (± 0.00)	1.6×10^{13} (± 0.00)

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FC	400 MPN/ 100 ml	2.35×10^{13} ($\pm 7.07 \times 10^{11}$)	2.35×10^{12} ($\pm 7.07 \times 10^{11}$)	90	4.0×10^{8} ($\pm$ 0.00)	5.8×10^{9} ($\pm 4.53 \times 10^{9}$)

*Not available ** Higher effluent K compared with influent (63.84) *** Higher effluent pH compared with influent (11.17) **Table: 4.3 Physico-chemical qualities of influent treated for the month of July**

Physico-chemical parameters	**Acceptable limit**	**Influent (mg/L) S1**	**Treated Final Effluent (mg/L) S2**	**% Removal Efficiency**	**100m Upstream (mg/L) S3**	**100m Downstream (mg/L) S4**
TDS	1000 mg/L	2950 ($\pm$ 14.14)	1487.5 ($\pm$ 6.36)	49.58	56.5 ($\pm$ 0.85)	76.95 ($\pm$ 0.07)
TSS	25 mg/L	4805 ($\pm$ 7.07)	954 ($\pm$ 8.49)	80.15	7 ($\pm$ 1.41)	12 ($\pm$ 1.41)
Fe	*	1.8 ($\pm$ 0.28)	1.65 ($\pm$ 0.07)	8.33	0.755 ($\pm$ 0.12)	1.4 ($\pm$ 0.20)
COD	250 mg/L	6260($\pm$ 0.00)	2445 ($\pm$7.07)	60.94	15 ($\pm$ 1.41)	32 ($\pm$ 1.41)
BOD	50 mg/L	1530 ($\pm$ 14.14)	1230 ($\pm$ 14.14)	19.68	4.65 ($\pm$ 0.21)	6.9 ($\pm$ 0.42)
N	75 mg/L	34580 ($\pm$ 28.28)	14260 ($\pm$ 14.14)	58.76	6970 ($\pm$ 42.42)	8910($\pm$ 0.00)
P	2 mg/L	8385 ($\pm$ 7.07)	6305 ($\pm$ 148.49)	24.81	310($\pm$ 0.00)	715 ($\pm$ 7.07)
K	*	185750 ($\pm$ 777.810	229450 ($\pm$ 14.14)	**	1400($\pm$ 0.00)	7150 ($\pm$ 14.14)
Pb	0.1 mg/L	2.785 ($\pm$ 0.04)	0.6 ($\pm$ 0.13)	78.46	0.23 ($\pm$ 0.07)	0.305 ($\pm$ 0.05)
pH	6-9	5.85 ($\pm$ 0.014)	6.46 ($\pm$ 0.014)	***	5.73 ($\pm$ 0.23)	5.575 ($\pm$ 0.01)
Biological parameters	**Acceptable limit**	**Influent (MPN/ 100 ml)**	**Treated Final Effluent (MPN/ 100 ml)**	**% Removal Efficiency**	**100 m Upstream (MPN/ 100 ml)**	**100 m Downstream (MPN/ 100 ml)**
TC	400 MPN/ 100 ml	1.696×10^{10} ($\pm 1.84 \times 10^{10}$)	9×10^{9}	46.93	2.35×10^{10}	4.05×10^{10} (4.2×10^{10})

and biological

			(± 0.00)		(7.1× 10^8)	
FC	400 MPN/ 100 ml	2.35 × 10^9 (± 7.07 × 10^7)	2.35 × 10^8 (± 7.07 × 10^6)	90	4 × 10^8 (± 0.00)	5.8 × 10^9 (4.52 × 10^9)

*Not available ** Higher effluent K compared with influent (23.53) *** Higher effluent pH compared with influent (10.43)

4.1.1 Removal efficiency for TDS

High TDS values with mean influent concentration of 16680 (± 127.28) mg/L in wastewater were recorded for May. Mean influent concentrations of 10400 (± 141.42) mg/L and 2950 (± 14.14) mg/L in wastewater were obtained for June and July respectively. Again, mean effluent concentration of TDS in wastewater for May was lower than that of June but higher than that of July ($P<0.05$). Effluent concentrations were above the recommended EPA standard of 1000 mg/L. Moreover, TDS concentrations of water samples from upstream of Oda River were low. Mean values of 88.2 (± 1.83) mg/L, 56.85 (± 3.46) mg/L and 56.5 (± 0.85) mg/L were obtained in May, June and July respectively. These concentrations increased slightly to 317.5 (± 0.71) mg/L, 78.95 (± 3.46) mg/L and 76.95 (± 0.07) mg/L downstream the river for the same months. However, no significant changes were observed between effluent and downstream values of TDS as well as upstream and downstream values ($P>0.05$).

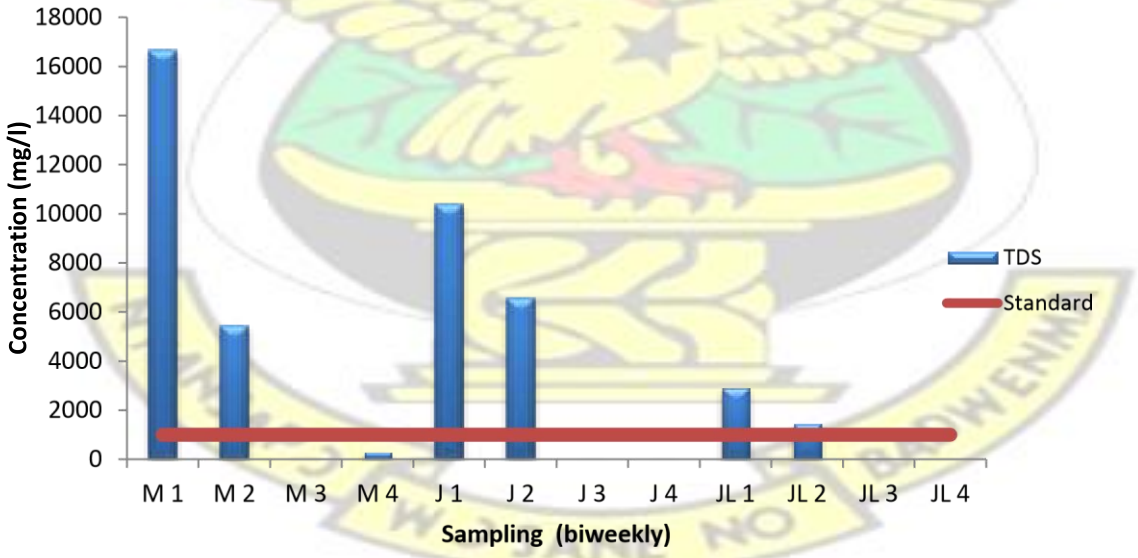

Figure 4.1 Mean TDS values for river water and wastewater samples compared with EP standards

4.1.2 Removal efficiency for TSS

High influent concentrations of TSS in wastewater with mean concentrations of 35500 (± 28.84) mg/L were recorded for May. However, mean influent concentrations decreased for the subsequent months. A mean TSS concentration of 12675 (± 35.36) mg/L was recorded for June and 4805 (± 7.07) mg/L for July. Significant effluent concentrations ($P<0.05$) of 1120 (± 28.28) mg/L, 1200 (± 70.71) mg/L and 954 (± 8.49) mg/L were recorded for these same months. However, these effluent values were above the recommended EPA standard of 25 mg/L for TSS. Total suspended solids in river water samples were very low with 7.5 (± 0.71) mg/L, 15 (± 2.83) mg/L and 7 (± 1.41) mg/L mean values for upstream concentrations. 26.5 (± 0.71) mg/L, 25 (± 19.79) mg/L and 12 (± 1.41) mg/L on the other hand were recorded for downstream concentrations of river water samples. But, there was no significant difference ($P>0.05$) between upstream and downstream values of TSS.

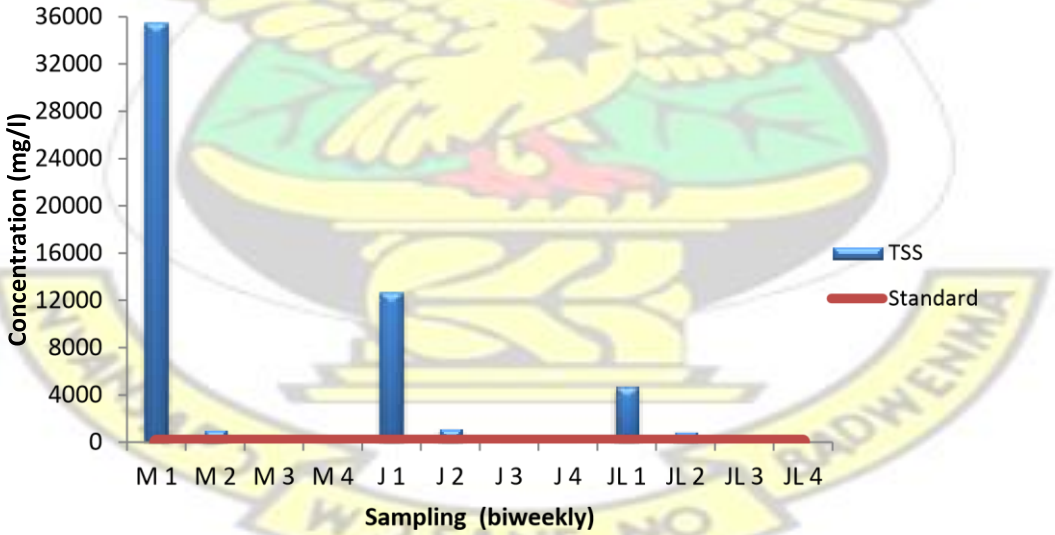

Figure 4.2 Mean TSS values for river water and wastewater samples compared with EPA standards

4.1.3 Removal efficiencies for Fe and Pb

General concentrations for heavy metals in both wastewater and water samples were lower than other parameters. Influent mean values of 12 (± 2.83) mg/L, 35 (± 6.48) mg/L and 1.8 (± 0.25) mg/L for Fe were recorded for the three sampling months. Corresponding effluent concentrations of 9 (± 1.41) mg/L, 8.75 (± 1.06) mg/L and 1.65 (± 0.07) mg/L were obtained (P>0.05). For lead, mean concentrations of 4.615 (± 0.021) mg/L, 0.945 (± 0.05) mg/L and 2.785 (± 0.05) mg/L were found in influent wastewater. Corresponding mean effluent concentrations of 0.36 (± 0.04) mg/L, 0.83 (± 0.21) mg/L and 0.6 (± 0.13) mg/L were recorded for May, June and July respectively (P<0.05). These effluent concentrations for lead were above the recommended EPA standard of 0.1 mg/L. Furthermore, downstream concentrations for both Fe and Pb were slightly higher, compared with upstream concentrations for all the months under review. There was no statistical significant difference between upstream and downstream concentrations of both metals (P>0.05).

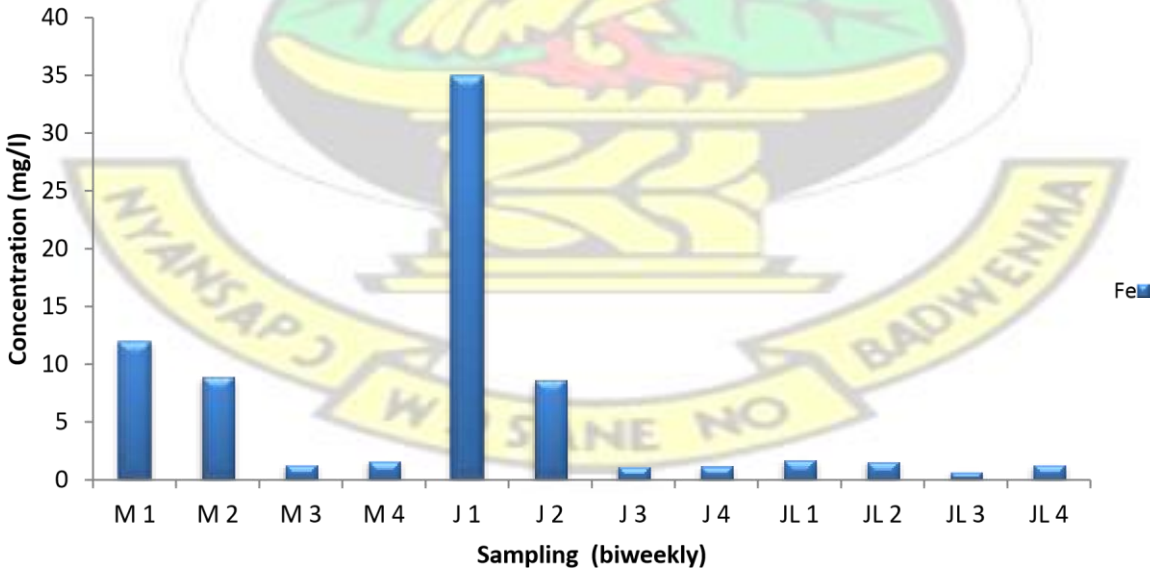

Figure 4.3 Mean Fe values for river water and wastewater samples

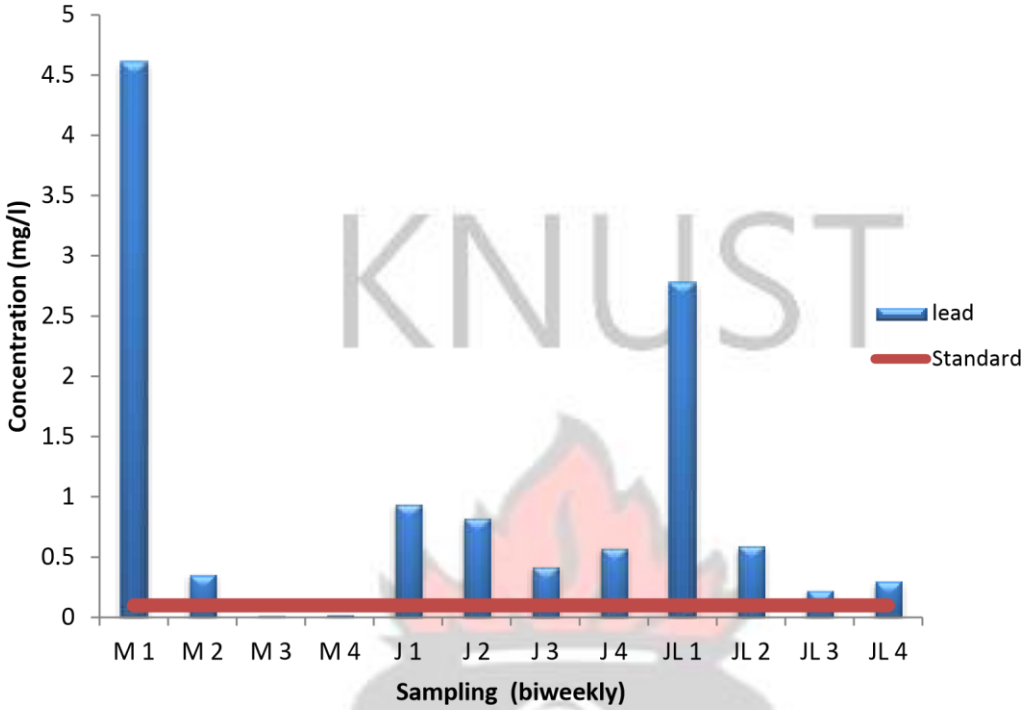

Figure 4.4 Mean Pb values for river water and wastewater samples compared with EPA standards

4.1.4 Removal efficiencies for N, P and K

The nutrient concentrations of wastewater and water samples generally varied significantly ($P<0.05$). Potassium (K) concentrations in wastewater and water samples were higher compared with that of N and P. Influent mean concentrations for P with respect to the months under review ranged between 6765 (± 1938.69) – 10000 mg/L. Ranges between 32085 (± 7.07) –37075 (± 176.78) mg/L and 185750 (± 777.81) – 24600 (± 1414) mg/L were also obtained for nitrogen and potassium respectively. Effluent concentrations reduced slightly for K, N and P ($P<0.05$). Effluents for N and P were above the recommended EPA standard of 75 mg/L and 2 mg/L respectively.

Moreover, upstream concentrations of nutrients were low as compared with downstream concentrations. Upstream mean concentrations of 8245 (± 572.76) mg/L, 5705 (± 120.21) mg/L and 6970 (± 42.42) with corresponding downstream concentrations of 8560 mg/L, 9270 mg/L and 8910 mg/L were recorded for N. But no significant difference was recorded among upstream and downstream mean values of N ($P<0.05$).

Again, P also recorded upstream concentrations of 42 (± 7.07) mg/L, 500 mg/L and 310 (± 7.07) mg/L against downstream concentration of 115 (± 7.07) mg/L, 1000 mg/L and 715 mg/L ($P<0.05$) for May, June and July respectively.

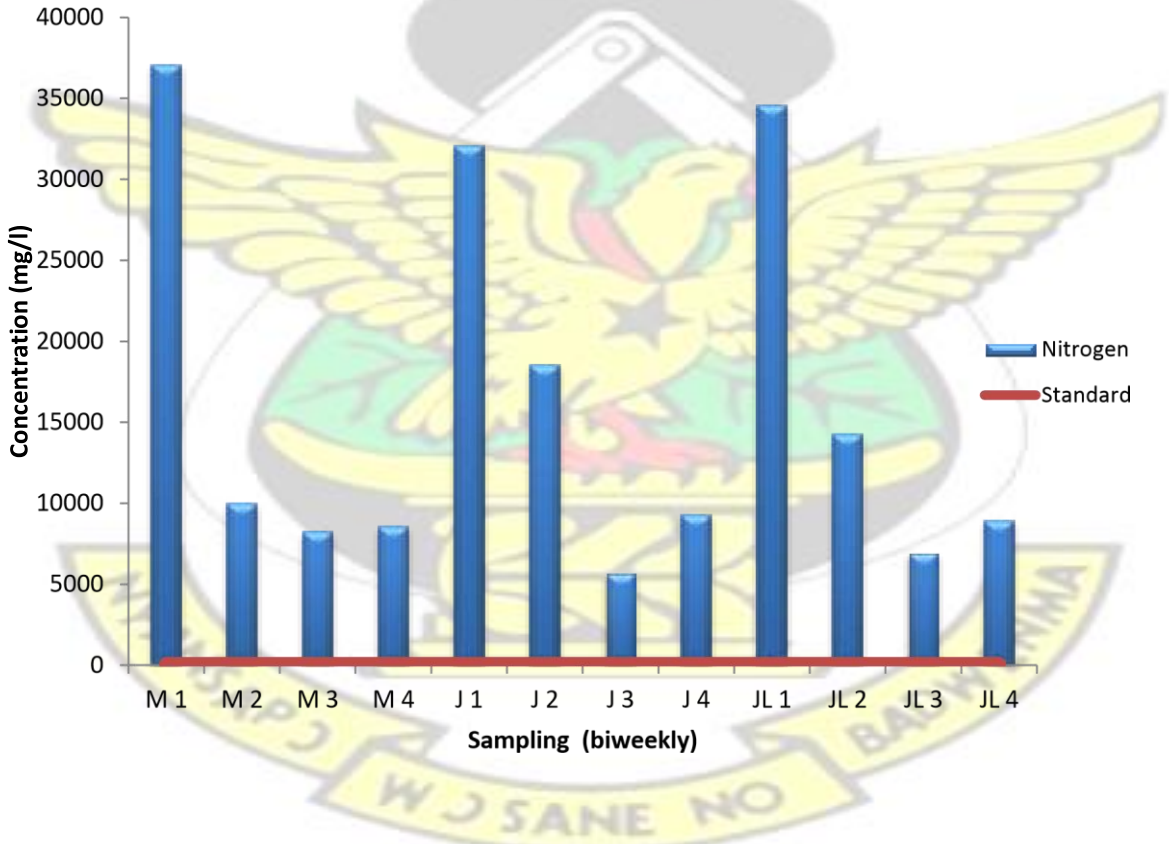

Figure 4.5 Mean N values for river water and wastewater samples compared with EPA standards

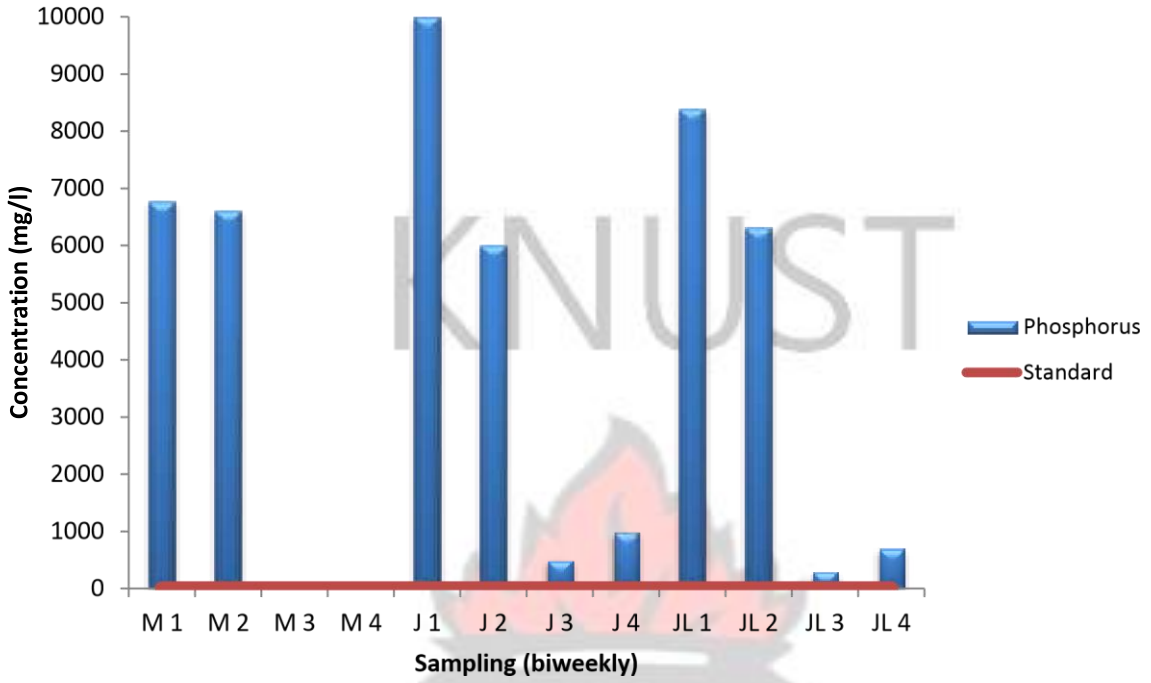

Figure 4.6 Mean P values for river water and wastewater samples compared with EPA standards

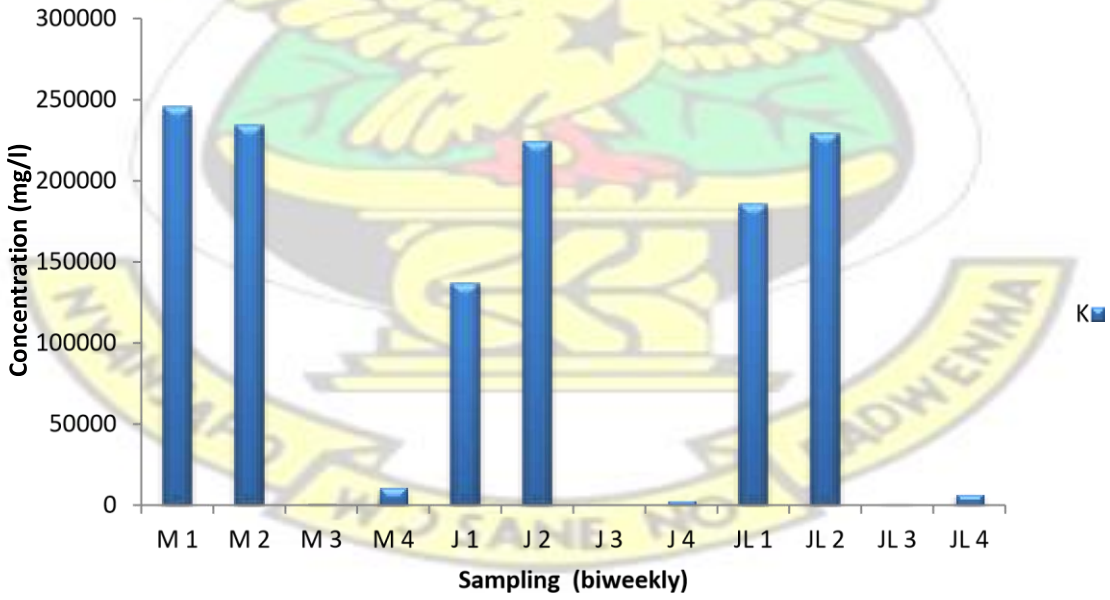

Figure 4.7 Mean K values for river water and wastewater samples

4.1.5 Removal efficiencies for COD and BOD

Figures 4.8 and 4.9 represent organic matter concentrations for COD and BOD respectively. Values for COD were higher than that of BOD for wastewater and water samples. Mean influent concentrations of 73600 (± 395.98) mg/L, 29800 (± 282.84) mg/L and 6260 (± 0.00) mg/L were obtained for May, June and July respectively. Corresponding mean effluent values of 3280 (± 141.42) mg/L, 4200 (± 141.42) mg/L and 2445 (± 7.07) mg/L were obtained for the same months with a statistical significant difference of $P<0.05$. However, effluent did not meet the recommended standard of 250 mg/L for COD. For BOD, influent values of 2575 (± 106.06) mg/L, 1890 (± 0.00) mg/L and 1530 (± 14.14) mg/L were obtained for wastewater as depicted by fig. 4.9. Effluent concentrations of 210 (± 84.85) mg/L, 195 (± 63.64) mg/L and 1230 (± 14.14) mg/L were obtained in wastewater for the same months ($P<0.05$). These were above the recommended EPA standard of 50 mg/L. Furthermore, concentrations of COD and BOD for river water samples upstream were lower than downstream concentrations. In contrast, mean COD concentration upstream and downstream for the month of June was the same. A value of 148 mg/L was obtained. There was no statistical significant difference between upstream and downstream concentrations of both parameters ($P>0.05$).

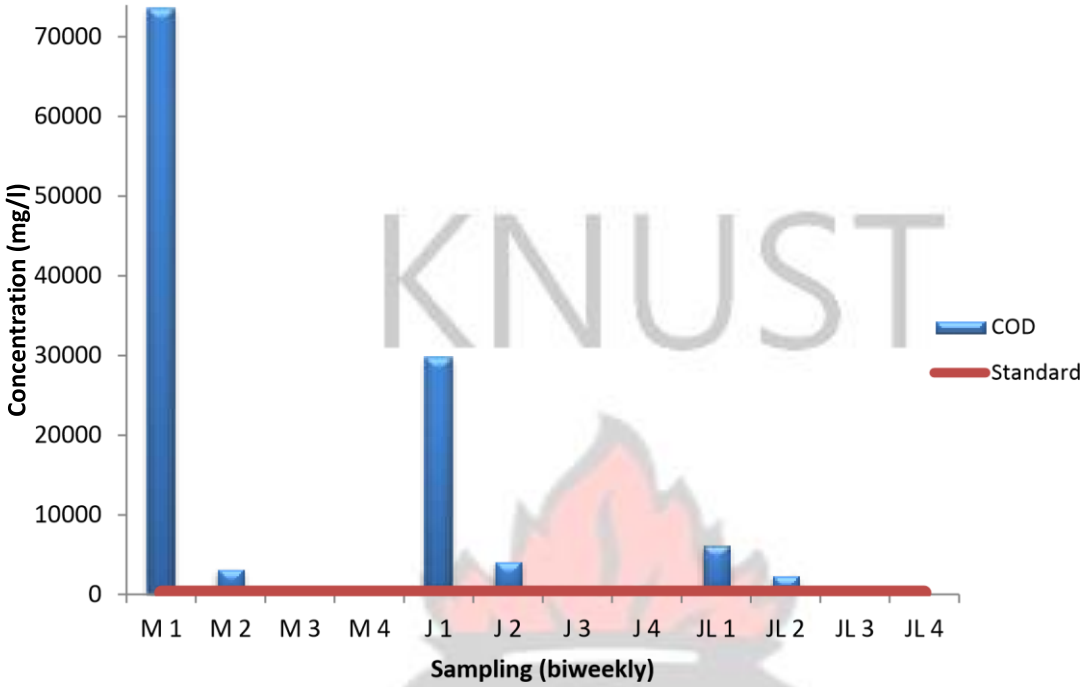

Figure 4.8 Mean COD values for river water and wastewater samples compared with EPA standards

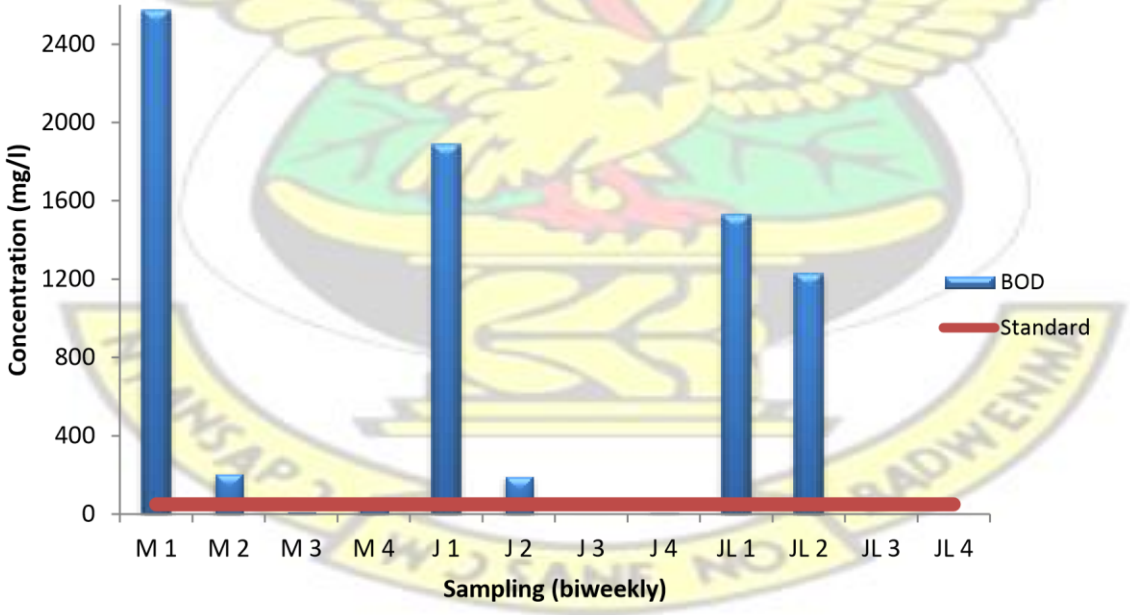

Figure 4.9 Mean BOD values for river water and wastewater samples compared with EPA standards

4.1.6 Removal efficiencies for faecal and total coliforms

General total and faecal coliform numbers did not vary significantly ($P>0.05$). Figures 4.10 and 4.11 depict reduction in total and faecal coliforms for wastewater through the treatment plant as well as water samples before and after effluent discharge from treatment plant. Mean influent concentrations of 3.45×10^{11} ($\pm 6.3\times 10^{10}$), 2.4×10^{14} ($\pm$ 0.00) and 1.696×10^{10} ($\pm 1.84 \times 10^{10}$) MPN/100 ml were obtained for May, June and July respectively, for total coliforms. Corresponding effluent concentrations of 9.0×10^{9} ($\pm$ 0.00), 9.3×10^{13} ($\pm$ 0.00) and 9.0×10^{9} ($\pm$ 0.00) MPN/100 ml were obtained for the same months. For faecal coliforms mean influent concentrations of 2.35×10^{11} ($\pm 7.07 \times 10^{9}$), 2.35×10^{13} ($\pm 7.07 \times 10^{11}$) and 2.35×10^{9} ($\pm 7.07 \times 10^{7}$) MPN/100 ml were observed for May, June and July respectively. Changes in effluent concentrations were observed for May, June and July with corresponding counts 9.15×10^{8} ($\pm 2.12 \times 10^{7}$), 2.35×10^{12} ($\pm 7.07 \times 10^{7}$) and 2.35×10^{9} ($\pm 7.07 \times 10^{6}$) MPN/100 ml. Mean effluent concentration of both faecal and total coliforms were above the recommended EPA standard of 400 counts per 100 ml.

River water samples presented low total and faecal coliform concentrations as compared with that of the wastewater. Mean total coliform concentrations of 2.35×10^{11} ($\pm 7.07 \times 10^{9}$), 1.6×10^{9} ($\pm$ 0.00) and 2.35×10^{11} ($\pm 7.07 \times 10^{8}$) MPN/100 ml were obtained for May, June and July upstream the Oda River. Corresponding downstream concentrations of 4.05×10^{14} ($\pm 4.17 \times 10^{14}$), 1.6×10^{13} ($\pm$ 0.00), 4.05×10^{10} ($\pm 4.2 \times 10^{10}$) MPN/100 ml

were obtained for these same months. Faecal coliform concentrations reduced downstream. Mean upstream concentrations of 3.55×10^9 ($\pm$ 3.45×10^9), 4.0×10^8 ($\pm$ 0.00) and 4.0×10^8 ($\pm$ 0.00) MPN/100 ml were obtained for total coliforms. On the other hand, mean concentrations of 2.75×10^{10} ($\pm$ 1.76×10^{10}), 5.8×10^9 ($\pm$ 4.53×10^9) and 5.8×10^9 ($\pm$ 4.52×10^9) MPN/100 ml for downstream concentrations where observed for the same period.

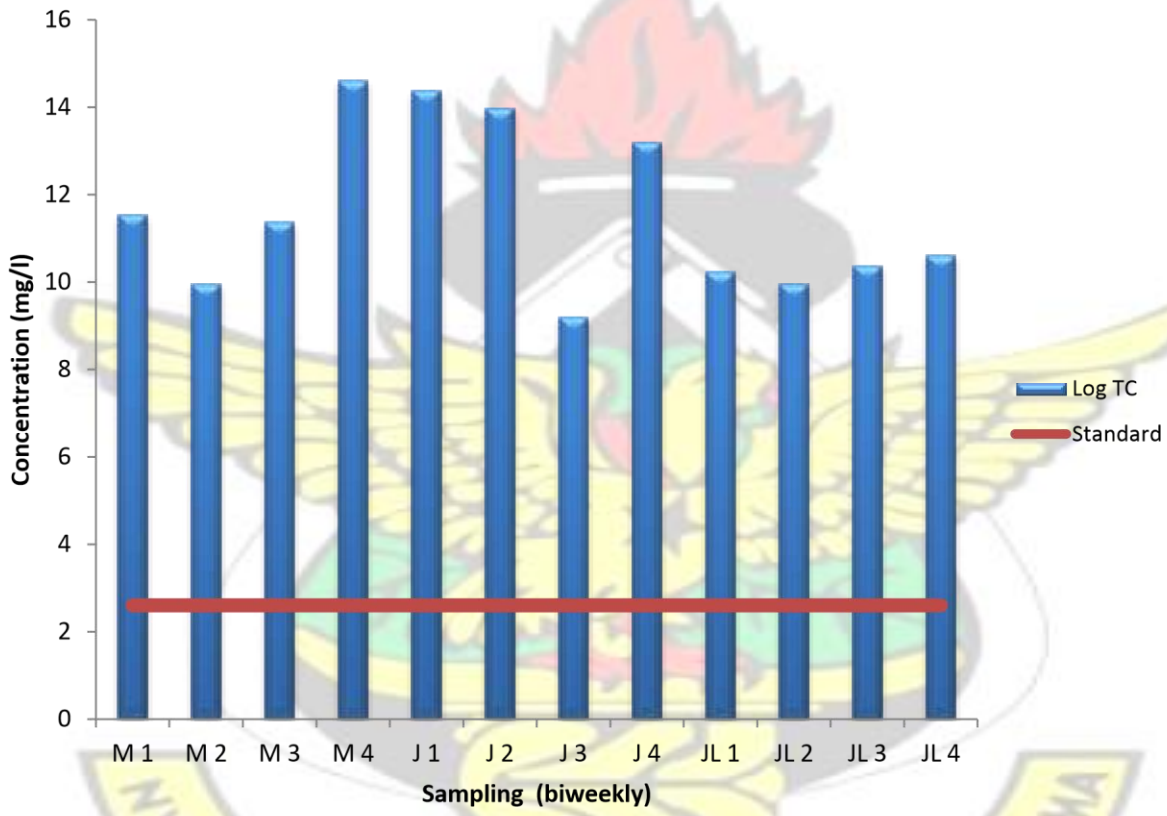

Figure 4.10 Mean total coliform values for river water and wastewater samples compared with EPA standards

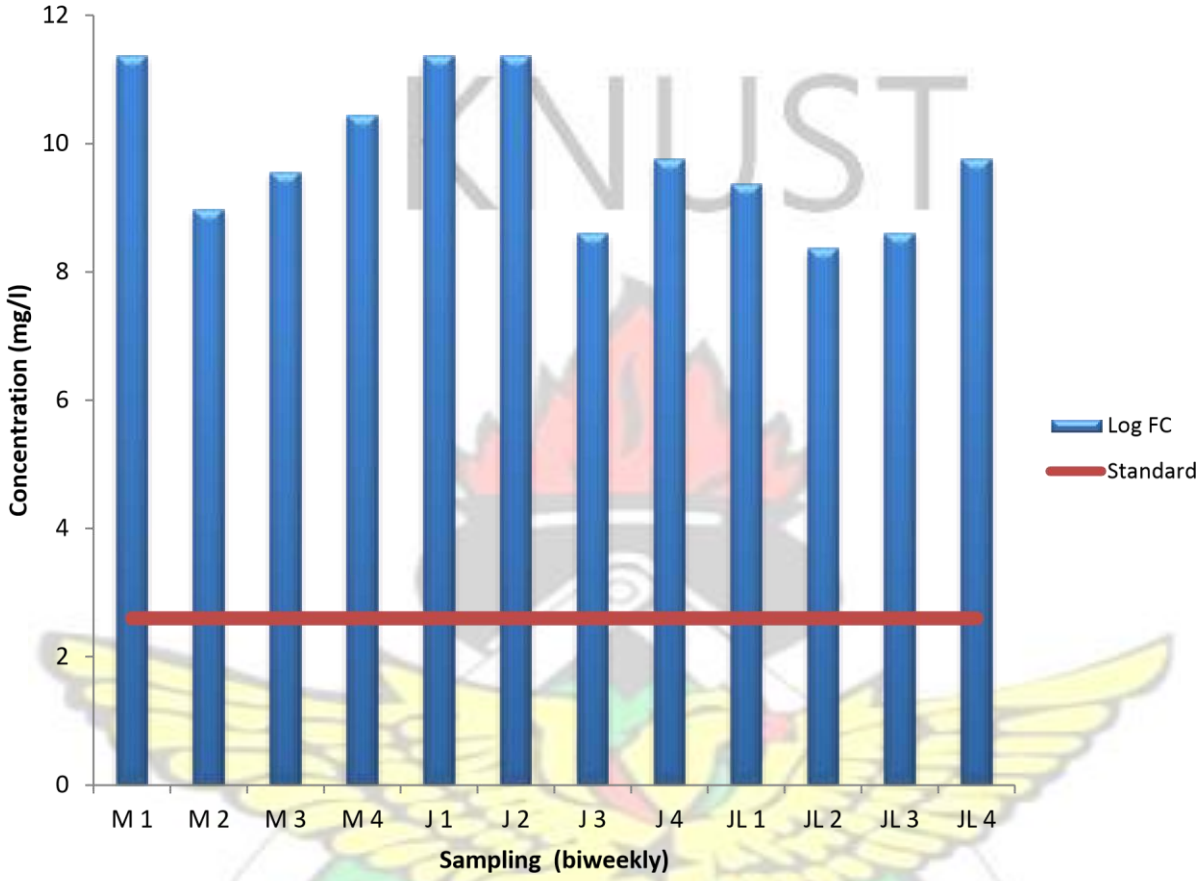

Figure 4.11 Mean faecal coliform values for river water and wastewater samples compared with EPA standards

4.2 Changes in pH

The pH values vary significantly ($P < 0.05$) for the sampling months. Mean influent pH for wastewater was in a range of 5.79 (± 0.02) – 5.92 (± 0.01). Significant changes in effluent concentrations ($P<0.05$) were within a range of 6.46 (± 0.01) – 6.48 (± 0.01).This conformed to the recommended EPA standard of 6-9. Upstream concentration of river

water samples had higher pH values compared with downstream concentrations ($P<0.05$). Values of 5.8 (± 0.01), 5.67 (± 0.33) and 5.73 (± 0.23) were obtained for May, June and July respectively. Downstream concentrations on the other hand, were 5.66 (± 0.01), 5.5 (±0.04) and 5.575 (± 0.01) for the same months.

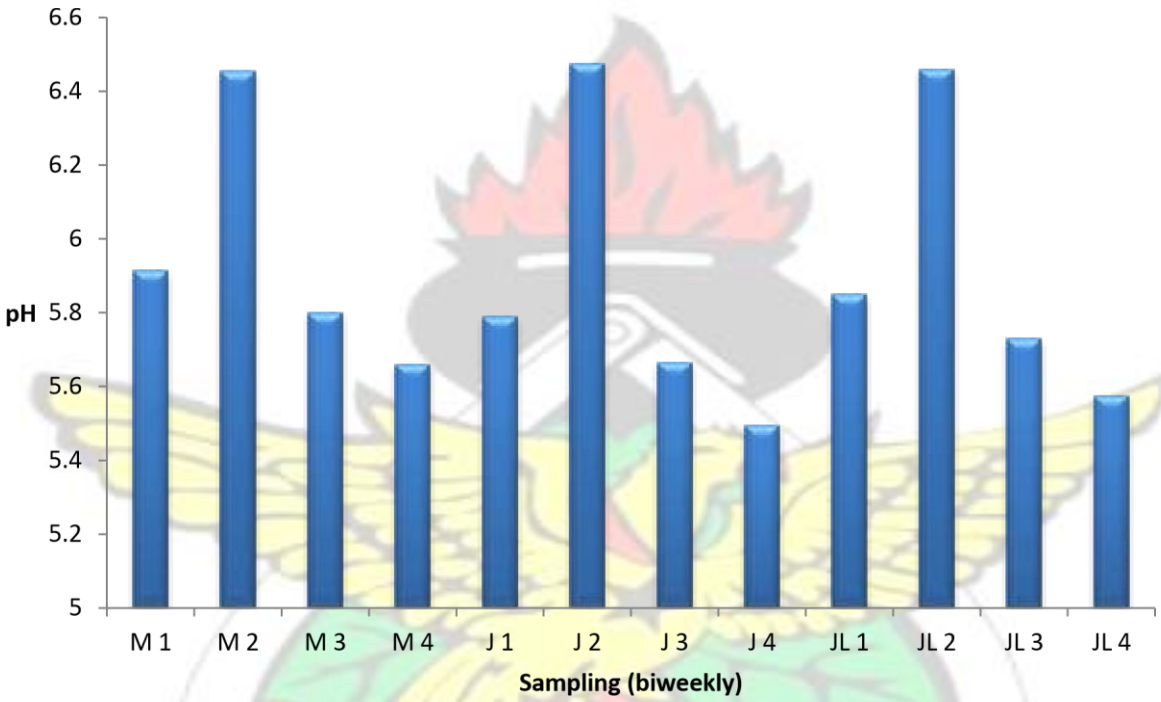

Figure 4.12 Mean pH values for river water and wastewater samples compared with EPA standards

CHAPTER FIVE

5.0 DISCUSSION

5.1 Total dissolved solids

Total dissolved solid contents indicate the ability of water to dissolve the organic and inorganic constituents. A high concentration of dissolved solids increases the density of

dissolving water and reduces the solubility of oxygen gas, creating danger for aquatic life (Bangash *et al.*, 2006). From the results, the total dissolved solid concentrations gradually decreased from influent to the final effluent. This gradual decrease becomes evident in a mean percentage removal efficiency of 51.23 (± 15.34) %. The poor percentage removal can be attributed partly to high overloading of the ponds and heavy accumulation of sludge in the primary sedimentation ponds. However, differences in the mean values of influent and effluent treatment groups was significant ($P<0.05$), an indication that the reduction was as a result of the treatment process within the waste stabilization pond system.

But, mean effluent concentrations were above the Ghana EPA effluent guideline of 25 mg/L (fig.4.1). Hence, these TDS concentrations can automatically influence the quality of the Oda River. Elevated TDS can reduce water clarity, hinder photosynthesis, and lead to increased water temperatures (Mason, 1998). Furthermore, no significant difference was recorded between the means of effluent, upstream and downstream values of TDS. This indicates that the assimilative ability of the Oda River to reduce TDS concentrations is minimal and any reductions might be due to other external factors.

5.2 Total suspended solids

Suspended solids occur naturally in surface waters as a result of erosion, transport of material from the bottom of the river and tributary inflows. They are also added by erosion caused by human activity and by effluents. Municipal wastewater effluents are responsible for a long-term continuous input of suspended solids to the environment (Horner *et al.*, 1994). From the results changes in TSS concentrations from influent to effluent was high

(fig.4.2). This is evident in a higher mean removal percentage of 89.18 (± 8.43) % with a significant difference of $P<0.05$, an indication that the reduction was as a result of the treatment process within the waste stabilization pond system.

However, this reduction was not reflected in effluent quality compared with the EPA guideline of 25 mg/L (fig.4.2). This result is in line with work done by Reed *et al*., (1988) and Bitton, (2005). Both works states that Oxidation pond effluents often have a high level of suspended solids composed mostly of algal cells and wastewater solids. Accumulation of sludge is a major contributing factor to these changes in TSS. Suspended solids released into receiving waters, mainly effluent discharges, can cause a number of direct and indirect environmental effects, including reduced sunlight penetration smothering of spawning grounds and physical harm to fish (Horner *et al*., 1994). Trace metals and organic contaminants, harmful to human health and the environment, can adhere to TSS and enter receiving water bodies through effluents
(Nantel, 1996). Again, the algae often exert an oxygen demand in the receiving stream (Bitton, 2005) and the growth and survival of some species may be affected.
Increases in downstream values of TSS were evident, mainly due to high concentrations of effluent discharge. Moreover, no significant change was observed among upstream and downstream values after discharge of effluent. This indicates that reduction in TSS after discharge is minimal. Thus, effluent need to undergo major treatment, if possible by intermittent sand filters, micro strainers or constructed wetlands as suggested by Steinmann *et al*., (2003) before discharge in surface water.

5.3 Heavy metals

Metals ions are adsorbed onto suspended solids in the wastewater treatment plants, settle and are removed (Gökhan, 2009). Primarily, removal of metals is achieved in anaerobic ponds where settling removes a proportion of metals which are either insoluble or adsorbed onto particulate matter. Further metal removal occurs in the secondary biological stage of wastewater treatment, usually through adsorption of dissolved metals or fine particulate metals onto sludge flocs, as reported by Gökhan (2009). But Brown *et al.*, (1973), found that for some metals (chromium, copper and lead) removal efficiency was greater in secondary treatment than in a primary process, while for zinc, the average removal percentage was similar at both stages.

Therefore, influent values are generally higher than effluent values. Moreover, changes were seen periodically in the metal concentrations (Pb, and Fe). This is because in aerobic maturation ponds Fe^{2+} will convert to Fe^{3+} leading to precipitation of iron as $Fe(OH)_3$; which will in turn adsorb other metal ions thus reducing effluent levels and improving removal efficiency.

But removal efficiencies for Fe and Pb were 36.11 (± 34.45) % and 60.94 (± 42.79) % respectively. This shows a lesser decrease in the concentrations of Fe and Pb. An indication that rainfall had a positive effect on the pond system by causing dilution of the ponds and reducing the retention time, such that metals were not able to adhere effectively onto particulate matter. ANOVA results indicated significant changes in the concentration of Pb ($P<0.05$). This shows that the pond achieved some form of reduction in terms of lead. For Fe, no significant difference was recorded among mean values.

Because metal values in the waste stabilization pond effluents were higher than metal values in the receiving environment, higher concentrations were observed for both Pb and Fe downstream. These changes in metal concentrations can be attributed to dilution caused by the receiving water body, the variable flow rate of the receiving water and the difference between pH values between the treated and receiving water. This accession is supported by work done by Shi *et al*., (1998). However, no significant differences were recorded among upstream and downstream parameters of each metal ($P>0.05$).Changes in concentrations within the river for theses metals might not occur. Moreover, effluent concentrations for lead exceeded the EPA recommended standard of 0.1 mg/L (fig.4.4). Metals are persistent, continuing to cause long term effects in the environment through deposition to and remobilization from sediments. Additionally, some metals biomagnify in the food chain, thereby causing indirect effects on predators (CCME, 2006). Long term impacts of these metals as a result of biomagnifications and bioaccumulation is a source of concern. Hence effluents must be treated to the required recommended standards.

Generally, the complex formation and dissolution value for each metal is different (Lester, 1983). Metal removal efficiencies in waste stabilization ponds vary by metal and type of waste stabilization pond system, but in general, removal improves with the number of ponds in the system, particularly if the final ponds are aerobic maturation ponds (Craggs, 2005). The presence of heavy metals in the wastewater indicates the possibility of industrial waste entering the treatment plant.

5.4 Nutrients

Some nutrients, particularly phosphorus and nitrogen, are essential for plant production in all aquatic ecosystems. However, an oversupply of nutrients can lead to the growth of large algal blooms and extensive weed beds.

In rivers and streams, the addition of nutrients tends to encourage the growth of periphyton, the stringy algae that grow on rock surfaces, and rooted aquatic plants. Excessive enrichment, however, can result in deoxygenation of the water and a consequent decline in the productivity of periphyton, as well as reductions in populations of bottom-dwelling invertebrates (Meena *et al*., 2010). Discharging wastewater effluent rich in nutrients into receiving rivers poses a number of problems on receiving water bodies, including impact on human health and marine ecology (Mayo and Bigambo, 2005). Therefore, nutrients must be removed to preserve water and the environment and to protect aquatic life and health of water users downstream.

5.4.1 Nitrogen

Nitrogen removal is essential to reduce ammonia toxicity to aquatic life, reduce the oxygen demand in receiving water bodies, prevent acidification of ground water aquifers due to nitrification in the soil and reduce the potential for surface water eutrophication (Kashaigili *et al*., 2005). Even though high influent nitrogen concentrations were recorded, reduction in effluent nitrogen was evident. This is depicted in figure 4.5. A mean percentage removal of 58.02 (± 15.43) % was recorded for the final effluent. Gradual changes in concentrations can be attributed to the uptake of inorganic nitrogen by algae, followed by sedimentation and volatilization of ammonia gas from the surface of the

system (Metcalf and Eddy, 2001). However, this process is greatly affected by increased sludge deposition that affects algal growth and dilution from rainfall that affect algal functioning by reducing the retention time of influents in sedimentation tanks. There influent concentrations greater than 600 m^3/day is likely to affect the removal efficiencies of some parameters and nitrogen removal in particular. Even though, a lower percentage removal was recorded, changes in influent and effluent values were significant ($P<0.05$). This can be attributed to performance of the pond.

Effluent nitrogen concentrations were higher than upstream concentrations of nitrogen in river Oda. This caused an increase in downstream concentrations of N with a significant difference at $P<0.05$. Use of nitrogen by resident microorganisms and absorption by sediments that play a major role in reductions of N were poor. This is in line with work done by Erni *et al*., (2010) which states that concentrations for N downstream in Kumasi, is 14 times higher than upstream values as a result of failing sanitation.

Moreover, effluent nitrogen exceeded the recommended EPA standard of 75 mg/L (fig.4.5) and this implies that the wastewater effluent water presents significant risk of polluting the receiving water and other forms of environmental damage.

5.4.2 Phosphorus

Phosphorus is an essential macronutrient that is a limiting factor to plant growth. It is essential to all life as a component of nucleic acids and a universal energy molecule (Sharpley *et al.,* 1994). In excess, phosphorus triggers eutrophic conditions which involve the prolific growth of algal and other aquatic plants. Algal growths can have

lethal impacts on aquatic life and, at high concentrations, can be toxic. The absorption of sunlight by algal blooms reduces the amount of light reaching aquatic plants in sediment. If an algal bloom is prolonged, aquatic plants will die. Large amounts of decaying algae result in the consummation of large quantities of oxygen by the bacteria and fungi that break it down. This results in the dramatic reduction of oxygen concentrations in the water column, particularly at night (Galbrand *et al.*, 2008).

From the results, total phosphorus was abnormally high in all influent and effluent concentrations. Gradual decrease from influent to effluent with a mean percentage removal of 22.39 (± 18.93) % was observed (fig.4.6). However, significant differences were observed among mean values ($P<0.05$). This is in line with work done by Picot *et al.*, (1992) which states that phosphorus removal in waste stabilization pond is highly variable, with an average removal of between 15 and 50%. This is a good justification to state that phosphorus removal from the wastewater was not effective. An elevated pH can cause phosphates to precipitate by complexation with metal ions such as calcium, magnesium, and iron present in the wastewater causing a reduction in phosphorus concentrations (Powell *et al.*, 2008). But pH in both influent and effluent wastewater ranged between 5.79 and 6.48. This might play a part in low phosphorus reduction. Growth of microalgae also consumes phosphorus as an essential element needed for cellular constituents such as phospholipids, nucleotides, and nucleic acids (Powell *et al.*, 2008). However, high accumulation of organic phosphates in the pond sludge reduces algal growth (Ramadan and Pounce, 2004a), thereby reducing uptake of phosphorus.

Effluent phosphorus concentrations were higher than upstream concentrations of phosphorus in river Oda, resulting in higher concentrations downstream ($P<0.05$).

Furthermore, effluent concentration of phosphorus did not meet the EPA recommended standard of 2 mg/L. Moreover, no significant changes were observed among upstream and downstream values after discharge of effluent ($P<0.05$). This can adversely affect the Oda River

But, the increase in nutrients in the Oda River (N, P) cannot be solely attributed to poor efficiency of the treatment plant. According to Erni *et al.*, (2010) the higher values obtained upstream can be attributed to failing sanitation.

5.4.3 Potassium

Concentrations of potassium for wastewater and water samples were very high. However, pond performance for nitrogen and phosphorus was appreciable compared with that of potassium. Effluent concentrations for potassium for the months of June and July were higher than influent (Tables 4.2 and 4.3). However, significant changes were observed among mean values ($P<0.05$). This is an indication that potassium concentrations were increased in the ponds due to accumulation of sludge. Therefore percentage removals for these months were negative.

Moreover, effluent potassium concentrations were higher than upstream concentrations of potassium in river Oda. But these concentrations increase downstream the Oda River because of the high effluent concentrations in wastewater ($P<0.05$). Thus, even though the EPA has no effluent standards for potassium, damaging effects on receiving water body is of greater concern.

5.5 Chemical oxygen demand

Chemical oxygen demand (COD) is a measure of the amount of oxygen required to chemically oxidize reduced minerals and organic matter (Galbrand *et al.,* 2008). Higher levels of COD were observed in influent but were reduced, with a mean percentage removal efficiency of 80.9 (± 17.85) % in effluent. This explains the significant difference between influent and effluent values of BOD as a result of pond performance ($P<0.005$). Upstream concentrations for COD in River Oda were low. However, concentrations increased downstream 100 m after the point of discharge. This increase in COD can be attributed to an increase in the addition of both organic and inorganic substance from the wastewater treatment plant. Again, no significant difference was recorded among upstream and downstream parameters of COD ($P>0.05$). This suggests a decrease in the assimilative ability of the river to reduce contaminants that increase the COD below the effluent discharge point. Continuous discharge of effluent might impact the receiving water body to some extent and this may have negative effects on the quality of the freshwater and subsequently cause harm to the aquatic life especially fish, downstream. This accession is in line with work done by Morrison *et al*., (2001). Furthermore, COD effluent concentrations were above the recommend EPA standard of 250 mg/L (fig.4.8) despite high percentage removal efficiency. This is due to very low algal populations to cause chemical activity that will reduce the COD.

5.6 Biochemical oxygen demand

Biological Oxygen demand (BOD) is the measure of the oxygen required by

microorganisms whilst breaking down organic matter. It is well reported that wastewater effluents with high concentrations of BOD can cause depletion of natural oxygen resources, which may lead to the development of septic conditions (Hodgson, 2007). The BOD removal and the consequent quality of the effluent depend on the amount of oxygen present, retention time and temperature of the ponds (Hodgson, 2007). The BOD removal efficiency was 91.84% for May and 89.68 % for June which is high and comparable to other waste stabilization ponds which give BOD removal efficiencies greater than 70 % (Arceivala, 1981). On the other hand July recorded a percentage removal of 19.68 %. This may be due to increased rainfall, resulting in the reduction of retention time, temperature and oxygen concentration of the pond. Again, maximum influent capacity will also affect the ability of a facility to perform efficiently (Yuronich, 2000). Wastewater treatment plants are designed to handle a limited amount of waste.If the plant is at or near its maximum design capacity, heavy flow due to large amounts of precipitation may present a potentially hazardous situation (Yuronich, 2000). Nevertheless, changes in influent and effluent concentrations of BOD showed a significant difference ($P<0.05$) indicating a high pond performance for reductions in

BOD.

Furthermore, septage or faecal sludge from septic tanks contains a large variety of substances and also different concentration of constituents. This will depend on how often the tanks are cleaned. Since regulation of faecal sludge into the treatment plant is lacking, producing a consistent effluent quality that will meet the required standard is always questionable. However, Abis (2002) reported that the removal of algal and other solids from effluent increases the BOD removal range between 89.7 to 99.7 % with a mean of

97.3 %. Increases in TDS and TSS values affect the BOD removal efficiency and is clearly depicted in the results obtained. This observation is supported by work done by Mara *et al*., (1992a, b) and Bradley (1983) who stated that an increase between 50 and 90 % of the BOD in a tertiary lagoon effluent is due to the algal content. Again, Mayo (1996) reported that an increase of 160 to 240 % in effluent BOD from a site in Tanzania was due to suspended solids, particularly biomass.

Effluent concentrations of wastewater for BOD did not meet the required EPA standard of 50 mg/L (fig.4.9). This caused downstream concentrations below the point of discharge to increase. Reductions in concentrations of BOD are minimal within the Oda River after the point of discharge. Therefore effluents that meet acceptable limits should be ensured.

5.7 pH

Effect of pH on wastewater treatment plant performance is evident. Hodgson and Larmee (1998) reported that coliforms were reduced to zero (0) in the final effluent when the pH was above 10.7. Again, an elevated pH can cause phosphates to precipitate by complexation with metal ions such as calcium, magnesium, and iron present in the wastewater causing a reduction in phosphorus concentrations (Powell *et al.*, 2008). However, exceedances of pH above the recommended guidelines have been associated with many adverse effects. One of the most significant impacts of pH in water bodies is the effect that it has on the solubility and thus the bioavailability of other substances such as iron, manganese and ammonia (Galbrand *et al.*, 2008).

From the results, effluent wastewater was in a pH range of 6.46 to 6.48. This range meets the recommended EPA guideline of 6 to 9, indicating that pH of the effluent wastewater

will not have any adverse effect on River Oda. Changes in the values of pH from influent to effluent were significant showing a gradual increase in the values of pH through the treatment process ($P<0.05$).

Moreover, upstream values of pH were in a range of 5.73 to 5.8. But these values reduced to a range of 5.5 to 5.6 with a significant difference of $P<0.05$. A clear indication that natural attenuation processes within the receiving water influences pH.

5.8 Microorganisms

Oxidation ponds remove a significant percentage (90–99 %) of indicator and pathogenic bacteria (Bitton, 2005). The factors that influence coliform removal in both primary facultative and maturation ponds include retention time, temperature, pH and light intensity (Hodgson, 2007). Even though high influent counts of microorganisms were recorded for both faecal and total coliforms, changes in effluent values were evident. For the month of May, a high percentage removal of 97.39 and 96.61 was recorded for total and faecal coliforms respectively (table 4.1).

Campos *et al.*, 2002, reported that sedimentation of microorganisms is the main mechanism responsible for microbial cell death in aerobic ponds. Helminth eggs sediment due to their weight, whereas viruses adhere onto solids which subsequently sediment. Aerobic conditions created by organic matter decomposition play a major role. Furthermore, retention time increases the die-off (Rangeby *et al.*, 1996).

In facultative ponds the death and removal of indicator microorganisms is a very complex process. Factors such as sedimentation, solar radiation, high pH, low CO_2 levels, high

concentrations of dissolved O_2, algal toxins, presence of predators and retention time can affect the microorganism removal rate. Sensitizing molecules present in the water and inside the microorganisms produce toxic oxygen derivatives such as free radicals and superoxide ions by photochemical reactions. This type of O_2 derivate, combined with high pH, will affect cell membranes causing cell death (Campos *et al.*, 2002). These conditions aided in a high percentage removal in May.

For the month of June and July, rainfall was high and this affected the retention time as well as the pH required for maximum cell death. A reduction in pond performance for faecal coliforms was observed with a percentage of 61.25 and 46.93 for June and July respectively. Again, coliform die-off decreases with an increase in BOD and pond depth (Saqqar and Pescod, 1992). Therefore, high BOD levels obtained from the results is a contributing factor to low coliform die-off.

Effluent counts were high above the recommended EPA guideline of 400 MPN/100 ml of effluent wastewater for all the months. Even though upstream concentrations of microorganisms were high in River Oda due to non-point sources of pollution and surface run-off, effluent discharges further increased these concentrations downstream. Over flooding of some of the ponds during times of heavy rainfall is another contributing factor. Standard requirements must be adhered to, in preventing pollution of the Oda River.

CHAPTER SIX

6.0 CONCLUSIONS AND RECOMMENDATIONS

6.1 CONCLUSIONS

The study showed that waste stabilization pond system is an efficient method for the cotreatment of landfill leachate and faecal sludge. However, to achieve the desired effluent quality, operational requirements such as overloading of ponds, frequent emptying of septic tanks, desludging of ponds and maintenance of microbe at their optimal operational levels must be strictly adhered to.

The findings further revealed that even though the Dompoase wastewater treatment plant, achieved higher percentage removals for most parameters the treatment plant exhibited effluent qualities that met acceptable standards in only pH. This suggest that effluents fell short of standard requirements that are critical to the provision of clean and safe water such as organic matter (BOD and COD); solids (TSS and TDS); nutrient (N, K and P); heavy metals (Fe and Pb) and microorganisms (total and faecal coliforms).

Furthermore, downstream concentrations of most parameters were higher than upstream values after the effluent discharge point in River Oda. Therefore, it can be concluded that wastewater effluents from the Dompoase treatment plant has an effect on the physical and microbial qualities of the Oda River. Even though natural assimilative ability of the river accounts for a reduction in contaminant numbers, adverse effects can be experienced in a

long term. Since some contaminants can adhere to sediments, bioaccummulate and biomanify.

6.2 RECOMMENDATIONS

1. It is recommended that wastewater effluents and water sources be routinely monitored by operators of the treatment plant to ensure that strict adherence to effluent discharge standards is met.
2. The Environmental Protection Agency (EPA) should enforce its rules in an effort to protect water resources.
3. Management should ensure desludging of anaerobic ponds to increase pond performance. Pond capacity should not be exceeded, in an effort to achieve the best form of efficiency required from the plant and to improve maintenance.
4. Further, research incorporating the dry season should be considered. This will indicate the appropriate time at which pond performance is at the maximum, to help make better designs for future ponds.

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APPENDIX

Appendix A

A1: Sampling values for May

Sample location	TDS (mg/L)	TSS (mg/L)	Fe (mg/L)	COD (mg/L)	BOD (mg/L)	N (mg/L)	P (mg/L)	K (mg/L)	Pb (mg/L)	pH	Total coliform	Faecal coliform
Influent (S1)	16460	35700	10	73880	2650	36950	8150	247000	4.63	5.92	3.9×10^{11}	2.3×10^{11}
	16900	35300	14	73320	2500	37200	5380	245000	4.6	5.91	3.0×10^{11}	2.4×10^{11}
Effluent (S2)	5540	1140	10	3380	270	10000	6690	270100	0.33	6.46	9.0×10^{9}	9.3×10^{8}
	5360	1100	8	3180	150	9970	6520	199200	0.39	6.45	9.0×10^{9}	9.0×10^{8}
Upstream (S3)	89.5	7	2	96	30	8650	47	1300	0.03	5.88	2.4×10^{11}	1.1×10^{9}
	86.9	8	1.4	40	10	7840	37	1600	0.02	5.72	2.3×10^{11}	6.0×10^{9}
Downstream (S4)	317	27	1.4	280	70	8560	120	11000	0.02	5.64	7.0×10^{14}	4.0×10^{10}
	318	26	1.38	368	85	8560	110	11700	0.04	5.68	1.1×10^{14}	1.5×10^{10}

A2: Sampling values for June

Sample location	TDS (mg/L)	TSS (mg/L)	Fe (mg/L)	COD (mg/L)	BOD (mg/L)	N (mg/L)	P (mg/L)	K (mg/L)	Pb (mg/L)	pH	Total coliform	Faecal coliform
Influent (S1)	10300	12700	29	29200	1890	32090	10000	146800	0.98	5.8	2.4×10^{14}	2.3×10^{13}
	10500	12650	41	30000	1890	32080	10000	127000	0.91	5.77	2.4×10^{14}	2.4×10^{13}
Effluent (S2)	6950	1250	9.5	4100	240	18380	5500	239700	0.68	6.47	9.3×10^{13}	2.3×10^{12}
	6200	1150	8	4300	150	18700	6500	208900	0.98	6.48	9.3×10^{13}	2.4×10^{12}
Upstream (S3)	59.3	17	1.14	160	5.4	5620	500	1400	0.5	5.9	1.6×10^{9}	4.0×10^{8}
	54.4	13	1.29	136	3	5790	500	1200	0.35	5.43	1.6×10^{9}	4.0×10^{8}
Downstream (S4)	81.4	11	1.23	72	14.4	9270	1000	2690	0.66	5.52	1.6×10^{13}	2.6×10^{9}
	76.5	39	1.34	224	12.6	9270	1000	3400	0.5	5.47	1.6×10^{13}	9.0×10^{9}

A3: Sampling values for July

Sample location	TDS (mg/L)	TSS (mg/L)	Fe (mg/L)	COD (mg/L)	BOD (mg/L)	N (mg/L)	P (mg/L)	K (mg/L)	Pb (mg/L)	pH	Total coliform	Faecal coliform
Influent (S1)	2940	4800	1.6	6260	1540	34600	8380	185200	2.81	5.86	3.9×10^{9}	2.3×10^{9}
	2960	4810	2	6260	1420	34560	8390	186300	2.76	5.84	3.0×10^{10}	2.4×10^{9}
Effluent (S2)	1483	948	1.6	2450	1220	14270	6200	229440	0.51	6.45	9.0×10^{9}	2.3×10^{8}
	1492	960	1.7	2440	1240	14250	6410	229460	0.69	6.47	9.0×10^{9}	2.4×10^{8}
Upstream (S3)	57.1	6	0.84	16	4.8	6940	310	1400	0.27	5.89	2.4×10^{10}	4.0×10^{8}
	55.9	8	0.67	14	4.5	7000	310	1400	0.19	5.57	2.3×10^{10}	4.0×10^{8}
Downstream (S4)	77	11	1.26	31	6.6	8910	720	7160	0.34	5.58	7.0×10^{10}	2.6×10^{9}
	76.9	13	1.54	33	7.2	8910	710	7140	0.27	5.57	1.1×10^{10}	9.0×10^{9}

APPENDIX C: Water Quality Guidelines for Discharge into Water Bodies or Water Courses

PARAMETER/DESCRIPTION	Maximum Permissible Levels New Facilities	Maximum Target (Permissible) Level (Existing Facilities)
*p*H	6 - 9(in the range of)	6 - 9(in the range of)
Temperature*	<3˚C above ambient	<3˚C above ambient
Colour (TCU)	20	100
Oil and grease (mg/L)	20	20
Oil	No visible floating oil	No visible floating oil
BOD (mg/L)**	50	200
COD (mg/L)**	250	1000
Total Dissolved Solids (mg/L)	1000	1000
Total Suspended Solids (mg/L)	50	50
Turbidity (NTU)**	75	75
Conductivity (µS/cm)**	1500	1500
Total coliforms (MPN/ 100 ml)	400	400
E.coli (MPN/ 100 ml)	10	10
Ammonia as N (mg/L)**	1	10
Nitrate (mg/L)**	75	100
Flouride (mg/L)**	10	20
Phenol (mg/L)	1	1
Sulphide (mg/L)	1.5	1.5
Total phosphorus (mg/L)	2	10
Total cyanide (mg/L)	1	1
Free Cyanide (mg/L)	0.2	0.2
Soluble Arsenic (mg/L)	0.1	0.1
Cadmium (mg/L)	<0.1	<0.1

PARAMETER/DESCRIPTION	Maximum Permissible Levels New Facilities	Maximum Target (Permissible) Level (Existing Facilities)
Chromium (mg/L)	0.1	0.1
Chromium (+6) (mg/L)	0.1	0.1
Total Chromium (mg/L)	0.5	0.5
Copper (mg/L)	2.5	2.5
Lead (mg/L)	0.1	0.1
Nickel (mg/L)	0.5	0.5
Selenium (mg/L)	1	1
Zinc (mg/L)	5	5
Mercury (mg/L)	0.005	0.005
Silver (mg/L)	0.1	0.1
Tin (mg/L)	5	5
Aluminium (mg/L)	5	5
Antimony (mg/L)	1.5	1.5
Benzo (a) pyrene (mg/L)	0.05	0.05
Chloride (mg/L)**	250	2500
Sulphate (mg/L)**	300	3000
Chlorine (mg/L) (Total residual chlorine)	250	250
Trichloroethylene (µg/l)	7	50
Total Hardness (mg/L)**	500	2000
Barium (mg/L)	0.7	0.7
PCBs (Trichloronebezene (µg/l)	20	20
Manganese (Mn) (mg/L)**	0.1	2.5
Perchloroethylene (µg/l)	40	40
Benzen (µg/l)	10	50
Influent raw water/Upstream raw water	IR + 15% raw water parameter	IR + 15% raw water parameter
Total (all) metals (mg/L)	10	

Toxic metals (mg/L)***	5	

*****Applicable at the edge of the zone where initial mixing and dilution takes place. Where the zone is not defined, 100 meters from the point of discharge shall be used.

** Values for existing facilities differ markedly from new facilities.

*** Toxic metals means antimony, arsenic, beryllium, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, thallium, vanadium, zinc, etc.

Appendix D

D1: One Way Analysis of Variance for TDS

Group Name	N	Missing	Mean	StdDev	SEM
Influent TDS	6	0	10010.000	6168.184	2518.150
Effluent TDS	6	0	4504.167	2404.725	981.725
Upstream TDS	6	0	67.183	16.379	6.687
Downstream TDS	6	0	157.800	123.716	50.507

Source of Variation	DF	SS	MS	F	P
Between Groups	3	397238376.725	132412792.242	12.080	<0.001
Residual	20	219223840.942	10961192.047		
Total	23	616462217.666			

The differences in the mean values among the treatment groups are greater than would be expected by chance; there is a statistically significant difference (P = <0.001).

Power of performed test with alpha = 0.050: 0.998

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):
Overall significance level = 0.05

Comparisons for factor:

Comparison	Diff of Means	t	Unadjusted P	Critical Level
Influent TDS vs. Upstream TDS	9942.817	5.202	<0.001	0.009
Influent TDS vs. Downstream T	9852.200	5.154	<0.001	0.010
Influent TDS vs. Effluent TDS	5505.833	2.880	0.009	0.013
Effluent TDS vs. Upstream TDS	4436.983	2.321	0.031	0.017
Effluent TDS vs. Downstream T	4346.367	2.274	0.034	0.025
Downstream T vs. Upstream TDS	90.617	0.0474	0.963	0.050

Comparison	Significant?
Influent TDS vs. Upstream TDS	Yes
Influent TDS vs. Downstream T	Yes
Influent TDS vs. Effluent TDS	Yes
Effluent TDS vs. Upstream TDS	No
Effluent TDS vs. Downstream T	No
Downstream T vs. Upstream TDS	No

D2: One Way Analysis of Variance for TSS

Group Name	N	Missing	Mean	StdDev	SEM
Influent TSS	6	0	17660.000	14450.709	5899.477

Effluent TSS	6	0	1091.333	117.348	47.907
Upstream TSS	6	0	9.833	4.262	1.740
Downstream TSS	6	0	21.167	11.392	4.651

Source of Variation	**DF**	**SS**	**MS**	**F**	**P**
Between Groups	3	1349238850.833	449746283.611	8.614	<0.001
Residual	20	1044184593.000	52209229.650		
Total	23	2393423443.833			

The differences in the mean values among the treatment groups are greater than would be expected by chance; there is a statistically significant difference (P = <0.001).

Power of performed test with alpha = 0.050: 0.971

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):
Overall significance level = 0.05

Comparisons for factor:

Comparison	**Diff of Means**	**t**	**Unadjusted P**	**Critical Level**
Influent TSS vs. Upstream TSS	17650.167	4.231	<0.001	0.009
Influent TSS vs. Downstream T	17638.833	4.228	<0.001	0.010
Influent TSS vs. Effluent TSS	16568.667	3.972	<0.001	0.013
Effluent TSS vs. Upstream TSS	1081.500	0.259	0.798	0.017
Effluent TSS vs. Downstream T	1070.167	0.257	0.800	0.025
Downstream T vs. Upstream TSS	11.333	0.00272	0.998	0.050

Comparison	**Significant?**
Influent TSS vs. Upstream TSS	Yes
Influent TSS vs. Downstream T	Yes
Influent TSS vs. Effluent TSS	Yes
Effluent TSS vs. Upstream TSS	No
Effluent TSS vs. Downstream T	No Downstream
T vs. Upstream TSS	No

D3: One Way Analysis of Variance

Group Name	N	Missing	Mean	StdDev	SEM
Influent Fe	6	0	16.267	15.729	6.421
Effluent Fe	6	0	6.467	3.816	1.558
Upstream Fe	6	0	50.263	76.086	31.062
Downstream Fe	6	0	1.223	0.469	0.191

Source of Variation	DF	SS	MS	F	P
Between Groups	3	8743.016	2914.339	1.926	0.158
Residual	20	30256.233	1512.812		
Total	23	38999.249			

The differences in the mean values among the treatment groups are not great enough to exclude the possibility that the difference is due to random sampling variability; there is not a statistically significant difference (P = 0.158).

Power of performed test with alpha = 0.050: 0.211

The power of the performed test (0.211) is below the desired power of 0.800. Less than desired power indicates you are less likely to detect a difference when one actually exists. Negative results should be interpreted cautiously.

D4: One Way Analysis of Variance

Group Name	N	Missing	Mean	StdDev	SEM
Influent Pb	6	0	2.782	1.642	0.670
Effluent Pb	6	0	0.597	0.238	0.0973
Upstream Pb	6	0	0.305	0.252	0.103
Downstream Pb	6	0	0.227	0.187	0.0763

Source of Variation	DF	SS	MS	F	P
Between Groups	3	26.496	8.832	12.396	<0.001
Residual	20	14.249	0.712		
Total	23	40.746			

The differences in the mean values among the treatment groups are greater than would be expected by chance; there is a statistically significant difference (P = <0.001).

Power of performed test with alpha = 0.050: 0.998

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):
Overall significance level = 0.05

Comparisons for factor:

Comparison	Diff of Means	t	Unadjusted P	Critical Level	Sig.
Influent Pb vs. Downstream Pb	2.555	5.243	<0.001	0.009	Yes
Influent Pb vs. Upstream Pb	2.477	5.082	<0.001	0.010	Yes
Influent Pb vs. Effluent Pb	2.185	4.484	<0.001	0.013	Yes
Effluent Pb vs. Downstream Pb	0.370	0.759	0.457	0.017	No
Effluent Pb vs. Upstream Pb	0.292	0.598	0.556	0.025	No
Upstream Pb vs. Downstream Pb	0.0783	0.161	0.874	0.050	No

D5: One Way Analysis of Variance

Group Name	N	Missing	Mean	StdDev	SEM
Influent COD	6	0	36553.333	31060.183	12680.267
Effluent COD	6	0	950.000	1159.051	473.181
Upstream COD	6	0	61.000	68.150	27.822
Downstream COD	6	0	168.000	143.108	58.424

Source of Variation	DF	SS	MS	F	P
Between Groups	3	5886890268.500	1962296756.167	8.125	<0.001
Residual	20	4830517555.333	241525877.767		
Total	23	10717407823.833			

The differences in the mean values among the treatment groups are greater than would be expected by chance; there is a statistically significant difference (P = <0.001).

Power of performed test with alpha = 0.050: 0.960

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):
Overall significance level = 0.05

Comparisons for factor:

Comparison	Diff of Means	t	Unadjusted P	C.level
Influent COD vs. Upstream COD	36492.333	4.067	<0.001	0.009
Influent COD vs. Downstream COD	36385.333	4.055	<0.001	0.010
Influent COD vs. Effluent COD	35603.333	3.968	<0.001	0.013
Effluent COD vs. Upstream COD	889.000	0.0991	0.922	0.017
Effluent COD vs. Downstream COD	782.000	0.0872	0.931	0.025
Downstream C vs. Upstream COD	107.000	0.0119	0.991	0.050

Comparison	Significant?
Influent COD vs. Upstream COD	Yes
Influent COD vs. Downstream COD	Yes
Influent COD vs. Effluent COD	Yes
Effluent COD vs. Upstream COD	No
Effluent COD vs. Downstream COD	No
Downstream C vs. Upstream COD	No

SEM

D6: One Way Analysis of Variance

Group Name	N	Missing	Mean	StdDev	
Influent BOD	6	0	1998.333	478.682	195.421
Effluent BOD	6	0	545.000	532.795	217.512
Upstream BOD	6	0	9.617	10.262	4.190
Downstream BOD	6	0	32.633	35.205	14.372

Source of Variation	DF	SS	MS	F	P
Between Groups	3	15721249.395	5240416.465	40.754	<0.001
Residual	20	2571756.815	128587.841		
Total	23	18293006.210			

The differences in the mean values among the treatment groups are greater than would be expected by chance; there is a statistically significant difference (P = <0.001).

Power of performed test with alpha = 0.050: 1.000

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):
Overall significance level = 0.05

Comparisons for factor:

Comparison	Diff of Means	t	Unadjusted P	Critical Level	Sig
Influent BOD vs. Upstream BOD	1988.717	9.606	<0.001	0.009	Yes
Influent BOD vs. Downstream B	1965.700	9.495	<0.001	0.010	Yes
Influent BOD vs. Effluent BOD	1453.333	7.020	<0.001	0.013	Yes
Effluent BOD vs. Upstream BOD	535.383	2.586	0.018	0.017	No
Effluent BOD vs. Downstream B	512.367	2.475	0.022	0.025	Yes
Downstream B vs. Upstream BOD	23.017	0.111	0.913	0.050	No

One Way Analysis of Variance

Group Name	N	Missing	SEM
	6		
	6		
	6		
	6		

D7: **for N**

		Mean	StdDev	
Influent N	0	34580.000	5938.781	2424.497
Effluent N	0	14261.667	6837.814	2791.526
Upstream N	0	8913.333	317.532	129.632
Downstream N	0	6973.333	1298.409	530.073

Source of Variation	DF	SS	MS	F	P
Between Groups	3	2878842845.833	959614281.944	45.799	<0.001
Residual	20	419057550.000	20952877.500		
Total	23	3297900395.833			

The differences in the mean values among the treatment groups are greater than would be expected by chance; there is a statistically significant difference (P = <0.001).

Power of performed test with alpha = 0.050: 1.000

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):
Overall significance level = 0.05

Comparisons for factor:

Comparison	Diff of Means	t	Unadjusted P	Critical Level	Sig.
Influent N vs. Downstream N	27606.667	10.446	<0.001	0.009	Yes
Influent N vs. Upstream N	25666.667	9.712	<0.001	0.010	Yes
Influent N vs. Effluent N	20318.333	7.688	<0.001	0.013	Yes

One Way Analysis of Variance

Group Name	N	Missing			SEM	
	6					
	6					
	6					
	6					
Effluent N vs. Downstream N		7288.333	2.758	0.012	0.017	Yes
Effluent N vs. Upstream N		5348.333	2.024	0.057	0.025	No
Upstream N vs. Downstream N		1940.000	0.734	0.471	0.050	No

One Way Analysis of Variance

Group Name	N	Missing	SEM
	6		
	6		
	6		
	6		

D8: **for P**

		Mean	StdDev	
Influent P	0	8383.333	1747.440	713.389
Effluent P	0	6303.333	439.211	179.307
Upstream P	0	308.333	172.211	70.305
Downstream P	0	585.667	440.283	179.745

Source of Variation	DF	SS	MS	F	P
Between Groups	3	298566422.000	99522140.667	114.724	<0.001
Residual	20	17349795.333	867489.767		
Total	23	315916217.333			

The differences in the mean values among the treatment groups are greater than would be expected by chance; there is a statistically significant difference (P = <0.001).

Power of performed test with alpha = 0.050: 1.000

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):
Overall significance level = 0.05

Comparisons for factor:

Comparison	Diff of Means	t	Unadjusted P	Critical Level	Sig.
Influent P vs. Upstream P	8075.000	15.017	<0.001	0.009	Yes
Influent P vs. Downstream P	7797.667	14.501	<0.001	0.010	Yes
Effluent P vs. Upstream P	5995.000	11.149	<0.001	0.013	Yes
Effluent P vs. Downstream P	5717.667	10.633	<0.001	0.017	Yes
Influent P vs. Effluent P	2080.000	3.868	<0.001	0.025	Yes
Downstream P vs. Upstream P	277.333	0.516	0.612	0.050	No

One Way Analysis of Variance

Group Name	N	Missing	SEM
	6		
	6		
	6		
	6		

D9: **for K**

		Mean	StdDev	
Influent K	0	189550.000	50289.472	20530.591
Effluent K	0	229466.667	29631.920	12097.181
Upstream K	0	1383.333	132.916	54.263
Downstream K	0	7181.667	3734.115	1524.446

Source of Variation	DF	SS	MS	F	P
Between Groups	3	257586738845.833	85862246281.944	100.393	<0.001
Residual	20	17105214750.000	855260737.500		
Total	23	274691953595.833			

The differences in the mean values among the treatment groups are greater than would be expected by chance; there is a statistically significant difference (P = <0.001).

Power of performed test with alpha = 0.050: 1.000

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):
Overall significance level = 0.05

Comparisons for factor:

Comparison	Diff of Means	t	Unadjusted P	Critical Level	Sig.
Effluent K vs. Upstream K	228083.333	13.508	<0.001	0.009	Yes
Effluent K vs. Downstream K	222285.000	13.165	<0.001	0.010	Yes
Influent K vs. Upstream K	188166.667	11.144	<0.001	0.013	Yes

One Way Analysis of Variance

Group Name	N	Missing			SEM	
	6					
	6					
	6					
	6					
Influent K vs. Downstream K		182368.333	10.801	<0.001	0.017	Yes
Effluent K vs. Influent K		39916.667	2.364	0.028	0.025	No
Downstream K vs. Upstream K		5798.333	0.343	0.735	0.050	No

D10: One Way Analysis of Variance for pH

Group Name	N	Missing	Mean	StdDev	SEM
Influent pH	6	0	5.850	0.0593	0.0242
Effluent pH	6	0	6.463	0.0121	0.00494
Upstream pH	6	0	5.732	0.196	0.0801
Downstream pH	6	0	5.577	0.0766	0.0313

Source of Variation	DF	SS	MS	F	P
Between Groups	3	2.716	0.905	75.325	<0.001
Residual	20	0.240	0.0120		
Total	23	2.956			

The differences in the mean values among the treatment groups are greater than would be expected by chance; there is a statistically significant difference (P = <0.001).

Power of performed test with alpha = 0.050: 1.000

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):
Overall significance level = 0.05

Comparisons for factor:

Comparison	Diff of Means	t	Unadjusted P	Critical Level
Effluent PH vs. Downstream PH	0.887	14.009	<0.001	0.009
Effluent PH vs. Upstream PH	0.732	11.560	<0.001	0.010
Effluent PH vs. Influent PH	0.613	9.691	<0.001	0.013
Influent PH vs. Downstream PH	0.273	4.319	<0.001	0.017
Upstream PH vs. Downstream PH	0.155	2.449	0.024	0.025
Influent PH vs. Upstream PH	0.118	1.870	0.076	0.050

Comparison	Significant?
Effluent PH vs. Downstream PH	Yes
Effluent PH vs. Upstream PH	Yes
Effluent PH vs. Influent PH	Yes
Influent PH vs. Downstream PH	Yes

Upstream PH vs. Downstream PH	Yes
Influent PH vs. Upstream PH	No

D11: One Way Analysis of Variance for total coliforms

Group Name	N	Missing	Mean	StdDev
Influent total coliform	6	0	8.014E+013	1.238E+014
Effluent total coliform	6	0	3.101E+013	4.802E+013
Upstream total coliform	6	0	1.403E+014	2.773E+014
Downstream total coliform	6	0	8.670E+010	1.153E+011

Group Name	SEM
Influent total coliform	5.055E+013
Effluent total coliform	1.960E+013
Upstream total coliform	1.132E+014
Downstream total coliform	47084597623.144

Source of Variation	DF	SS	MS	F	P
Between Groups	3	6.755E+028	2.252E+028	0.953	0.434
Residual	20	4.726E+029	2.363E+028		
Total	23	5.402E+029			

The differences in the mean values among the treatment groups are not great enough to exclude the possibility that the difference is due to random sampling variability; there is not a statistically significant difference (P = 0.434).

Power of performed test with alpha = 0.050: 0.049

The power of the performed test (0.049) is below the desired power of 0.800. Less than desired power indicates you are less likely to detect a difference when one actually exists. Negative results should be interpreted cautiously.

D12: One Way Analysis of Variance for faecal coliforms

Group Name	N	Missing	Mean	StdDev
Influent faecal coliform	6	0	7.912E+012	1.208E+013
Effluent faecal coliform	6	0	783716666666.667	1.214E+012
Upstream faecal coliform	6	0	5050000000	3455864580.680
Downstream faecal coliform	6	0	9433333333.333	16073041612.174

Group Name	SEM
Influent faecal coliform	4.931E+012
Effluent faecal coliform	495470513879.933
Upstream faecal coliform	1410850807.137
Downstream faecal coliform	6561791760.724

Source of Variation	DF	SS	MS	F	P
Between Groups	3	2.655E+026	8.850E+025	2.402	0.098
Residual	20	7.368E+026	3.684E+025		
Total	23	1.002E+027			

The differences in the mean values among the treatment groups are not great enough to exclude the possibility that the difference is due to random sampling variability; there is not a statistically significant difference (P = 0.098).

Power of performed test with alpha = 0.050: 0.307

The power of the performed test (0.307) is below the desired power of 0.800. Less than desired power indicates you are less likely to detect a difference when one actually exists. Negative results should be interpreted cautiously.

Appendix E

Locality map for study area