PERFORMANCE EVALUATION OF PILOT-SCALE CONSTRUCTED WETLANDS FOR THE TREATMENT OF DOMESTIC WASTEWATER IN ADDIS ABABA, ETHIOPIA

By:

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Declaration

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I declare that the above thesis is my own work and that all the sources that I have used or quoted have been indicated and acknowledged by means of complete references.

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ABSTRACT

An experimental study was carried out to evaluate the performance of pilot scale constructed wetlands for the treatment of domestic wastewater in Addis Ababa, Ethiopia. Three parallel sets of constructed wetlands; consisting of one Horizontal Flow (HF), one Vertical Flow (VF) and one hybrid of HF and VF-constructed in series were built in Addis Ababa. The wetland systems had identical wetland fill media and macrophytes but with different wastewater flow types. The total surface area of the wetland systems was 72 m^2 /24 m^2 for each/ and designed to treat 3.15 m^3 of domestic wastewater per day. Triplicate grab samples were taken from the influent and effluents every 15 days for one year and analyzed within 24 hours. Temperature, pH, DO and EC were measured onsite and the nutrient content of macrophytes was determined twice during the monitoring period.

During the first 12 months monitoring period, the average removal efficiencies of the HFCW, VFCW and hybrid CW were: BOD (89.1%, 92.2% and 93.4%), COD (80.6%, 82.1% and 84.0%), TSS (89.1, 83.8% and 84.7%), NH_4^+ (58.6%, 66.2% and 65.4%), NO_3^- (64.0%, 71.5% and 73.5%), TN (49.1%, 54.9% and 58.7%), PO_4^{3-} (45.4%, 50.3% and 48.4%), TP (58.0%, 51.7% and 54.4%) and FC (98.6%, 96.6% and 96.5%), respectively. The hybrid system showed relatively higher removal efficiencies for most pollutants. Again, the wetland systems showed relatively higher percent reduction during the dry seasons /from Dec - May/. The areal removal rate constants of BOD_5, TN, PO_4^{3-} and TP were higher than the literature values while the values of COD and TSS were lower compared to the literature values.

Concerning the nutrient content of the wetland plant, the average TN contents of the below-ground and above-ground plant part were 1.56% and 2.27% for the HFCW, 1.75% and 2.74% for the VFCW and 1.80% and 2.63% for the hybrid system, respectively. Meanwhile, the average TP contents of the below-ground and above-ground plant part were 0.139% and 0.064% for the HFCW, 0.167% and 0.067% for the VFCW and 0.115% and 0.065% for the hybrid systems, respectively.

In general, the results showed that properly designed constructed wetland systems could be used as effective wastewater treatment method in Ethiopia.

Key words: Wastewater treatment, constructed wetlands, media/substrate, macrophytes, *Cyprus papyrus*, influent, horizontal flow, vertical flow, hybrid constructed wetland, nutrient uptake, mass loading rate, areal removal rate constant, pollutant removal, seasonal performance.

DEDICATION

I dedicate this dissertation to my late mother whom I lost in the middle of my study. I wish that she would have been around at the end of this study.

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Table of Contents

Contents Page

List of Figures

List of Tables

Tables Pages*F*

List of Abbreviations

AAU	Addis Ababa University
BOD_5	5-day Biological Oxygen Demand
BE PLC	Beles Engineering Private Limited Company
COD	Chemical Oxygen damand
CSA	Central Statistics Agency
CWs	Constructed Wetlands
DO	Dissolved Oxygen
ECA	Economic Commission for Africa
EEC	European Economic Community
EEPA	Ethiopian Environmental Protection Authority
EFCCC	Environment, Forest and Climate Change Commission
US EPA	United States Environmental Protection Agency
EU	European Union
FAO	Food and Agriculture Organization
FDRE	Federal Democratic Republic of Ethiopia
FDREPCC	Federal Democratic Republic of Ethiopia Population Census Commission
FWSCW	Free Water Surface Constructed Wetlands
GEPD	Georgia Environmental Protection Division
HF	Horizontal Flow
HFCW	Horizontal Flow Constructed Wetlands
HSSF	Horizontal Subsurface Flow
ICCIWMEE	International Conference on Chemical, Integrated Waste Management and Environmental Engineering
IWA	International Water Association
MDHSS	Missouri Department of Health and Senior Services
MoE	Ministry of Education
N	Nitrogen
NMA	National Meteorological Agency of Ethiopia
Ortho-P	Ortho-phosphate
P	Phosphorous

PE	Population Equivalents
RCS	Ramsar Convention Secretariat
RE	Removal Efficiency
SSs	Suspended Solids
SSFB	Subsurface Flow Bed
SSFCW	Subsurface Flow Constructed Wetland
TN	Total Nitrogen
TP	Total Phosphorous
TSS	Total Suspended Solids
UN	United Nations
UNESCO	United Nations Education Science and Culture Organization
UNHSC	University of New Hampshire Stormwater Center
NEIWPCC	New England Interstate Water Pollution Control Commission
UNISA	University of South Africa
USDC	United States Department of Commerce
USCS	United States Commercial Service
Vs	Versus
VF	Vertical Flow
VFCW	Verticla Flow Constructed Wetland
VSSF	Vertical Subsurface Flow
WWTP	Wastewater Treatment Plant
WHO	World Health Organization
WI	Wetland International

CHAPTER ONE

INTRODUCTION

1.1 Study background

For many years, conventional wastewater treatment methods, which demand high cost and considerable energy, were popular to treat domestic wastewater all over the world (UN-Habitat, 2008b; Trivedy and Siddharth, 2010). However, these treatment methods necessitate the search for other alternatives that can fill the gaps. This prompts a revisit to the biological treatment technologies that include constructed wetlands for the treatment of various types of wastewater since the 1960s (Trivedy and Siddharth, 2010; Kurniadie, 2011; Vergeles *et al*, 2015).

Constructed wetlands (CWs) system is one of the natural treatment systems that rely on natural processes utilizing systems composed of plants, support medium and microorganisms, which are cost-effective and present good operational steadiness (Sarmento *et al*, 2012). According to UN-Habitat (2008b), constructed wetlands can be defined as:

> ‘a natural, low-cost, eco-technological biological wastewater treatment technology designed to mimic processes found in natural wetland ecosystems, which is now standing as the potential alternative or supplementary systems for the treatment of wastewater’

They can be a viable alternative for developing countries with tropical climates due to ease of construction and operation (Sarmento *et al*, 2012). They are simple, low cost, long-lived and an eco-friendly wastewater treatment system that can be widely applied in developing countries (Azni *et al*, 2010; Zapater-Pereyra *et al*, 2013; Prashant *et al*, 2013; Amaral *et al*, 2001; Vergeles *et al*, 2015). This is because of the application of natural systems instead of energy demanding conventional technology for wastewater treatment (Shutes, 2001). Properly designed and operated constructed wetlands could also be used for secondary and tertiary wastewater treatment (Kurkusuz *et al*, 2004).

Kathe Seidel (1960s) as cited in Vymazal (2009) recognized CWs as an appropriate alternative treatment technology in Germany and they were initially used for treating municipal or domestic wastewater. But at present, their potential of treating wastewater from various sources leads to the rapidly expansion and applicability of the technology in almost every country (Vymazal, 2009; Vymazal, 2014a). Now, wetlands are being used for the treatment of virtually all types of wastewaters including municipal, industrial, agricultural, acid mining drainage, animal, and leachate as well as storm water (Liu *et al*. 2008; Kadlec and Wallace, 2009; Azni *et al.* 2010; Vymazal, 2010). Although natural systems offer potentially cheaper and low-energy treatment alternatives to treat wastewaters, CWs are able to attract more attention in recent works as they are easily regulated (Vymazal, 2009; Trivedy and Siddharth, 2010; Siti *et al*, 2011).

The extent of CWs applicability ranges from small scale level to serve single household or institution to large-scale centralized municipal systems. There are different configurations, scales, and designs and the efforts to optimize the performance of CWs (Azni *et al*, 2010; Chazarenc *et al*, 2015). From the previous trend, the free water surface system is extensive in North America while the subsurface flow system is predominantly used in Europe (Azni *et al*, 2010). As different types of constructed wetlands are employed for primary, secondary or tertiary treatment level, the right design or configuration should be chosen and applied based on the treatment objective (Van, 2010; Zhang *et al*, 2016). The right design and configuration and proper operation help to overcome the influences that may occur by different environmental factors (UN-Habitat, 2008b; Kadlec and Wallace, 2009; Bai *et al*, 2016).

Meanwhile, a number of studies on the performance of pilot-scale or full-scale constructed wetlands were carried out in many African countries such as South Africa (Schulz *et al*, 2003), Egypt (Abou-Elela and Hellal, 2012), Tunisia (Abidi *et al*, 2009), and Nigeria (Erakhrumen and Agbontanor, 2007). Similar studies were conducted in the Eastern African countries; e.g. in Uganda (Kyambadde *et al*, 2004), Kenya (Kelvin and Tole, 2011; Odinga *et al*, 2011) and Tanzania (Mashauri *et al*, 2000; Mairi *et al*, 2012).

In Ethiopia, the full-scale application of this technology for wastewater treatment is limited to few institutions and some studies related to the performance of CWs have been conducted (Birhanu, 2007; Asaye, 2009; Tadesse, 2010; Kenatu 2011).

1.2 Statement of the Problem

Wastewater is a by product of water used for different purposes. Large volume of domestic wastewater is produced as a result of increasing water consumption due to urbanization and industrialization and improvement of people's living standards (Wallace, 2004; Mekala *et al*, 2008). The major constituents of typical wastewater pollutants include: oxygen-demanding materials, pathogens, nutrients, suspended solids, sediments, grease/oil, heavy metals and other hazardous materials (Manahan, 2000; Kayombo *et al*, 2003). According to UNESCO (2012), it is estimated that domestic wastewater generated in urban areas of Asia-Pacific region lies in the range between 150 and 250 million m^3 per day, which can have the potential to create environmental pollution. Therefore, to protect the environment, the pollutants would have been removed from the wastewater before discharging into the nearby water bodies.

However, more than 80% of the wastewater does not receive any level of treatment and is directly discharged into water bodies or leaches into the surface of the Earth. The situation is worse in developing countries (Biswas, 2010; UNESCO, 2012). Under these circumstances, water bodies such as lakes, rivers and oceans are being highly polluted due to the practice of discharging untreated or partially treated wastewater. Consequently, a grave water crisis may result in most developing countries if the trend is not reversed in the future. The discharge of contaminants with untreated domestic wastewater from urban areas into water bodies is also causing public health concerns and environmental pollution (Biswas, 2010; Kurniadie, 2011; Saravanan *et al*, 2011).

Due to the public health and environmental problems associated with discharging of untreated wastewater and need to reclaim and reuse the enormous amount of wastewater, appropriate treatment technologies must be evaluated and applied (Mekala *et al,* 2008; IWA, 2018; MDHSS, 2018). Wastewater reuse for irrigation is common worldwide and it accounts for 10% of the total irrigated surface (FAO, 2010). It is used by households practicing agriculture in and around the urban areas to improve agricultural productivity and food security in low income countries (Jimenez, 2006; Scheierling *et al*, 2011). The livelihoods of low income households in urban areas often depend on wastewater reuse (Sheierling *et al*, 2011). Countries like Jordan, Chile and Israel are well known for their successful achievement in wastewater reclamation and the need for continuous improvement over many years is learned from the experiences of these countries in order to achieve safe wastewater irrigation (Mara *et al*, 2007; Murray and Ray, 2010; Wu *et al*, 2015b). There are 3,300 water reclamation sites identified worldwide (FAO, 2010).

The developed countries in Europe, North America, and Australia apply different treatment technologies to treat various types of wastewater at the desired level before being discharged to the environment (Mekala *et al,* 2008). For instance, the United States alone treat nearly 150 billion liters of wastewater per day by using more than 15,000 wastewater treatment plants (Charles and Ian, 2009).

But many developing nations of Africa and Asia are not able to treat their wastewater to the desired levels which is attributed mainly to lack of adequate funds, high treatment costs of the conventional treatment systems, lack of skilled manpower and fast increase in wastewater volumes (Nhapi and Gijzen, 2005; Schertenleib, 2005; Mohammed and Eibably, 2016). As a result, they are discharging untreated domestic wastewater into the nearest water bodies or leaching into the environment despite of the negative impact on the environment. For instance, the amount of wastewater discharged to the environment in China was about 65 billion cubic meters by the year 2011. The problem is expected to get worse unless appropriate measures are taken to control and treat effluents (UN-Habitat, 2008a, USDC, 2013).

In Africa, South Africa has shown remarkable development in wastewater treatment that brings the country at the front line in the continent. As of 2008, the country had about 900 wastewater treatment plants with capacity of treating 5,000,000 to 7,000,000 m^3 of wastewater each day (UN-Habitat, 2008a). The gap in wastewater management between many African countries and the developed countries such as North America, Europe, Australia and Japan with regard to wastewater management is implausible. The later allocate and utilize huge funds to protect water bodies and the environment from pollution (Abdel-Halim and Rosenwinkel, 2005; UN-Habitat, 2008a).

The problem of wastewater management in Ethiopia is severe. Wastewater stabilization ponds have been employed in Addis Ababa since the end of 1970s; however, the proportion of domestic wastewater treated by wastewater stabilization ponds in Addis Ababa is not greater than 9.8% of the total volume of domestic wastewater generated in the city. In recent times, expansion and rehabilitation works for treatment plants and sewerage lines have been carried out and also new wastewater treatment plants have been deployed. The newly constructed treatment plants (some of them are operational) include Upflow Anaerobic Sludge Blanket (UASB) and Membrane Bioreactor (MBR) which are expensive and energy intensive methods. The MBR treatment plants are constructed and operated to treat wastewater generated from condominiums. Under the circumstances, both untreated domestic and industrial wastewater is discharged into the nearby rivers, which are used as water sources for multiple purposes including domestic uses and irrigation. Smallholder farmers in and around the city use the untreated wastewater for crop production (AAWSA, 2002; Alebel *et al*, 2009; BE PLC, 2014; World Bank, 2015). Therefore, this study was done to evaluate the performance of constructed wetlands as alternative technology for the treatment of domestic wastewater under the Ethiopian climatic conditions.

1.3 Study justification/Rationale

Constructed wetlands have been used as alternative method of wastewater treatment for more than fifty years. Most applications have been employed to treat domestic or municipal wastewater. But in recent times, they are successfully applied to different types of wastewaters (Vymazal, 2014a). Although constructed wetlands can be successfully used to remove pollutants from various types of wastewater, the applications of constructed wetland systems in Ethiopia is limited to few institutions for the treatment of domestic wastewater.

The primary purpose of this study was to evaluate the removal efficiency of three pilot scale subsurface flow constructed wetland systems in Addis Ababa in treating domestic wastewater. It was also intended to compare the performance which can be affected by the rate of microbial activity, macrophytes growth and other removal mechanisms. The removal mechanisms of pollutants from wastewater are usually influenced by seasonal or temperature differences.

The available literature regarding the impacts of seasonal variations on the performance of CWs in treating different wastewaters are not consistent (Siti *et al*, 2011). However, a number of studies are in favor of the thought that seasonal removal of constituents from wastewater using CWs can change with location and targeted constituent, so an initial pilot-scale study could be beneficial prior to construction of a full-scale system to estimate the removal rate coefficients of targeted constituents (Alley *et al*, 2013). Seasonal variations seem to affect the performance of CWs technology even if the performance is not consistent for all wastewater parameters. Lower effluent concentrations were observed during the warm period especially for certain pollutants. The performances improved as wastewater temperature rises (Song *et al*, 2006; Prochaska *et al*, 2007; Rousseau *et al*, 2008; Mustafa *et al*, 2009; Mietto *et al*, 2015; Ramprasad and Philip, 2016).

In addition, constructed wetlands have diverse design configurations and their application varies from location to location. For instance, from the previous trend, the free water surface system is extensive in North America; while the subsurface flow system is predominantly used in Europe, Australia and South Africa. The subsurface flow systems are the most common treatment methods in nearly every country of the world at this time. Because of their low maintenance costs and simple operational management subsurface flow CW systems are good choices for wastewater treatment (UN-Habitat, 2008b; Kadlec and Wallace, 2009; Azni *et al*, 2010; Vymazal and Brezinova, 2015; Wijaya *et al*, 2016).

In this study the subsurface flow CWs with horizontal flow, vertical flow and hybrid of horizontal and vertical flow types were evaluated to determine their effectiveness in removing pollutants from domestic wastewater. Therefore, the result can be highly relevant to use the system at full scale level in areas of similar climatic conditions.

1.4 Research Questions

In consideration of attaining the stated objectives of the study, the following are the research questions on which the study focused on:

I. How efficient could constructed wetlands be in treating domestic wastewater in the country?

II. Could the different wastewater flow types have effects on constructed wetlands applied for the treatment of domestic wastewater under similar environmental conditions?

III. Would the performance of constructed wetland systems be different in different seasons of the year?

IV. How will the values of areal removal rate constants of wastewater pollutants compare with literature values?

V. Will the different wetland plant parts have accumulated different nutrient (nitrogen and phosphorous) content?

1.5 Objectives of the Study

1.5.1 General Objective

The general objective of the study was to evaluate the performance of horizontal, vertical and hybrid of both horizontal and vertical subsurface flow pilot scale constructed wetlands for the treatment of domestic wastewater based on several physico-chemical parameters and also to determine the TN and TP content of the above-ground and below-ground part of the macrophytes (*Cyperus papyrus*).

1.5.2 Specific objectives

The specific objectives of the study were:

- To evaluate and compare the wastewater treatment performance of subsurface horizontal flow, vertical flow and hybrid of the horizontal and vertical flow constructed wetland systems in different seasons;
- To evaluate the association between removal rate of wastewater pollutants and loading rate;
- To determine the removal rate constants of wastewater pollutants; and
- To determine the nutrient (N and P) content of the below-ground and above-ground parts of macrophytes (*C. papyrus*) used in the pilot scale constructed wetland systems.

1.6 Ethical Considerations

Prior to conducting the actual study, ethical consent was obtained from the two pertinent offices. Addis Ababa Environmental Protection Authority (AA EPA) confirmed that the study will not have any negative impact on the environment, while Addis Ababa Water and Sewerage Authority (AAWSA) showed its willingness to carryout the study at Kotebe wastewater treatment plant.

Accordingly, the approval of ethics clearance was obtained from the ethics review committee of the College of Agriculture and Environmental Science, UNISA, after submitting all the required documentation (Ref. Nr.: 2013/CAES/162).

CHAPTER TWO

LITERATURE REVIEW

2.1 Wetlands

Natural wetlands commonly represent the areas which are found between land and water bodies. They are intermediary areas linking the two surfaces and have been recognized as natural resource throughout the history of mankind. The submissive nature of wetlands arising from different factors makes it almost impossible to give one general definition. So that, they can be defined in several ways depending on those factors which include: personal perspective, existing water and plant condition, landscape position/geographic setting, and wetland diversity and function (Thomas and William, 2001; Scholz, 2006; Kadlec and Wallace, 2009). Although there are a variety of ways to define the wetland system, the most widely agreed definition was formulated by the International Union for the Conservation of Nature and Natural Resources (IUCN) in the Ramsar Convention, in 1980. According to the convention, wetlands are defined as:

> 'areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters.'
>
> *Article 1.1*

Wetlands are well known for giving magnificent services as biological filters to protect water resources of both surface and groundwater. Natural wetlands have acted as ecological buffer all the time to protect the environment. But conducting researches and the advancement of using wetland treatment technology for treating various wastewaters is a development started in the early 1950s, in Germany. In the United States, researches on wetlands began in the late 1960s and extended in scope during the 1970s. Following this, treating wastewater using wetlands emerged as an appropriate alternative technology globally (Thomas and William, 2001).

Wetlands have distinctive characteristics which make them different from major ecosystems plainly known (Kadlec and Wallace, 2009). As nearly all forms of biological productivity highly depend on the amount of water available, water is the most important factor affecting the wetland environment and the associated forms of life (RCS, 2013). Ramsar Convention (2013) pointed out that:

'*Wetlands are among the world's most productive environments. They are cradles of biological diversity, providing the water and primary productivity upon which countless species of plants and animals depend for survival. They support high concentrations of birds, mammals, reptiles, amphibians, fish and invertebrate species. Wetlands are also known to be important storehouses of plant genetic material.*'

(Ramsar convention, 2013: p.8).

In view of this, wetlands are recognized for giving protection for the environment especially water bodies by removing pollutants in discharged wastewater ranging from rainfall runoff to strong wastewater such as community sewage over many years (Robert, 2004). They have been employed as convenient wastewater discharge sites for as long as sewage has been collected, at least 100 years in some locations (Kadlec and Wallace, 2009), so that they have been receiving increasing attention as effective alternatives for wastewater treatment (Charles and Ian, 2009). In this regard, bacterial metabolism and physical sedimentation are the two major processes acknowledged for the performance of wetlands in treating wastewater (Kadlec and Wallace, 2009; Zhai et al, 2016; Szklarek *et al*, 2018).

Furthermore, immense and highly indispensable services for human well-being and poverty reduction are offered by wetland ecosystems (Kent, 2001; WRI, 2005; Vymazal, 2010). Some of the most important ecosystem services include: aquatic and wildlife habitat (Kent, 1994; Vymazal, 2010; Si *et al*, 2014), educational and scientific venues, flood flow alteration (Vymazal, 2010), groundwater recharge, elemental transformation, particle retention and sources of raw materials, recreation, and soil stabilization (Kent, 2001; WRI, 2005).

In Ethiopia, it is estimated that the wetlands covered 2% of the country's land surface (Afework, 2005). But much of these resources are exposed to exploitation and signs of wetland degradation have become out of control across the country. This is mainly due to the absence of clear policy frameworks except the scattered efforts made by different sectors. In the mean time, scholars have been contributing a lot to create awareness on the benefits and status of wetlands since the beginning of the 1990s. There have been different views among scholars regarding wetland policies and strategies; some arguing that the issue of wetlands has been addressed in the general framework of the existing policies and development strategies and others have stressed the need for a standalone wetland development policy. Currently, the Federal Environment, Forest and Climate Change Commission (EFCCC) seems to have been working in this line for realizing the signing of the Ramsar Convention and approval of the draft wetland policy (Tadesse and Solomon, 2014).

2.2 Constructed Wetlands as an Attractive Technology for WWT; an Overview

Constructed wetlands (CWs) are artificial or engineered wetlands designed and constructed to reproduce and improve the processes of wastewater treatment which take place in natural wetlands (Mara, 2003). They are basins having low depth, filled more often with sand or gravel as filter media and planted with aquatic plants. The wastewater to be treated is conveyed through the inlet pipes into the basins and flows through the substrate (media) or over the surface, and finally the effluent is discharged out of the system through the outlet pipes that are designed to keep the depth of the wastewater in the basin. Azni *et al*, (2010) pointed out that CWs can be created from existing marshlands or built at any land with limited alternative uses. The most important sections of a constructed wetland include; basin, substrate, vegetation, liner, inlet/outlet pipes (UN-Habitat, 2008b). Constructed wetlands have been successfully used as treatment systems for domestic wastewater effluent, from single-residence (Gikas and Tsihrintzis, 2010) wetlands to large municipal wastewater treatment facilities (Kent, 2001).

Seidel (1953) as cited in Kadlec and Wallace (2009) was the person to begin experimenting with aquatic plants to improve water quality, and she is acknowledged for the development of constructed wetlands in Europe. CWs can function without any electromechanical devices such as aerators to supply oxygen and therefore they are usually called natural treatment systems. Wetland plants are the main sources of oxygen for the oxidation of organic pollutants by the heterotrophic bacteria (Trivedy and Siddharth, 2010).

Rousseau *et al* (2008) pointed out that if constructed wetlands are designed and maintained carefully, they can yield an effluent which meets reuse requirements and concurrently provide some opportunities to recycle nutrients and to accommodate wildlife. Advances in the innovative design and operation of CWs have greatly increased contaminant removal efficiencies, thereby improving the sustainable applications of these treatment systems. For treatment wetlands, optimization and innovation in design, operation, and maintenance, could help to make this treatment technology much more attractive (Wu *et al*, 2015a).

Based on energy synthesis, CWs are found to be less energy-intensive with relative low cost Ecological Waste Removal Efficiency (EWRE). They are also environmental friendly and cheaper than Cyclic Activated Sludge Systems (CASS) either in construction or during operation and maintenance. All these imply that CWs still gain some advantages over the conventional treatment plants, especially in the developing countries (Zhou *et al*, 2007; Vymazal, 2010). Amaral *et al* (2013) revealed that CWs gained attention of the present-day for alternative wastewater treatment method since they are simple to operate and maintain and require low resource consumptions.

Although, developing countries which are in need of such effective treatment methods do not give adequate attention (Kivaisi, 2001), the application of CWs as wastewater treatment technology has been rapidly growing in many parts of the world starting from 1985. This is because, for one thing, the construction of the technology can be made by using human labor and locally available materials, providing low cost and low maintenance but high technology alternative for developing countries (Hench *et al*, 2003; Konnerup *et al*, 2009; Tao *et al*, 2014). Secondly, wetland systems are able to achieve high treatment performance as a result of complex hydrological and biological processes which take place in removing contaminants even though they are mechanically simple (Merlin *et al*, 2002; Kadlec and Wallace, 2009).

In addition to their potential as effective treatment technology, CWs can also provide other ecosystem services such as conservation of biodiversity. Therefore, in view of sustainability, they are considered as a good sanitation solution with great potential of protecting natural resources and the environment. Solar energy is the only energy source on which the system depends for pollutant removal processes and hence this ensures its sustainability (Merlin and Lisollo, 2010). However, institutional limitations, relatively large land requirement and less public awareness are expected to be the major ongoing challenges that possibly avert the wider application of CWs (Liu *et al*, 2008; Tao *et al*, 2014; Langergraber *et al*, 2014).

The performance of CWs in tropical areas, where most of the developing countries are located (Kivaisi, 2001) is satisfactory and it has more or less steady performance throughout the year. There is rapid ecological succession with tropical biodiversity mix in those areas where the temperature is high, so that, treatment of domestic wastewater can be achieved to an acceptable standard by natural processes. Greater treatment efficiencies as a result of the complex natural processes can be achieved by CWs than widely used waste stabilization ponds (Kelvin and Tole, 2011).

Zurita *et al* (2009) described that the type of flow in CW system is the most important factor that affects the rate of removing wastewater contaminants. Higher removal rate of almost all pollutants except NO_3^-, TN and TSS can be achieved by using Vertical Flow Constructed Wetlands (VFCWs). Because there is better aeration/oxygen supply/ that enables to enhance wastewater nitrification in VFCWs.

2.3 Advantages of Constructed Wetlands

Nowadays, CWs are able to replace conventional wastewater treatment methods, and treat not only domestic wastewater but also other wastewaters from various sources. They are designed and constructed to take the advantage that virtually all processes occur naturally in a more controlled environment (Haberl *et al*, 2003; Azni, 2010). Hence, CWs have a number of advantages including: they have a long life time with minimum maintenance requirement, they can treat broad spectrum of contaminants in wastewater simultaneously to an acceptable level, and they can play a major role in increasing biodiversity and then to provide ecosystem services in themselves and the surrounding environment (Habler *et al*, 2003; Siracusa and Rosa, 2006; Azni *et al*, 2010; Zhang *et al*, 2014).

According to (Kayombo *et al*, 2003; Russo, 2008; Massoud *et al*, 2009; Mthembu *et al*, 2013; Qasaimeh *et al*, 2015), the advantages of CWs are summarized as follows;

- can often be less expensive to build than other treatment options,
- can be built and operated simply,
- utilize natural processes,
- their operation and maintenance expenses (energy and supplies) are low,
- are able to tolerate fluctuations in flow and pollutant concentration,
- are able to treat wastewaters with very different constituents and concentration,
- are characterized by a high process stability (buffering capacity),
- are characterized by low excess sludge production,
- Facilitate water reuse and recycling, and
- Provide other indirect benefits such as green space, wildlife habitats and recreational and educational areas.

2.4 Types of Constructed Wetlands

Constructed wetlands are designed and constructed in various ways according to the theoretical basis of earlier studies to take the advantage of many complex processes that occur naturally by microbial community, vegetation, substrate and wastewater (USEPA, 2000; UN-Habitat, 2008b; Vymazal, 2009). The present wetlands are designed to employ specific characteristics for improved treatment capacity

(Kadlec and Wallace, 2009). Haberl (1999) as cited in UN-Habitat (2008b) described that CWs have diverse design configurations and the basis for their classification comprise of:

- Life form of the dominating macrophytes (free-floating, emergent, submerged),
- Flow pattern in the wetland systems (FWS flow; SS flow: horizontal and vertical),
- Type of configurations of wetland cells (hybrid systems, one-stage, multi-stage systems),
- Type of wastewater to be treated,
- Treatment level of wastewater (primary, secondary or tertiary),
- Type of pretreatment,
- Influent and effluent structures,
- Type of substrate (gravel, soil, sand, etc.), and
- Type of loading (continuous or intermittent loading).

But the type of water flow is the most important factor among others that are used as the basis for the classification of constructed wetlands (Vymazal, 2009). Although constructed wetlands have a lot of characteristics in common, based on the water flow type, they are in practice classified into two general types: Free Water Surface (FWS) wetlands (also called surface flow (SF) wetlands) and Subsurface Flow (SSF) wetlands (also known as Vegetated Submerged Bed (VSB) systems) (Kayombo *et al.* 2003; Russo, 2008; Kadlec and Wallace, 2009).

The water flows above the ground and it is exposed to the atmosphere in free water systems. However, in subsurface flow systems, the water usually flows through a bed made with a porous media such as sand, gravel or aggregates and hence, the water is not exposed to the atmosphere (USEPA, 2000; Kayombo *et al.* 2003; Russo, 2008; Kadlec and Wallace, 2009). Concerning their application distribution, Cole (1998) as cited in Thomas and William (2001) described that subsurface flow wetlands are the common systems in Europe for treating domestic wastewater while the free water systems types are widely used in North America.

The selection of either of the system during the design phase should focus on the required quality of the effluent coming out of the wetland system. For instance, in areas where there is enough space and when the removal of merely SS and BOD is needed, SFSs with one-unit SSF type can be adequate. But multistage or combined systems having both horizontal flow (HF) types and vertical flow (VF) types should be used in areas where there is more stringent discharge limits. On the other hand, phosphorous adsorption capacity and hydraulic conductivity are the two major criteria used in the selection of the medium used in SSF constructed wetlands. As multistage systems contain more treatment beds, they are more expensive than one-unit system. But they are still notably cheaper than the conventional treatment technologies (Brix, 1995).

2.4.1 Free Water Surface systems

Free water surface (FWS) constructed wetlands are areas of open water systems and there is no any barrier between the water surface and the surrounding atmosphere. The appearance and function of these systems closely resemble natural wetlands in that: they have open-water which is in contact with the atmosphere, emergent vegetation, varying water depths, and other typical wetland features. They are designed in such a way that the processes taking place in natural wetlands can be used effectively in the treatment of wastewater (US EPA, 2000; Charles and Ian, 2009).

The common features of FWS include; a basin with impermeable bottom, inlet structures, open-water areas with vegetation, and outlet structures. The size, shape, and complexity of the system design usually attributed to the characteristics of the site than the perceived criteria prior to its construction (US EPA, 2000; Thomas and William, 2001; Thullen *et al*, 2002; Charles and Ian, 2009; Azni *et al*, 2010). Free water surface wetlands are distinguished by a relatively shallow layer (usually ranges between 0.3 to 0.4m) of surface water flowing over the impermeable soil with low flow velocity. Macrophytes play a major role in regulating the flow of water in a long basin which helps to maintain the plug-flow condition in the system (Beharrell, 2004; Polprasert, 2004).

These types of wetlands are chosen in many parts of the world since they are cheaper than other treatment methods including SSF constructed wetlands, and beyond this the values of wetland habitat and reuse opportunities are highly associated with FWS systems (US EPA, 1993; Beharrell, 2004; Han *et al*, 2014; Mohammadpour *et al*, 2014). A wide-range of wildlife including insects, amphibians, birds, reptiles and mammals are greatly attracted by FWS wetlands (Kadlec and Knight, 1996). FWS systems are extensively used in North America typically for treating large flow municipal wastewater. The predominant wetland type is FWS and they are applied at larger sizes even if wetlands of smaller sizes are used in some localities (Polprasert, 2004; Beharrell, 2004).

Every part of the world including the coldest northern hemisphere can be suitable site to employ FWS wetlands for wastewater treatment. However, ice formation can occur in extreme cold conditions and the ice can cover the water surface, which results in decreasing of the transfer of oxygen from the atmosphere to the wetland system. This situation has a negative impact on some removal processes. For example the rate of conversion of nitrogen decreases in this kind of situation. Conversely, TSS removal efficiency increases under ice comparing to summer season. In general, the removal efficiency is higher in warm seasons, so that collecting and storing wastewater at cold seasons and treating during the warm seasons is the best approach (Kadlec and Wallace, 2009).

Flocculation and sedimentation are the two most important processes in FWS wetlands which are known for their key role in removing pollutants in wastewater while the wastewater flows through stands of wetland vegetation. Sometimes, the physical pollutant removal processes in some FWS systems can be complemented by aerobic bio-oxidation process (US EPA, 2000). So, the retention time of the wastewater to be treated in a FSW constructed wetland is the most important factor in evaluating the effectiveness of the purification capacity of the system (Su *et al*, 2009).

In FWS system, there is high probability of human exposure to disease causing micro-organisms present in wastewater. As a result, the system is not commonly considered as a good alternative for secondary treatment (US EPA, 2000; Kadlec and Wallace, 2009). But effluents coming out of other treatment technologies such as lagoons, trickling filters, and activated sludge can be further polished by FWS systems (Kadlec and Wallace, 2009).

As reported by Siti *et al* (2011) the SFCW system are not reliable treatment method for treating wastewater with high ammonium concentration particularly in situations where there is short retention time. Therefore, to increase oxygen concentration within the system, and then to enhance the capacity of ammonia removal, intermittent or batch flow type to alternating basin should be encouraged. The presence of macrophytes in FWS systems improves the performance in treating concentrated ammonia in secondary-treated effluent (Thullen *et al,* 2002; Trias *et al*, 2012).

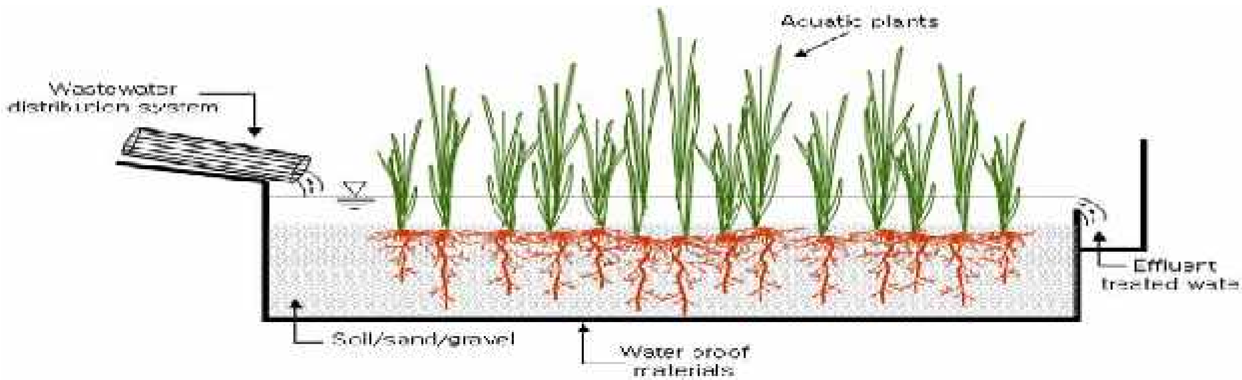

Figure 2.1: Free water surface flow constructed wetland (Russo, 2008)

The FWS system is further divided into three sub-groups based on the various vegetation types dominantly grown in the wetland system (US EPA, 2000; Russo, 2008).

- A floating macrophyte system – these systems make use both of floating species that are rooted in the substrate (e.g. *Nymphae* spp *Nuphar* spp. (waterlilies), *Potamog etonnatans* (pondweed), *Hydrocotyle vulgaris* (pennyworth)) and species which are free floating on water surface (e.g. *Eichhornia crassipes* (water hyacinth), *Lemna* spp., *Spirodella* spp. (duckweed));
- A submerged macrophyte system – the plants used in these systems have their photosynthetic tissue entirely submerged with the flowers being exposed to the atmosphere. Two types of submerged aquatics are usually recognized: the elodeid type (*e.g*. *Elodea* spp., *Myriophyllum aquaticum* (parrot feather), *Ceratophyllum* spp.) and the isoetid (rosette) type (*e.g. Isoetes*, *Littorella*, *Lobelia)*;
- A rooted emergent macrophyte system – these systems use plants which are the dominating form of life in the natural wetlands. Plants grow at well above the water level, producing aerial stems and an extensive root and rhizome system. These comprise species like the *Phragmites australis* (common reed), *Thypha* spp. (cattails), *Scirpus* spp. (bulrushes), *Iris* spp. (blue and yellow flags) *Juncus* spp. (rush), *Saggitaria latifolia* (duck potato), *Phalaris arundinocea* (reed canary grass), *Carex* spp. (Sedges), *Zizania aquatica* (wild rice), *Eleocharis* spp. (Spikerushes) and *Glyceria* spp. (mannagrasses).

2.4.2 Subsurface Flow (SSF) systems

The other wetlands type is subsurface flow (SSF) system, which is designed to create subsurface water flow and to keep the wastewater to be treated below the surface of the bed. These systems are supportive to avoid bad odor and other nuisance condition that may cause disease incidence as a result of pathogenic microorganisms found in wastewater. The system commonly uses gravel, aggregates, sand or soil as a porous media, on which the macrophytes are rooted and grown (Kayombo *et al*, 2003; Vymazal, 2010). SSF systems are usually planted with emergent wetland vegetation (Vymazal, 2010).

In SSF wetland, the bed is filled with a porous media and the depth of the media is about 0.3 to 0.6m deep and the bottom is covered with geosynthetic impermeable layer to prevent underground infiltration or seepage. The bed should have 1% slope at the bottom to avoid water flows over the bed. In the meantime, perforated pipes are buried at the inlet zone to keep maximum flow through the treatment zone and the effluent is then collected by the outlet pipes buried at the base of the media, which is about 0.3-0.6 m below the surface of the bed. Then, the wastewater to be treated flows from the top inlet to the bottom outlet direction under the surface of the wetland bed (Kadlec and Wallace, 2009; Nelson *et al*, 2009; Azni *et al*, 2010; Pedescoll *et al*, 2012).

The application of SSF systems is extensive in Europe, Australia, South Africa, and nearly every country of the world and they are the most common treatment plants at this time. One of the peculiar features of SSF systems, unlike the free water surface flow system, is that insect vectors do not get any opportunity to breed in the system as there is no contact between the water column and the surrounding atmosphere. So, the likelihood of incidence of public health problem which is associated with the application of SSF constructed wetlands is extremely low. As a result, they can offer better option for primary wastewater treatment (Kayombo *et al*, 2003; Robert, 2004; Kadlec and Wallace, 2009).

Compared to the free water surface system, the performance of subsurface flow constructed wetlands for nutrient removal is higher. The removal efficiency of constructed wetlands is dependent on aerobic and anaerobic condition within the wetland cells and the differences of water flow in the system (Li *et al*, 2008). Yang *et al* (2014) pointed out that even slightly polluted drinking water source could be effectively treated by applying SSF systems and consequently the quality of drinking water source could be improved which in turn reduces the burden on drinking water treatment.

Basically, SSF constructed wetlands are further grouped into two categories based on the direction of water flow. These are vertical up or down flow (VF) and horizontal flow (HF) types (Vymazal *et al*, 1998). In addition, employing of the hybrid/combined system of the two wetland types to effectively exploit the advantages of each type is becoming common practice in many areas now a day (Merlin, Pajean and Lisollo, 2002).

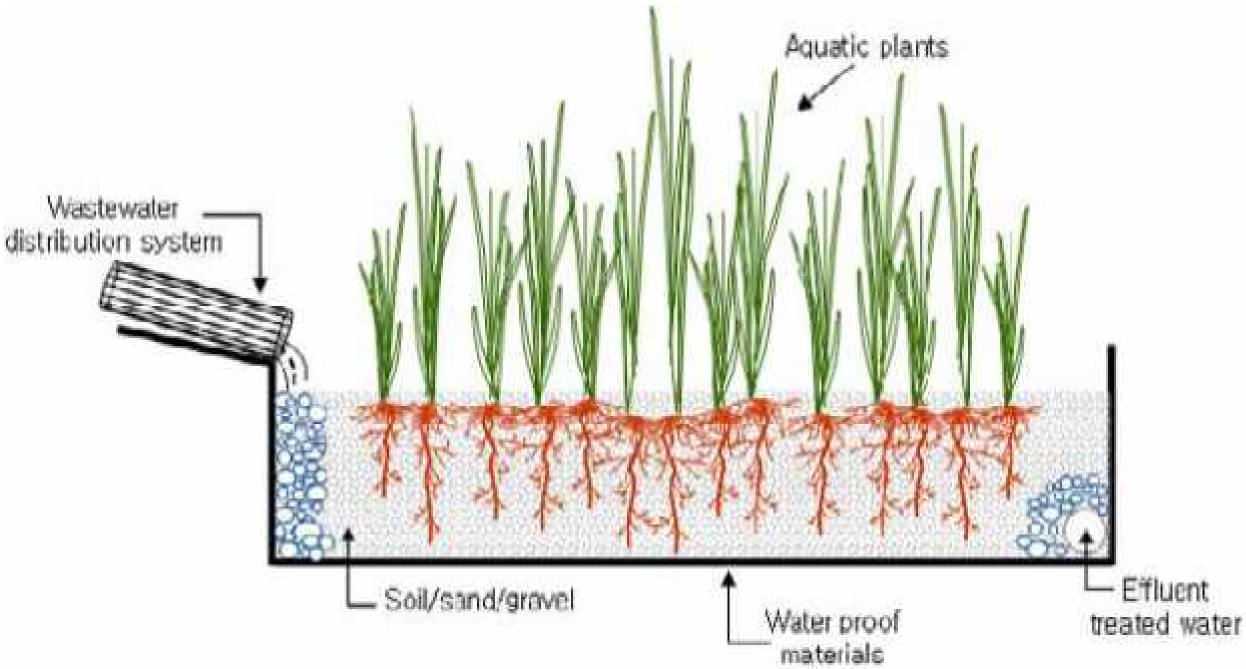

Figure 2.2: Subsurface flow constructed wetland (Russo, 2008; p. 329)

2.4.2.1 Horizontal Subsurface Flow Constructed Wetlands (HSSF CWs)

In horizontal subsurface flow system, the influent from the inlet zone flows horizontally thorough the porous media below the surface of the bed until it gets to the outlet zone. At the time of its flow, the wastewater undergoes in different processes that are taken place in aerobic, anoxic, and anaerobic zones of the bed. The aerobic zones are those sites around roots and rhizomes where oxygen is released into the media of the constructed wetland. The effluent or the treated wastewater is then collected in the outlet zone before leaving the system (Vymazal *et al*, 1998; Russo, 2008).

Toscano *et al* (2015) described that the performance of HSSF constructed wetland in reducing major physical, chemical, and microbiological concentration of contaminants in municipal wastewater is very high. He also emphasized the active role of vegetations in the removal processes of pollutants in wastewater treatment using wetland system. HSSF constructed wetlands showed high and steady removal performance over many years of operation for organic pollutants like BOD5, COD, TSS and oil and grease with satisfactory effluent quality for being discharged into the environment. But their performance is poor in terms of contaminants in wastewater such as phosphorous, ammonium nitrogen and organic matter due to oxygen deficiency (Haberl *et al*, 1995; Vymazal, 2005; Naz *et al*, 2009; Cakir *et al*, 2015; Costa *et al*, 2015; Albalawneh *et al*, 2016). Despite this, (Costa *et al*, 2015) pointed out that good P removal efficiency (70%) and fair N removal efficiency (40%) can be achieved by using HSSF constructed wetlands.

2.4.2.2 Vertical Subsurface Flow Constructed Wetlands (VSSF CWs)

In vertical SSF CWs, whether it is ascending or descending, vertical direction of flow through the media is established by using various designs of wastewater feeding or collection mechanisms. This can be achieved by applying the wastewater to be treated into the cell intermittently or by burying inlet pipes into the bed at certain depths. This kind of wetland system is known as "infiltration wetlands" since wastewater infiltration occurs through the medium (Vymazal *et al*, 1998).

The performance of VF bed is significantly better than HF for the removal of BOD_5, COD, Kjeldahl-nitrogen and ammonia-nitrogen. This mainly occurred as the unsaturated flow condition in VF bed presents more oxygen for the oxidation-reduction potential to take place in VSSF wetland (Pandey *et al*, 2013). Kurniadie (2011) mentioned that the effective removal of organic matter, nutrients and pathogenic bacteria can be achieved by the proper application of VSSF CWs planted with macrophytes and the reduction of concentration of COD, NO_3-N, PO_4-P, and total coliforms in the final effluent is very low.

In a wetland system, carbon degradation is carried out chiefly by bacteria while fungi have minor role. In VSSF CWs, as more than 80% of the growth or multiplication of microbes is taken place inside the first 10 cm of the filter media, the depth of the filter media should not be less than 10 cm to maintain steady performance and filtration process (Tietz *et al*, 2008). Additionally, the design with two-stage VSSF CW can enhance the performance in treating wastewater (Xie *et al*, 2011; Langergraber *et al*, 2014).

In VF constructed wetlands, DO levels increase initially and then decrease vertically from top to bottom. There is a positive correlation between the levels of DO and the biofilm mass, showing the presence of other sources of oxygen supply in addition to the oxygen in the influent, particularly in the upper part of the wetland bed. This incident supports the assumption that the major oxygen source for VFCWs is atmospheric reoxygenation, and of course the contribution of atmospheric reoxygenation in the process of domestic wastewater treatment is more than 99.9% of the total oxygen supply to the VFCWs. The upper part of VFCWs usually encompasses 0 - 10cm below wastewater distribution system is supplied by just about 50% of atmospheric reoxygenation (Ye *et al*, 2012).

2.4.2.3 Hybrid Constructed Wetlands

A number of treatment processes which take place in CWs can be more effective in removing pollutants in wastewater when different wetland types combine. This is possible as a result of the occurrence of supplementary abiotic/biotic pollutant removal pathways which is attributed to different physico-chemical conditions present at different wetland configurations. For instance, anaerobic removal pathways are predominant in HSSF CWs, while VSSF CWs are more appropriate for pollutants that can be easily biodegraded under aerobic conditions. Similarly, FWS wetlands can take the advantages of the effect of photo-oxidation and other processes for the removal of emerging organic contaminants (Avila *et al*, 2015). Moreover, Masi and Martinuzzi (2007) revealed that it is possible to reduce the total surface area when hybrid configuration is employed and consequently the water loss via evapotranspiration is decreased. In general different environmental conditions such as aerobic, anoxic, and anaerobic conditions can increase the performance of CWs (Haberl *et al*, 2003).

Hybrid constructed wetland system is being applied by combining various types of constructed wetlands in order to achieve higher treatment efficiency especially for nitrogen removal. Hybrid systems combining VF and HF beds are the most common ones and proved to be more efficient for practical application (Vymazal *et al*, 1998; Merlin *et al*, 2002; Vymazal, 2005).

High loads of organic matter, nitrogen, suspended solids, pathogens and chemicals can be removed using the hybrid subsurface CW system and the efficiency is high and steady during both cold and warm seasons. But N transformation and concentration is affected by total carbon concentration available in the system (Rousseaou, Vanrolleghem and Pauw, 2004; Masi and Martinuzzi, 2007; Abidi *et al*, 2009; Vymazal and Kropfelova, 2015; Zhang *et al*, 2016). Abidi *et al* (2009) reported that HF bed seems to be a promising design for denitrification while VF bed a potential design for the process of nitrification. But in general, VF-HF hybrid system showed great potential for the accomplishment of nitrification to the level that is required although its capability for the removal of nitrate nitrogen is not good. In the meantime, significant differences of pollutant removal processes are demonstrated where the VF and HF beds are alternated, so that the desired configuration should be chosen to attain highest removal efficiency (Gaboutloeloe *et al*, 2009).

2.5 Design Factors of Constructed Wetlands

Several complex processes take place in constructed wetlands treating wastewater. The removal of pollutants in wastewater results from the supportive and mutually dependent actions of several components. These are: substrate, vegetation, and microorganisms. So, the selection of those components based on adequate knowledge and skill helps to achieve high system performance. In line with this, various design factors of CWs are being considered in order to optimize the capability of the wetland system. On the other hand, the key design factors include: wetland plants, substrates, retention time, and water depths (Dordio and Carvalho, 2013; Upadhyay *et al*, 2016).

Alley *et al* (2013) pointed out that giving adequate attention to seasonal factors for instance temperature and evapotranspiration during designing helps to optimize the wetland performance in removing targeted pollutants for the seasons. Similarly, Valsero *et al* (2012) revealed that pollutant removal efficiency of constructed wetlands is marked by seasonal difference. On the other hand, the effect of temperature is not clearly known and the available data on it is sometimes contradictory, and also contradictory formulas were formulated by wetland designers to determine the hydraulics and size of CWs in areas which have different climatic conditions (Siti *et al*, 2011). Generally, CW configurations applied under similar environmental and hydraulic load greatly vary (Valsero *et al*, 2012).

2.5.1 Wetland Plants

Wetland plants and its litter are among essential components of CWs system in improving the performance and giving attractive aesthetic value. They make most of the major visible structure of CWs. Wetland plants or macrophytes can grow well in wetland system and show significant removal efficiency in treating different wastewaters. It is realized that they are used in virtually all wetland types for increasing the performance and getting better effluent quality which can meet the discharge requirements. The reports of many studies on planted and unplanted wetland system concluded that the performance of wetlands is high in the presence of macrophytes (US EPA, 1993; Thomas and William, 2001; UN-Habitat, 2008b; Kadlec and Wallace, 2009; Upadhyay *et al*, 2010).

Wetlands are typically dominated by macrophytes. Wetland plants can acclimatize water saturated environment and they can also tolerate anaerobic environments caused by the excess water content. Compared with terrestrial plants, they show a worldwide similarity. This similarity overrules climatic conditions and is imposed by a free water supply common characteristics and oddly harsh chemical environment that must be tolerated by plants. Macrophytes develop different functional mechanisms to survive the unfavorable environmental conditions (Russo, 2008). Besides this, the basic nutritional requirements of macrophytes and other terrestrial plants are similar (Robert, 2004). In general, pollutants and nutrients in wastewater are taken up by aquatic plants, as a removal pathway in treatment processes that take place in constructed wetlands (Sarmento *et al*, 2012; Bialowiec *et al*, 2014).

Zhang *et al*, (2009) reported that the removal efficiency of planted CWs is higher than unplanted CWs for certain pollutants such as TN and NH_4-N although the role of plants for the removal of BOD_5, COD and TP is limited. Moreover, the performance of planted wetlands in removing nitrogen is usually found to be efficient and steady in all months of the year (Lee and Scholz, 2007; Fonkou *et al*, 2011; Abou-Elela and Hellal, 2012; Mesquita *et al*, 2012).

Wetland plants have vital roles in providing attachment site for microorganisms, sufficient surface area for pollutant adsorption, and diffusion of atmospheric oxygen to the rhizosphere, adequate hydraulic residence time, and trapping and settlement of suspended wastewater constituents as a result of resistance to hydraulic flow. The stem and leaves in the water column on the other hand, help for improved sedimentation and used by microorganisms as a substrate for their multiplication. All these things can have an effect on the plant-microorganisms-wastewater interactions and then treatment performance of the wetland system. Therefore, the proper structural development and the general growth rate of wetland plants as supposed to be applied in constructed wetland system is highly important (Kyambadde *et al*, 2004; Chazarenc *et al*, 2004; GEPD, 2010; Dong *et al*, 2016).

Plants in aquatic environments can also have great transpiration potential. Evapotranspiration causes low treatment efficiency in CWs since it increases the concentration of dissolved compounds as water volume decreases, and creates the accumulation of pollutants in soil (Bialowiec *et al*, 2014).

Based on total solids (TS) analyses, the existence of wetland plants increases accumulation/production of solids and intensifies clogging, generating greater headloss and possible surface flow at the inlet of the planted wetlands, when compared to the unplanted systems (De Paoli and Sperling, 2013).

The role of macrophytes in the course of action of wastewater purification should not be undermined while CWs are employed as method of treatment (Dong *et al*, 2016). In general, the effects of wetland vegetations can be summarized as follows (Kadlec and Wallance, 2009; GEPD, 2010):

- ✓ The plant growth cycle seasonally stores and releases nutrients, thus providing a “flywheel” effect for a nutrient removal time series.
- ✓ The creation of new, stable residuals accretes in the wetland. These residuals contain chemicals as part of their structure or in absorbed form, and hence accretion represents a burial process for nitrogen.
- ✓ Submerged litter and stems provide surfaces on which microbes reside. These include nitrifiers and denitrifiers, and other microbes that contribute to chemical processing.
- ✓ The presence of vegetation influences the supply of oxygen to the water. Emergent vegetation blocks the wind, and shades out algae, presumably lowering re-aeration. Floating vegetation may provide a barrier to atmospheric oxygen transfer. Submerged vegetation may provide photosynthetic oxygen supply directly in the water. To some limited extent, plant oxygen flux supplies protective oxidation in the immediate vicinity of plant roots.
- ✓ The carbon content of plant litter supplies the energy need for heterotrophic denitrifiers.

Table 2-1: The role of plants (macrophytes) used in constructed wetlands.

Part of wetland plant	**Role**
Aerial plant tissues	light attenuation: reduced growth of phytoplankton influence on microclimate: insulation during winter reduced wind velocity; reduced risk of resuspension of solids aesthetic appearance nutrient storage
Plant tissue in water	filtering effect: filter out large debris reduced current velocity: increased rate of sedimentation: reduced risk of resuspension surface area for attached microorganisms excretion of photosynthetic oxygen: increased aerobic degradation nutrient uptake
Roots and rhizomes	Stabilizing the sediment surface: less soil erosion Prevents the medium from clogging in vertical flow systems Release of oxygen increase organic degradation and nitrification Nutrient uptake Secretions of antibiotics for detoxification of root zone: pathogen removal

Source: Russo, 2008; p.216

There are reports from several studies on the applicability of special plant species and their capability to improve the removal efficiency when compared to others. As far as known, the type of plants used plays a minor role for domestic wastewater treatment. But the selection of the right plant species can have significant role when unique organic compounds and/or heavy metals are found in the wastewater. In view of this, both the plant productivity and the pollutant removal efficiency are relevant in finding an appropriate plant for a given application (Haberl *et al*, 2003; Guittonny-Philippe *et al*, 2015.).

Haberl *et al* (2003) revealed that treatment efficiency can be improved if a combination of different environmental conditions (e.g. aerobic, anoxic, and anaerobic conditions) is provided and/or different plant species are used. However, there is very little known about the effects of the combination of plants regarding treatment efficiency.

In the selection of appropriate wetland plants for CW system, the most important and widely used criteria include (Russo, 2008; Merchand, *et al*, 2010):

- Ecological acceptability, that is, no significant weed or disease risks or danger to the
- Ecological or genetic integrity of surrounding natural ecosystems;
- Tolerance of local climatic conditions, pests and diseases;
- Tolerance of pollutants and hypertrophic water-logged conditions;
- Ready propagation, and rapid establishment, spread and growth; and
- High pollutant removal capacity, either through direct assimilation or storage, or
- Indirectly by enhancement of microbial transformations.

Additionally, the following points should also be taken in to consideration while wetland plants are selected (Azni *et al,* 2010):

- The species available or suitable for the proposed wetland site,
- The substrate on which the plants will prefer to grow (e.g., sand, mud, clay, peat),
- Aerobic vs. anaerobic conditions and when and where this is likely to occur within the wetland,
- The depth of water in which the plants normally grow, e.g., shallow or deep water,
- The frequency and depth of inundation, and
- Periods of drying and the ability of the plants to withstand drying.

Wetland plants are typically classified into three broad types based on their growth form. These are: floating, submerged, and emergent plants (Robert, 2004; Russo, 2008; Kadlec and Wallace, 2009; Azni *et al*, 2010):

- *Floating*. These plants are not attached to the wetland media; rather they freely float at the water surface. Hyacinth, pennywort, and duckweed are the best examples of floating species. Free floating plants are able to use oxygen and carbon dioxide directly from the surrounding atmosphere and mineral nutrients from the water.

  The roots of floating plants are directed downwards into the water column while the other part of the plants where photosynthesis is carried out is found at or right above the water surface. The process of photosynthesis is accomplished by taking up nutrients from the water, through the root system and using atmospheric oxygen and carbon dioxide. The root system in the water is an ideal place for adsorption or filtration and at the same time for bacterial growth. The development of plant roots, functions as treatment medium, can be affected by those factors such as quality of wastewater, temperature and frequency of harvesting.

The entrance of sunlight into the water column of the wetland system is appreciably limited in the presence of floating plants and the exchange of gases between the atmosphere and water is highly hampered as well. Thus anoxic or anaerobic conditions as a result of organic loading rate and the chosen floating plant species and coverage density can be created while the wastewater becomes algae free.

- *Submerged*. The submerged plant species can be either attached to the substrate or free floating although the stems and leaves of the plants are submerged permanently. Those plants whose flowers may be emergent are grouped under submerged plants.

  These plants can be submerged in the water column or rooted in the bottom sediments. The photosynthetic parts of the plants are usually found in the water. They are theoretically considered as an attractive alternative to polish effluents. But the practical importance can be compromised since there is a possibility of the plants to be harmed by anaerobic condition and to be covered by excessive algal growth.

  Submerged plants have a key role in removing organic nitrogen. The removal of ammonia in the presence of submerged plants is associated with photosynthetic processes. Unlike floating species, submerged species use carbon dioxide found in the water in photosynthesis, the processes that cause the raising of pH and then removing of gaseous ammonia through diffusion into the atmosphere. Ammonia, the gaseous form of nitrogen is usually known for its toxic effect for fish (Kadlec and Wallace, 2009).

- *Emergent*. These types of plants are commonly attached to the substrate of the wetland system. Their stems and leaves of the plants extend above the surface or float on the surface. Plants which can be grouped under emergent are overwhelmed either occasionally or permanently.

The plant root zone (rhizosphere) is the only site for the plants and the wastewater to be treated to get in contact since the flow of the wastewater is through the gravel or aggregates. The symbiotic relationships established between wetland plants with bacteria and fungi, excretion of root exudates and transfer of oxygen affect the surrounding environment of the root zone (Wallace, 2004; Chen *et al*, 2016). An important role is played by fine root than the role played by the entire root system in wastewater treatment and seasons and plant growth can affect removal efficiency (Yang *et al*, 2007; Chen *et al*, 2016).

Table 2-2: Recommended emergent plant species for constructed wetlands.

Recommended species	Maximum water depth[a]	Notes
Arrow arum (*Peltandra virginica*)	12 in.	Full sun to partial shade. High wildlife value. Foliage and rootstocks are not eaten by geese or muskrats. Slow grower. pH: 5.0–6.5
Arrowhead/duck potato (*Saggitaria latifolia*)	12 in.	Aggressive colonizer. Mallards and muskrats can rapidly consume tubers. Loses much water through transpiration
Common three-square bulrush (*Scirpus pungens)*	6 in.	Fast colonizer. Can tolerate periods of dryness. High metal removal. High waterfowl and songbird value
Softstem bulrush (*Scirpus validus*)	12 in.	Aggressive colonizer. Full sun. High pollutant removal. Provides food and cover for many species of birds. pH: 6.5–8.5
Blue flag iris (*Iris versicolor*)	3–6 in.	Attractive flowers. Can tolerate partial shade but requires full sun to flower. Prefers acidic soil. Tolerant of high nutrient levels
Broad-leaved cattail[b] (*Typha latifolia*)	12–18 in.	Aggressive. Tubers eaten by muskrat and beaver. High pollutant treatment, pH: 3.0–8.5
Narrow-leaved cattail[b] (*Typha angustifolio*)	12 in.	Aggressive. Tubers eaten by muskrat and beaver. Tolerates brackish water. pH: 3.7–8.5
Reed canary grass (*Phalaris arundinocea*)	6 in.	Grows on exposed areas and in shallow water. Good ground cover for berms
Lizard's tail (*Saururus cernuus*)	*6 in*	Rapid grower. Shade tolerant. Low wildlife value except for wood ducks
Pickerelweed (*Pontedaria cordata*)	12 in.	Full sun to partial shade. Moderate wildlife value. Nectar for butterflies. pH: 6.0–8.0
Common reed[b] (*Phragmites australis*)	3 in.	Highly invasive; considered a pest species in many states. Poor wildlife value. pH: 3.7–8.0
Soft rush (*Juncus effuses*)	3 in.	Tolerates wet or dry conditions. Food for birds. Often grows in tussocks or hummocks
Spikerush (*Eleocharis Palustris*)	3 in.	Tolerates partial shade
Sedges (*Carex* spp.)	3 in.	Many wetland and several upland species. High wildlife value for waterfowl and songbirds
Spatterdock (*Nuphar luteum*)	5 ft (2 ft min.)	Tolerant of fluctuating water levels. Moderate food value for wildlife, high cover value. Tolerates acidic water (to pH 5.0).
Sweet flag (*Acorus calamus*)	3 in.	Produces distinctive flowers. Not a rapid colonizer. Tolerates acidic conditions. Tolerant of dry periods and partial shade. Low wildlife value
Wild rice (*Zizania aquatica*)	12 in.	Requires full sun. High wildlife value (seeds, plant parts, and rootstocks are food for birds). Eaten by muskrats. Annual, non-persistent. Does not reproduce vegetatively

Source: Handbook of Environmental Engineering, Volume 11, p.335 (Azni *et al*, 2010).

[a] These depths can be tolerated, but plant growth and survival may decline under permanent inundation at these depths.
[b] Not recommended for storm water wetlands because they are highly invasive, but can be used in treatment wetlands if approved by regulatory agencies.

2.5.2 Substrate

The difference in pollutant retention capacity of various materials is attributed to the different characteristics they have (Dordio and Carvalho, 2013). Hence, one of the most important things in the application of CW technology is the scientific and practical selection of materials used as a substrate (Lu *et al*, 2016). Then, the most important approach in determining the usefulness and applicability of certain substrate in CWs is finding of the middle ground between adsorption capacity and hydraulic conductivity. There are also other factors which should be given the required attention. These are: local availability, cost, saturation time and recyclability of saturated filter media (Dordio and Carvalho, 2013).

Once the material is selected, the central single property that must be carefully evaluated when applying as media for constructed wetland is the texture, particularly distribution of the grain size (Arias *et al*, 2001). Danish EPA (1999), as cited in Arias, *et al* (2001), recommends the particle size distribution in terms of D_{10} and D_{60}, which are the typical in particle size distribution. Accordingly, the effective grain size d_{10} should be in range of 0.3±2.0 mm, d_{60} between 0.5 and 8 mm, whereas the uniformity coefficient d_{60}/d_{10} should be less than four in order to decrease the occurrence of clogging by ensuring sufficient hydraulic conductivity. In other words, the use of coarse media can maintain operation of CWs for long period of time by avoiding clogging (Meyer *et al*, 2013).

Throughout the period of setup of CW operation, porosity of the selected media, the growth of plant roots, and formation of biofilm show continuous progress until the operation reaches steady state phase. Accumulation of suspended matter, expansion of roots of plants and attaching of biofilm on the surfaces of the substrate are known to cause lessening in porosity for CWs bed and the media porosity value reaches steady phase after certain operation period. The growth of plant roots and bacterial biofilm attached to different plant parts and the substrate at the setup period increase contaminants accumulation, biodegradation, and finally treatment efficiencies (Zidan *et al*, 2015).

Shutes (2001) pointed out that wastewaters which contain pollutants such as heavy metals and hydrocarbons in it can cause accumulation of these pollutants in the substrate which in due course requires transport and disposal to a sanitary landfill site. However, substrates which are applied in CWs which are constructed for the purpose of treating domestic or agricultural wastewater, relatively free of toxic substances, can be used without substitution for a number of years.

Most of the time, the common media which are used in CWs include different soils, sand, gravels, and crushed stones, either alone or in combination (Kayombo *et al*, 2003). But other extra properties of the media shall be considered if there is a need to enhance the removal of pollutants (for example P and N) (Arias *et al*, 2001). For example, calcium based materials such as calcite and marble (Brix *et al*, 2001) are known for their superior capacity in removing phosphorous while zeolite shows higher ammonium nitrogen removal efficiency (Zou *et al*, 2012).

2.5.3 Retention Time

The wastewater to be treated must stay long enough in the wetland system before the completion of the treatment process, and this length of time is usually known as hydraulic retention time. It is one of the key factors to estimate the performance of a CW system. The physical, chemical, and biological processes which are carried out to remove pollutants in wastewater are greatly influenced by the retention time of the water: i.e. if there is longer retention time, the removal rate of all pollutants, except total coliforms and total suspended solids, will be faster. The removal of COD is highly sensitive to retention time. However, too long retention time may be associated with negative effects (Kayombo *et al*, 2003; Katayon *et al*, 2008; UN-Habitat, 2008b; Su *et al*, 2009; Masi *et al*, 2013; Cakir *et al*, 2015).

2.5.4 Water Depth

Water depth is another essential design criterion taken in to account in optimizing the pollutant removal efficiency of CWs system (Garcia *et al*, 2005). The movement or flow of water in CWs is slow since they have saturated media or shallow water depths. Then, the low water depth and the slow water flow create suitable environment for sediments to settle down while the water flows through the wetland system. Moreover, sufficient contact time among the water, substrate, and wetland surfaces can be obtained in these situations. So, a large variety of substances in wastewater can be decomposed by the action of microbial community as a considerable mass of organic and inorganic materials are available (Azni *et al*, 2010).

Subsurface flow CWs which have a water depth of 0.27m are more efficient for the removal of BOD_5, COD and ammonia, than SSF CWs with a water depth of 0.5m. In general, as the water depth increases from 0.27m to 0.5m, the pollutant removal efficiency decreases. Hydraulic loading rate and/or areal organic loading rate are other factors in regulating treatment efficiency of SSF. Subsurface flow with a medium size of 3.5mm produced effluents of better quality than SSF with a medium size of 10 mm; but the differences were smaller in comparison to the effect of water depth and HLR (Garcia *et al*, 2004;

Garcia *et al*, 2005). Water depth is assumed to have an effect on biochemical reactions which are key processes for organic matter degradation. The removal of organic matter is achieved more importantly by biochemical reactions like methanogenesis and sulphate reduction in CWs with a water depth of 0.5m than in those wetlands with a water depth of 0.27m (Garcia *et al*, 2005).

2.5.5 Seasons of the Year

The available literatures regarding the impacts of seasonal variations on the performance of CWs in treating different wastewaters are not consistent (Siti *et al*, 2011). Vymazal (2014) reported that there is no significant difference between the average outflow concentrations of all monitored parameters in summer and winter periods. For instance, CWs in mountainous regions in the Czech Republic showed very good treatment effect with overall treatment efficiencies between 88% and 94% for BOD_5, 67% and 85% for COD and 74% and 96% for TSS. The removal of these parameters was stable during the year and during the time of operation.

However, a number of studies are in favor of the thought that seasonal removal of constituents from wastewater using CWs can change with location and targeted constituent, so an initial pilot-scale study could be beneficial prior to construction of a full-scale system to estimate the removal rate coefficients of targeted constituents (Alley *et al*, 2013). Seasonal variations seem to affect the performance of CWs technology even if the performance is not consistent for all wastewater parameters. Lower effluent concentrations were observed during the warm period, especially for TN and NO_3-N, whereas the performances improved as wastewater temperature rises. The removal efficiency of NH_4-N can be affected by seasonal differences to a lesser degree in the VF than the HF. The observation depicts that the performance of CWs system during summer season is higher than winter season (Song *et al*, 2006; Prochaska *et al*, 2007; Rousseau *et al*, 2008; Mustafa *et al*, 2009; Mietto *et al*, 2015; Ramprasad and Philip, 2016).

Wu *et al* (2014) described that the operation of CWs at cold climate is a big challenge. A number of adaptation mechanisms are commenced through change of design, natural or artificial thermal insulation and upgraded operation approach, for example artificial aeration. Wetlands with green-house structures can be considered in highly frigid climatic conditions despite of the high investment cost.

In the meantime, the influence of climate on the removal of BOD and TSS by physical mechanisms such as sedimentation and flocculation is less. The absence of plant cover in colder seasons could permit the occurrence of atmospheric re-aeration and solar insolation, but the wetland hydraulics can be changed and solar insolation, atmospheric re-aeration and biological activities can be limited as a result of ice cover. However, there is no influence of the insulating layer created by ice cover on physical processes including filtration, sedimentation, and flocculation (USEPA, 2000).

As in other biological processes, growth rates in aquatic plant systems depend on temperature and the vegetated system show a much better performance during the warmer months of the year (Karathanasis *et al*, 2003; Wang and Li, 2014). The mean TN and TP removals were high in summer (23%) and fall (45%), respectively. Lee *et al* (2013) pointed out that the dependence of removal efficiency on temperature is significant due to plant uptake, which plays a significant role in nutrient removal. Greater bacterial activity is shown during the warmer season than the colder one (Chon and Cho, 2015). So, warmer climate improves performances, especially for nitrification (Masi *et al*, 2013; Molle *et al*, 2015).

Rai *et al* (2015) reported that accumulation of trace element in summer season was high in comparison to winter. A substantial change from winter to summer was observed for Zn (68.40–83.48%), As (63.18 - 82.23%) and Cr (64.5 - 81.63%) while other trace elements showed little difference.

2.6 Pollutant Removal Mechanisms in Constructed Wetlands

A number of complex processes engaging all physical, chemical, and biological mechanisms are undertaken in constructed wetland system to transform and separate various pollutants found in wastewaters. The different removal mechanisms can happen simultaneously or sequentially as the wastewater to be treated gets into the treatment system. Even though the removal processes of the wetlands system are identified, the measurement of these processes to obtain accurate quantitative value is becoming an existing challenge in most cases. The internal interactions, the external input parameters, and characteristics of the wetland are basic factors among others on which the major removal mechanisms and their reaction sequence depend on (US EPA, 2000; Qasaimeh *et al*, 2015; Gokalp *et al*, 2016).

In general, there are two key mechanisms in almost all wastewater treatment systems: namely, pollutant transformations and liquid/solid separations. Gravity separation, absorption, filtration, adsorption, ion exchange, stripping, and leaching are commonly involved in separations, where as transformations are resulted from chemical reactions, including re-dox reactions, precipitation, flocculation, acid-base reactions, or biochemical reactions taking place under anaerobic, anoxic, or aerobic conditions.

Therefore, in CW system, both mechanisms can play a key role in the removal of pollutants while the wastewater gets into the wetland and stay for a certain period of time **(**US EPA, 2000**).**

Adequate knowledge of the fundamental physical, chemical and biological processes governing the performance of wetlands raises the acceptance and application of CWs in a large extent. Similarly, in order to understand the structure and functions of the wetland system, a working knowledge of biogeochemical cycling, the movement and transformation of nutrients, metals and organic compounds among the living and non-living components of the ecosystem is required. This level of understanding and practical knowledge is helpful for evaluating the performance of CWs to remove pollutants (Russo, 2008).

2.6.1 Physical Mechanisms

Filtration and sedimentation are the two most significant processes representing the physical mechanisms. The physical removal of wastewater pollutants related to particulate matter occurs through filtration and sedimentation and these two processes are considered as highly efficient. The presence of plant biomass in all wetlands, and also the substrate in the case of SSF CWs are the chief factors promoting the physical contaminant removal processes in wetland technology (Kayombo *et al*, 2003; Russo, 2008).

Slow flow velocity really helps to improve sedimentation for the removal of suspended solids in CWs system (Brix, 2003; Noh *et al*, 2016). The flow of wastewater is hindered in the wetlands because of the resistance by wetland plants, and this event is responsible to advance sedimentation of SSs. Furthermore, floating plants can also have a primary role in the removal of suspended solids by limiting re-suspension of particulate matter already settled at the treatment bed. On the other hand, similar to the processes taking place in filtration, the media applied to the wetland system is another removal pathway of suspended solids (Russo, 2008).

Another route of physical removal mechanism in wetlands is volatilization, the process of diffusing of dissolved compounds in wastewater into the atmosphere. Several organic compounds and certain simpler inorganics formed as a result of mineralization like ammonia are volatile compounds, which are lost to the atmosphere from CWs. Because of its potential to cause air pollution, volatilization is not a preferred removal process (Russo, 2008).

2.6.2 Biological Mechanisms

There are six biological reactions indentified to be highly significant in the performance of CWs technology. These are: photosynthesis, respiration, fermentation, nitrification, denitrification and microbial phosphorous removal (Cheng *et al*, 2014; Akizuki *et al*, 2018). Carbon and oxygen are added to the wetland system as a result of photosynthesis process carried out by algae and wetland plants nitrification process is enhanced by carbon and oxygen. All living forms in the system oxidize organic carbon (respiration) to produce energy, CO_2 and water. Bacteria, algae, fungi, and protozoa are the typical living forms of wetland systems and they require favorable conditions for optimal performance. Fermentation is another process which refers to the microbial decomposition of organic carbon without the presence of oxygen. Nitrification/denitrification processes are also accomplished by microorganisms to remove nitrogen. Moreover, dissolved nutrients and other contaminants in wastewater are taken up by wetland plants to produce additional biomass (Kayombo *et al*, 2003; Azni *et al*, 2010).

Wetlands can rapidly remove readily degradable organic C compounds typically found in municipal wastewater. Unlike this, various recalcitrant organic compounds such as lignin-based compounds and products of petroleum can also be removed by the help of microbial action in wetlands, even if the rate of removal for these compounds are substantially lower compared to readily biodegradable compounds (Azni *et al*, 2010).

Enzymatic activity is one of the biological processes in CW which plays a major role in releasing nutrients from organic substances and higher nutrient loading into wetlands reduced the nutrients removal efficiency of wetlands. Also, enzyme activity is continuously contributing to the release of inorganic nutrients which may reduce the wetlands efficiency (Baddam *et al*, 2016).

It is obvious that a quantifiable amount of pollutant uptake and storage occur through microorganisms. But the decomposition of organic compounds throughout the conversion from complex to simple molecules is achieved by the metabolic processes undertaken by microbes. This is the most significant biological mechanism by which a wide range of pollutant organic compounds can be removed. In fact, the removal rate and efficiency of microbial degradation of organic carbon vary depending on the type of compounds (Russo, 2008). Nitrogen gas, in the form of N_2, is released from nitrate and ammonium as a result of microbial metabolism in wetlands, and lost to the atmosphere. This depicts that microbial metabolism can also remove inorganic nitrogen (Vymazal, 2007).

Additionally, plant uptake for the removal of inorganic pollutants, is probably the most widely recognized biological process. Of the contaminants of wastewater, some contaminants like ammonium, nitrate, and phosphate are essential nutrients for plants while others are toxic metals or compounds. Most wetland plants can uptake both types of contaminants and even the accumulation of considerable amounts of toxic chemicals in the plants tissue can occur over time (Russo, 2008; Kadlec and Wallace, 2009; Qasaimeh *et al*, 2015).

2.6.3 Chemical Mechanisms

The removal of pollutants can also be achieved by a number of chemical processes, and the chief chemical processes which are involved in the removal of contaminants are sorption and precipitation. Reduction-oxidation reactions, complexation, and hydrolysis are also other chemical processes which result in pollutant conversions/transformations which are prerequisite for contaminant removal through precipitation or adsorption. Certain unique groups of pollutants can be removed because of chemical processes such as photolysis and ionic exchange with mineral components of the substrates in CWs (Thomas and William, 2001; Russo, 2008).

Sorption, one of the chief chemical processes can be described as:

> ‘S*orption refers simultaneously to both adsorption and absorption phenomena, and the term is used whenever the extent to which each phenomenon is responsible for the compound’s removal is not clear or well defined. These chemical processes occur at the surfaces of plants roots and substrate, resulting in a short-term retention or long-term immobilization of the contaminants.*’ Russo (2008)

The sorption removal process which occurs in wetlands is influenced by the following three factors (Russo, 2008):

1. Substrate characteristics (texture, content in organic matter and ion exchange properties),
2. Wastewater characteristics (dissolved organic matter content, pH and electrolyte composition), and
3. Pollutant characteristics; when pollutants have acid-base properties, the pH in the liquid compartment can be the cause to influence the degree of sorption process to mineral surfaces through ion exchange properties.

Sometimes, the binding of contaminants to mineral surfaces is facilitated by the occurrence of complexation, although the complexation phenomenon itself also helps to increase the solubility of pollutants in another time (Russo, 2008).

Precipitation, which depends on re-dox condition and pH, is also another chemical process by which long-term pollutant removal occurs. The formation of insoluble oxides and hydroxides by hydrolysis and the conversion of soluble ionic compounds into neutral insoluble forms in protonation state are the two important conditions for precipitation to take place. In some cases, sorption process can assist co-precipitation event to happen (Russo, 2008).

The direction of most processes and reactions such as biological transformation, partitioning of ionized and unionized forms of acids and bases, cation exchange, and solubility of gases in wetland system is highly influenced by the pH of water and soils (Kayombo *et al*, 2003) .

Table 2-3: Summary of selected pollutant removal mechanisms in constructed wetlands.

Pollutant	Removal mechanism
Biochemical Oxygen Demand (BOD)	Oxidation Absorption Filtration Sedimentation Microbial decomposition
Suspended solids (SS)	Filtration Sedimentation
Nitrogen (N)	Adsorption Assimilation Absorption Ammonification–nitrification–denitrification
Heavy metals	Adsorption Cation exchange Bioaccumulation
Pathogenic bacteria and viruses	Adsorption Predation Sedimentation Sterilization by UV
Other pollutants	Precipitation Evaporation Evapotranspiration

Source: Handbook of Environmental Engineering, Volume I, p.329

2.7 Removal of Pollutants in Constructed Wetlands

2.7.1 Removal of Organic Compounds

Although physical processes (sedimentation and filtration) are capable of removing organic matter near the inlet of SSF, the major removal process of organic matter in CW system is microbial degradation (Garcia *et al*, 2005; Azni *et al*, 2010). Domestic wastewater usually contains readily biodegradable dissolved organic compounds and when the dissolved organic matter gets into the biofilms attached on the root system, submerged plant stems and litter, and the surrounding fill media, the biodegradation process takes place (Azni *et al*, 2010). Macrophytes supply oxygen to the root system and give support medium for the occurrence of microbial degradation (Kayombo *et al*, 2003; Robert, 2004; Azni *et al*, 2010). But the role of plant uptake for the removal of BOD_5 and COD is much lesser than NH_4^+-N and TN (Zhang *et al*, 2009).

The carbon content of organic matter reaches roughly 45–50% and a wide range of microbes in wetlands utilize the carbon as a source of energy. The process of breaking down of organic carbon to CO_2 by microorganisms to obtain energy for growth requires dissolved oxygen (Azni *et al*, 2010).

In fact, both aerobic and anaerobic decomposition processes can take place in the removal of organic matter. However, the nature of biochemical reactions depends on the conditions created as a result of the rate of oxygen transfer in the wetland and the load of internal and external organic load (Wallace, 2004). If there is adequate oxygen supply, aerobic decomposition is so rapid that the accumulation of organic matter in the wetland is small. But if the rate of oxygen transfer cannot fulfill the oxygen requirements, the removal process is anaerobic decomposition which results the accumulation of organic matter in the wetland (Garcia *et al*, 2005).

Generally, there are reports which describe that the performance of CWs for the removal of BOD_5 and COD from municipal/domestic wastewater is significantly high (Vymazal and Kropfelova, 2009; Azni *et al*, 2010; Vymazal, 2014b). Even, the HFCWs, in which the rate of transfer of oxygen in the system is expected to be relatively low, can show high performance in removing BOD_5 and COD and result quality effluent that fulfills the requirement for discharge in terms of the two parameters (Cakir *et al*, 2015; Albalawneh *et al*, 2016).

On the other hand, the degradation of organic matter by microorganisms is influenced by climatic conditions and consequently the rate of degradation in tropical areas is higher than temperate or cold areas (Anish *et al*, 2012). Song *et al* (2006) reported that removal efficiencies for BOD_5 and COD vary from season to season (10% less efficient in winter and autumn compared to summer and spring). In the contrary, Paing *et al* (2015a) revealed that temperature has no effect on the removal of BOD_5, COD, and many other parameters.

2.7.2 Suspended Solids

Suspended solids are the typical contaminants in wastewater and are originated from either internal or external sources. The external sources are usually the influent and atmospheric inputs whereas planktons and animal and plant detritus are created within the wetland. Wastewater commonly constitutes 99.9% water and the rest 0.1% is solids. Hence, suspended solids are an essential parameter in water quality monitoring and therefore applied to measure the quality of influent and effluent, and also to evaluate the performance of many processes (Thomas and William, 2001; Kayombo *et al*, 2003; Kadlec and Wallace, 2009).

Contaminants such as organic compounds, nutrients, and heavy metals are constituents of suspended solids. These contaminants can exist in particulate form in wastewater or they can be bound to other particulate matter either physically or chemically. Therefore, the removal of contaminants from wastewater and water source through sedimentation of suspended solids can be effective in conditions where the mass of the contaminant load binds with particulate matters (Thomas and William, 2001).

Wetlands are able to remove suspended solids from wastewater efficiently (Thomas and William, 2001; Avsar *et al*, 2007; Kadlec and Wallace, 2009). The system has normally extended HRTs and low flow velocity, which create desirable conditions for easily removal of settleable solids by gravitational settlement. Alternatively, processes including biodegradation, adsorption on submerged parts of the plant and wetlands media, filtration, and flocculation/precipitation are involved in the removal of colloidal or non-settling solids. In the removal of suspended solids, the nature and size of contaminant solids and the type fill media are the major factors on which the practical action of each removal mechanism depends (Kayombo *et al*, 2003; Robert, 2004; Vymazal, 2009; Azni *et al*, 2010).

Lana *et al* (2013) pointed out that the decreasing volume of wastewater to feed the wetland by increasing the batch frequency possibly raises the HRT. Then when the HRT becomes longer, the contact between bacteria in the wetland system and wastewater becomes better and the retention of suspended solids will be improved. This condition enables the system to present better pollutant removal efficiency.

In general, the settling rate of particles and the wetland length highly affects the efficiency in removing SSs (Thomas and William, 2001). The process of sedimentation is thought to be irreversible. Nevertheless, suspended solids may be released from the sediment as re-suspension as a result of high flow velocity, wind driven turbulence, animals and human disturbance, and gas lift occurred by oxygen, methane and carbon dioxides. Particle re-suspension does not occur in HSSF and VF CWs although excessive biological growth, creating head loss through the wetland system, leads to overland flow in

HSSF system and complete failure of VF wetland (Thomas and William, 2001; Kadlec and Wallace, 2009).

2.7.3 Removal of Nutrients

The other pollutants of concern for water bodies are nutrients; for the most part, nitrogen and phosphorous. Usually, either phosphorous or nitrogen is the limiting factor in an aquatic environment. Therefore, discharging of untreated wastewater, containing excess nutrients, into water bodies results the onset of eutrophication in water bodies. Moreover, it can cause other adverse effects, for example, ammonia toxicity to aquatic life and public health problem because of the presence of excessive nitrate in drinking water (Russo, 2008). Nicholas (2002) depicts that taking out N or P from wastewater is referred to as nutrient removal.

Different forms of nutrients undergo different routes of transformation among different wetland sections including substrate, plants, water, and litter. Then, wetlands are considered as nutrient sink following this transformation pathway for nutrient cycling. Hence, CWs are considered to be less efficient in removing nutrients (Rousseau *et al*, 2004; Vymazal, 2005; Konnerup *et al*, 2009; Abou-Elela *et al*, 2014).

However, they still have a good stand to be reliable alternative treatment methods to alleviate the problems associated with indiscriminate discharge of nutrients. The mechanisms for the removal of nutrients include: direct plant uptake, uptake by algae and bacteria, chemical precipitation, soil absorption, nitrification/denitrification, and reduction by fish and insect uptake (Li *et al*, 2008; Russo, 2008; Kadlec and Wallace, 2009; Qasaimeh *et al*, 2015).

Nutrient uptake by plants shows a higher fraction of N removal in FWS, and a higher fraction of P removal in SSF CWs. The contribution of plants in the removal of nutrients in wetlands is considered to be high (Konnerup *et al*, 2009; Lee *et al*, 2013; Bialowiec *et al*, 2014; Zheng *et al*, 2016; Wang *et al*, 2016). In order to increase the nutrient removal rate by plant uptake, it is suggested that the treatment regions, in the CW need to be covered by plants (Lee *et al*, 2013). Even though the capacity of plants to take up nutrients varies from species to species, it is more dependent on individual plant biomass irrespective of plant type, i.e., on the size of individual plants or plant density (Adhikari *et al*, 2011; Dzakpasu *et al*, 2015).

2.7.3.1 Nitrogen removal

Nitrogen is among the most important constituents in wastewater since it causes eutrophication, toxicity to aquatic life and undesirable consequence on the level of oxygen in the water bodies (Kadlec and Wallace, 2009). Proteinaceous matter and urea are the two principal components in which nitrogen can be available in domestic wastewater. If the condition in the pretreatment is kept anaerobic, ammonia, ammonium (NH_4^+) is produced as a result of the breaking down of protein and urea and the remaining organic N will be converted to ammonium by the process of ammonification (Wallace, 2004).

The presence of N in the environment has many forms even though the transformations among those forms may occur quickly. Substantial quantity of both organic and inorganic N forms can exist in wastewater. In wetlands that are designed to treat municipal or domestic wastewater, the most significant N forms are ammonia (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), nitrous oxide (N_2O), and dissolved elemental N or dinitrogen gas (N_2). In most cases, dissolved forms of N are commonly present in wetlands even if little particulate N can exist in settled wetland surface waters. Nitrite and, particularly, nitrate nitrogen are often found in waters where there is adequate oxygen while ammonium is the most common in anaerobic wetland soils (Thomas and William, 2001; Tanner, 2004; Kadlec and Wallace, 2009; Liu *et al*, 2014; Jahangir *et al*, 2016).

The total or dissolved forms of N which are determined using common analytical methods (APHA, 1999) include:

- Nitrate
- Nitrite
- Ammonia
- Total Kjeldahl nitrogen (TKN) = (organic +ammonia nitrogen)

So, the following formulas can be derived from the above basic measurements (Kadlec and Wallace, 2009):

Oxidized nitrogen = nitrate + nitrite Eqn. (2.1)

Inorganic nitrogen = oxidized nitrogen +ammonia Eqn. (2.2)

Organic nitrogen = TKN – ammonia Eqn. (2.3)

Total nitrogen = TKN + oxidized nitrogen Eqn. (2.4)

The major transformation processes functioning in constructed wetlands are:

ammonification (organic N $\rightarrow$ NH_4^+), nitrification (NH_4^+ $\rightarrow$ NO_2^- $\rightarrow$ NO_3^-) denitrification (NO_3^- $\rightarrow$ N_2O $\rightarrow$ N_2), biological fixation (N_2 $\rightarrow$ organic N), nitrate ammonification (NO_3^- $\rightarrow$ NH_4^+), anaerobic ammonia oxidation (ANAMMOX, NH_4^+ $\rightarrow$ N_2) and volatilization (NH_4^+ $\rightarrow$ NH_3) (Vymazal, 2009).

Nitrogen compounds also enhance plant growth, which in turn stimulates the biogeochemical cycles of the wetland. The wetland N cycle is very complex, and control of even the most basic chemical transformations of this element is a challenge in ecological engineering (Kadlec and Wallace, 2009; Despland *et al*, 2014). VF CWs remove ammonia-N successfully but very limited denitrification takes place in these systems. On the other hand, HFCWs provide good conditions for denitrification but the ability of these systems to nitrify ammonia is very limited. Therefore, various types of CWs may be combined (hybrid systems) with each other in order to exploit the specific advantages of the individual systems (Vymazal, 2007; Zurita and White, 2014; Haghshenas-Adarmanabadi *et al*, 2016).

The level of N removal in wetlands system is relatively high, despite of the fact that the natural background level in the effluent is frequently greater than 1 mg/L due mainly to breaking down and release of the native organic matter (Thomas and William, 2001). There are many N removal mechanisms involved in CWs (Figure 2.3). These include: volatilization, ammonification, nitrification/denitrification, uptake by plants and adsorption with wetland. The major removal mechanism in most of the CWs is microbial nitrification/denitrification. Ammonia is oxidized to nitrate by nitrifying bacteria in aerobic zones. Nitrates are converted to dinitrogen gas by denitrifying bacteria in anoxic and anaerobic zones (UN-Habitat, 2008b; Wu *et al*, 2013).

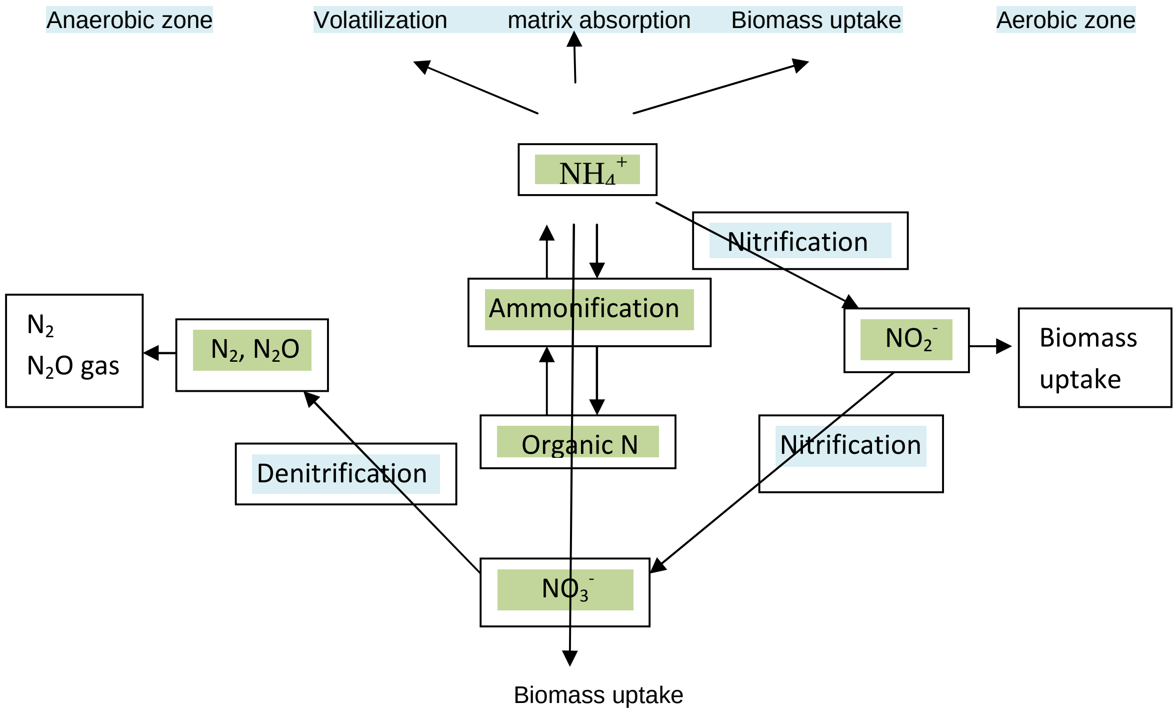

Figure 2.3: Nitrogen transformation in constructed wetlands (UN-Habitat, 2008b)

2.7.3.1.1 Volatilization

Diffusion of dissolved compounds into the atmosphere is one of the possible mechanisms of contaminant removal in wetlands and the process is referred to as volatilization. Quite a lot of organic compounds are readily lost to the atmosphere from wetlands and other water surfaces since they are volatile. Volatilization of unionized ammonia (NH_3) can result in considerable N removal if the pH of the water is high (greater than about 8.5). But if the water pH is low or neutral, ammonia N occurs virtually totally in the ionized form (ammonium, NH_4^+) which is not volatile (Thomas and William, 2001; Tanner, 2004; Kadlec and Wallace, 2009). In wetlands, high pH is created during the day time as a result of photosynthesis by algae and submerged macrophytes (Russo, 2008; Vymazal, 2009; Yin *et al*, 2016; Ding *et al*, 2018).

Vymazal and Kropfelova (2008) explained that volatilization of ammonia is a physicochemical process in which ammonium N is known to be in equilibrium between gaseous and hydroxyl forms as indicated by the following equation:

$$NH_3\,(aq) + H_2O \quad \rightarrow \quad NH_4^+ + OH^- \qquad \text{eqn. (2.5)}$$

The removal of NH_3 through volatilization from flooded soils and sediments are insignificant if the pH value is below 7.5 and very often losses are not serious if the pH is below 8.0. At a pH value of 8.0 approximately 95% of the ammonia N is in the form of NH_4^+. At pH of 9.3 the ratio between ammonia and ammonium ions is 1:1 and the losses via volatilization are considerably high (Vymazal, 2009).

In general, the rate of volatilization is controlled by the following factors: the NH_4^+ concentration in water, wind velocity, solar radiation, pH values, temperature, solar radiation, the nature and density of vegetation and the capacity of the system to change the pH value in diurnal cycles (absence of CO_2 increases volatilization) (Russo, 2008).

2.7.3.1.2 Ammonification

Ammonification (mineralization) is a process whereby N-containing organic compounds for example proteins, amino sugars, and nucleic acids are biologically degraded to ammonium (NH_4^+) (WI, 2003). It is the primary step in the mineralization of organic matter and can take place under aerobic or anaerobic conditions. The group of heterotrophic microorganisms is normally deemed to be involved in ammonification process (Wallace, 2004; Kadlec and Wallace, 2009). Therefore, in order to achieve higher ammonia utilization in the wetland, more favorable environmental conditions for ammonia oxidizing bacteria should be established in the HSSF wetland (Truu *et al*, 2005; Su *et al*, 2018).

The common ammonification reactions are shown below (Kadlec and Wallace, 2009):

Urea breakdown

$$NH_2CONH_2 + H_2O \rightarrow 2NH_3 + CO_2 \qquad \text{Eqn. (2.6)}$$

Amino acid breakdown

$$RCH(NH\ )COOH + H_2O \rightarrow NH_3 + CO_2 \qquad \text{Eqn. (2.7)}$$

Animal and plant tissues and excreted urea are the main sources of nitrogenous organic compounds. Domestic wastewater contains almost all N in the form of organic N or ammonia (Wallace, 2004; Kadlec and Wallace, 2009). Eventually all organic N is degraded into ammonia (NH_3), during pretreatment or soil-based treatment processes (Wallace, 2004). Ammonium (NH_4^+) is primarily resulted from mineralization of N within flooded wetland soils. The rhizome and the root systems of macrophytes can absorb the soil-bound ammonium and another process which is carried out by anaerobic microorganisms can again reconvert this ammonium to organic matter (Scholz, 2006).

Under aqueous conditions, ammonium (NH_4^+) is formed by the rapid hydrolysis of ammonia (NH_3), as shown in the following equation (Wallace, 2004).

$$NH_3 + H_2O \rightarrow NH_4^+ + OH^- \qquad \text{Eqn. (2.8)}$$

To all intents and purposes, the conversion of virtually all N to ammonium (NH_4^+) form can be deemed before the occurrence of further treatment (Wallace, 2004). Then the ammonium mineralized from N-containing organic compounds does not stay for long period of time in the soil. Rather, it will be converted quickly to other forms of N in the soil/plant system via different processes (Vymazal, 2009).

The rate of mineralization in treatment wetlands system depends on various factors including microbial biomass, C/N ratio of the residue, temperature, available nutrients, pH value, extracellular enzyme, soil conditions such as texture and structurs, soil redox conditions (Reddy and D'Angelo, 1997). The pH range 6.5 and 8.5 is the optimum pH value for the process of mineralization. Unlike a number of microbiological processes, ammonification requires a temperature range of 40 to 60^{o}C even though it is not likely to acquire these temperatures in the field (Reddy and Patrick, 1984; Vymazal, 2009).

Kinetically, the rate of ammonification is faster than nitrification reaction (Kadlec, and Knight 1996), and it takes place at all degrees of soil aeration, although the rate varies depending on the level of aeration. It goes on at a much slower rate in flooded soil system than in drained-soil system. Meanwhile, mineralization occurs at fastest rate in the oxygenated section, and the rate declines as mineralization changes from aerobic to facultative anaerobic and obligate anaerobic microflora (Reddy and Patrick 1984). Basically, in flooded soils, the aerobic zone has depth less than 1 cm, and hence the role of aerobic mineralization to the total mineralization could be very low, compared to facultative anaerobic and obligate anaerobic mineralization. Nitrogen mineralization can be stimulated by frequent drying and rewetting (Vymazal, 2009; Jia *et al*, 2017).

2.7.3.1.3 Nitrification

The process of converting ammonium nitrogen (NH_4^+), by chemoautotrophic bacteria to nitrate (NO_3^-) with nitrite (NO_2^-) as an intermediate product in the reaction, is known as nitrification (Vymazal *et al.*, 1998; Vymazal, 2007; Russo, 2008). In the first step (Eqn. 2.8), ammonium nitrogen is oxidized to Nitrite by *Nitrosomonas* and in the second step (Eqn. 2.9); the oxidation of nitrite to nitrate by *Nitrobacter* is taken place (WI, 2003). During the oxidation process, about 7.14 g of alkalinity as $CaCO_3$ are consumed, and 4.3g of 0_2 are required to convert 1g of ammonium nitrogen to nitrate. The process of nitrification depends on temperature and pH (US EPA, 1993; US EPA, 2000; Scholz, 2006).

$$2NH_4^+ + 3O_2 \xrightarrow{\textit{Nitrosomonas}} 2NO_2^- + 2H_2O + 4H^+ + \text{energy} \qquad \text{Eqn. (2.8)}$$

$$2NO_2^- + O_2 \xrightarrow{\textit{Nitrobacter}} 2NO_3^- + \text{energy} \qquad \text{Eqn. (2.9)}$$

Robert (2004) pointed out that nitrification and denitrification processes are the most important mechanisms for nitrogen removal. Nitrification may occur by suspended bacteria or within any aerobic biofilms in aerobic regions of the soil and surface water. Nitrate remains in the water or pore of water of the sediments as it is not immobilized by soil minerals. It may be absorbed by plants or microbes in assimilatory nitrate reduction or may undergo dissimilatory nitrogenous oxide reduction, denitrification (US EPA, 2000; Thomas and William, 2001). A little ammonium nitrogen which exists in NH_3 form is liberated to the atmosphere through volatilization at elevated pH of 10. Generally, the most suitable sites for nitrification to take place are the oxidized layer and the submerged portions of plants (Azni *et al*, 2010).

Nitrification occurs in virtually all types of CWs; but the availability of oxygen affects the degree of the process. In the majority of CW types nitrification is a limiting process for the removal of nitrogen since NH_3-N is the prevailing nitrogen types in various wastewaters, and in general, DO concentrations greater than 1.5mg/L are necessary for nitrification to take place (Thomas and William, 2001; Ye and Li, 2009).

Tuncsiper (2009) revealed that there is higher N removal efficiency in summer as average temperature rises to 23^{o}C. It is also demonstrated that HSSF wetland system shows higher NO_3^- removal while NH_4^+ removal is lower (Tuncsiper, 2009). Apparently, the system provides suitable environmental conditions for denitrification but limited conditions for nitrification. In contrast, the VSSF wetland system has a higher NH_4^+ removal efficiency as VSSF system has better aeration (Abou-Elela *et al*, 2013; Collins and Gillies, 2014).

In a FWS system, nitrification occurs at the aerobic zone in the water column. Then, the nitrate disperses into the sediments where anaerobic conditions required for denitrification exist. In this system, although the concentration of oxygen for nitrification is limited, both nitrification and denitrification processes can occur to remove nitrogen (Robert, 2004). Zhang *et al* (2012) described that the presence of wetland plants considerably improves both oxidation of ammonia and removal of TP in both batch and continuous types of operation as compared to that for unplanted beds.

The removal rate of N with HF CW system alone is low due to the deficiency of nitrification. On the other hand, single-stage VF CW cannot attain high N removal since environmental conditions that favor denitrification process lack. High removal of N can be realized in hybrid CWs system where the combination of HF and VF beds is applied (Canga *et al*, 2011).

2.7.3.1.4 Denitrification

The dissimilatory biological reduction of nitrate nitrogen to nitrogen gas under anaerobic or anoxic conditions is termed as denitrification (Eqn. 2.10). In this process, organic carbon is used as electron donor while nitrate act as alternate electron acceptor (US EPA, 1993) (Thomas and William, 2001; US EPA, 2000; Kayombo, 2003). Organic compounds are oxidized by chemoheterotrophic denitrifiers for energy and carbon source. Some of these denitrifying bacteria are: *Pseudomonas, Micrococcus, Bacillus* and *Achromobacter* (Brady and Weil, 2002; Russo, 2008). Denitrification is carried out mainly in the sediments of the wetland and in the periphyton films where the availability of carbon is high and the concentration of DO is low. For denitrification to take place the minimum carbon to nitrate-nitrogen ratio would be approximately 1 g C/g NO_3-N (US EPA, 2000). The process of denitrification is demonstrated in the following equation:

$$6(CH_2O) + 4NO_3^- \xrightarrow{\text{Denitrifying bacteria}} 6CO_2 + 2N_2 + 6H_2O \qquad \text{Eqn. (2.10)}$$

Where CH_2O represents biodegradable organic matter (Wallace, 2004)

This reaction occurs under anaerobic or anoxic conditions and it is irreversible by its nature. In the reaction nitrogen acts as electron acceptor instead of oxygen. Meanwhile, a number of evidences from pure culture studies have made it clear that denitrification can occur in the presence of oxygen. Therefore, nitrate reduction may start in water logged sediments before the oxygen is depleted (Laanbroek, 1990; Vymazal, 2009). Denitrification does not take place in the presence of oxygen theoretically. But the reaction was shown in suspended and attached growth treatment systems which constitute fairly low DO concentration, but not above 0.3–1.5 mg/L (US EPA, 1993; Kadlec and Wallace, 2009)

As denitrification progresses, the N_2 is then lost to the atmosphere for permanent removal, and is not stored in the wetland. The removal of ammonium in wetlands can occur as a result of the sequential processes of nitrification and denitrification. Ammonium is transformed to nitrate in aerobic regions of the soil and surface water. Then, the newly formed nitrate may undergo denitrification when it diffuses into the deeper or anaerobic regions of the soil. The coupled processes of nitrification and denitrification are universally important in the cycling and bioavailability of nitrogen in wetland and upland soils (Thomas and William, 2001; Li *et al*, 2015).

In most CW types, denitrification plays the major role in the removal of nitrogen although the nitrate concentration in wastewater is usually low (Thomas and William, 2001). Environmental factors known to influence denitrification rates include the absence of O_2, redox potential, substrate moisture, temperature, pH value, presence of denitrifiers, substrate type, organic matter, nitrate concentration and the presence of overlying water (Vymazal, 2007; Russo, 2008). However, Robert (2004) explained that pH, temperature, organic carbon, nitrate levels, and the ecological interactions and exposure times of the denitrifying bacteria within the system are the key factors. In general, although the reaction dependent on a number of factors, denitrification is the permanent removal of Nitrogen from the system (WI, 2003).

2.7.3.1.5 Plant Uptake

Plants take up nutrients to maintain normal metabolism processes and show an average N:P ratio of about 7:1 under natural conditions. But luxury uptake of N and P by plants can be resulted in situations where there is high concentration of these nutrients (Robert, 2004; Kadlec and Wallace, 2009). Nutrients uptake is usually the function of the roots in wetland soils of nutrients is normally the function of the root systems in the wetland soils and sometimes adventitious roots which are found in the water column. Nutrients may reach up to the leaves and stems of wetland plants (Kadlec and Wallace, 2009).

In addition to microbial removal mechanisms, plant uptake and storage of nutrients in the sediment could be the chief N conversion and removal routes in CWs in treating wastewater (Wu *et al*, 2013). In planted CWs system, there is higher N and COD removal (Wang *et al*, 2016), and plant uptake is one of the major means to remove nitrate produced by the process of nitrification (Robert, 2004).

Nitrogen in the mineralized state is taken up by wetland macrophytes and used to build plant biomass. As plants die out, some of the accumulated N in plant tissues can leach into the wetland system, and therefore, there is no net N removal through plant uptake (WI, 2003). But the removal of N by plant uptake can be efficient if plant biomass is regularly harvested and removed from the wetland system (Robert, 2004; Li *et al*, 2008; Azni *et al*, 2010). Adhikari *et al* (2011) pointed out that harvesting of aboveground plant parts is sufficient for N removal since there is usually higher N concentration in those parts. Lee *et al* (2012) reported that the average phosphorus contents in aboveground tissues of plants obtained ranges between 1.2± 0.7 to 2.4± 1.0 mg/g.

In the process of assimilation, the plants reduce inorganic N to organic N compounds, plant structure. There is significantly high rate of N uptake by wetland plants from water and sediments during the growing season. Increased immobilization of nutrients by microbes and uptake by algae and epiphytes also lead to retention of inorganic N. The net annual uptake of N by macrophytes approximately ranges between 0.5 to 3.3 g $N/m^2/yr$ (US EPA, 2000). However, N uptake by wetland plants reduces while its concentration and load in wastewater rises up. This shows that plants capability for N uptake is limited and it can be considered as efficient method under situations where the load of N is minimal (Avsar *et al*, 2007). Zheng *et al* (2016) reported that plants nutrients uptake accounted for a higher proportion of the N removal in FWS, and higher proportion of P removal in SSF wetland system.

Generally, macrophytes enhanced N removal and processing while reducing GHG fluxes compared to unplanted CWs (Landry *et al*, 2009). Plants utilize nitrate and ammonium, and decomposition processes release N back to the water. There are two direct effects of vegetation on N processing and removal in treatment wetlands (Kadlec and Wallace, 2009):

1. The plant growth cycle seasonally stores and releases N, thus providing a “flywheel” effect for a N removal time series.
2. The creation of new, stable residuals accretes in the wetland. These residuals contain N as part of their structure, and hence accretion represents a burial process for N.

2.7.3.1.6 Matrix Adsorption

Unlike the oxidized forms of N, ammonium nitrogen (NH_4^+-N) can bind to inorganic and organic solid substrates, because of the positive charge it possesses. Ammonium ion is adsorbed onto active cation exchange sites of the wetland bed matrix (Kadlec and Wallace, 2009). In FWS, the ionized ammonia in water can be removed through exchange with inorganic sediments and plant detritus, or with the wetland media in the case of SSF systems. At a given ammonia concentration in the water column, a fixed amount of ammonia is adsorbed to and saturates the available attachment sites. But when the ammonia concentration in the water column is reduced due to factors such as nitrification, some ammonia will be desorbed to regain the equilibrium with the new concentration. If the ammonia concentration in the water column is increased, the adsorbed ammonia also will increase (Thomas and William, 2001; Kadlec and Wallace, 2009).

But when the chemistry of the water is changed, the adsorbed ammonia is leached back into the water system since it is bound loosely to the substrate. Furthermore, the sorbed ammonium can be converted to the oxidized form, nitrate if there are conditions such as periodic draining, in which the substrate of the bed is exposed to oxygen. Hence, ammonium adsorption is a reversible removal process (WI, 2003; Kadlec and Wallace, 2009; Azni *et al*, 2010).

2.7.3.2 Phosphorus Removal

Phosphorus (P) is one of the essential macronutrients required by plants for growth, and is a limiting factor for the growth of vegetation (Kadlec and Wallace, 2009, Ohio EPA, 2011). Hence, addition of P to the environment often contributes to the occurrence of eutrophication of lakes and coastal waters (Thomas and William, 2001). A measure of relative ecosystem requirements is the proportion among the nutrient elements in the biomass, which is often represented as a molar proportion of C: N: P =106:16:1, or 41:7:1 on a mass basis (the Redfield ratio). Wastewaters do not have this ratio except by rare chance, and most often, there is excess P in domestic wastewater. The introduction of trace amounts of this element into receiving waters can have profound effects on the structure of the aquatic ecosystem (Kadlec and Wallace, 2009). Removal of P is required where the wastewater effluent is discharged into a lake or into a watercourse which later discharges into a lake (Brix, 2004).

The most reactive forms are the dissolved phosphates, which change hydration in response to pH. The most common species are mono- and dibasic phosphates, which dominate at all typical wetland pH values ($4 < pH < 9$) (Kadlec and Wallace, 2009)

$$H_2PO_4^- \rightleftharpoons HPO_4^{2-} + H^+ \qquad \text{Eqn. (2.11)}$$

The generic term used for these inorganic phosphate ions is *orthophosphate* (PO_4-P) (Kadlec and Wallace, 2009). Soluble reactive P is the analytical term given to biologically available orthophosphate, which is the primary inorganic form. The availability of P to plants and micro-consumers is limited due to the following main effects: (Scholz, 2006).

- Under aerobic conditions, insoluble phosphates are precipitated with ferric iron, calcium and aluminium;
- Phosphates are adsorbed onto clay particles, organic peat, and ferric and aluminium hydroxides and oxides; and
- Phosphorus is bound up in organic matter through incorporation into bacteria, algae and vascular macrophytes.

In wetland soils, P occurs as soluble or insoluble, organic or inorganic complexes. Its cycle is sedimentary rather than gaseous and predominantly forms complexes within organic matter in peatlands or inorganic sediments in mineral soil wetlands. Over 90% of the P load in streams and rivers may be present in particulate inorganic form (Scholz, 2006).

A combination of physical, chemical, and biological processes is employed in treatment wetlands for removing P from wastewaters (Thomas and William, 2001; Kadlec and Wallace, 2009). Organic forms of P are much less biologically and chemically reactive in wetlands than orthophosphate. Settling of particulate organic P is an important means for its removal. Both dissolved and particulate organic P ultimately may be biologically broken down to inorganic P (mineralization) and subsequently removed through different processes (Thomas and William, 2001). Bacteria removal and plant uptake are responsible for $P\text{-}PO_4^{-3}$ removal, while precipitation and adsorption are responsible for the removal of all P forms (Kadlec and Knight, 1996). Removal of total P varied between 40 and 60% in all types of CWs with removed load ranging between 45 and 75g N/m^2yr depending on CW type and inflow loading (Vymazal, 2007).

The average total P concentration reduction for FWS wetlands was 3.78 to 1.62 mg/l, and that for SSF wetlands, 4.41 to 2.97 mg/l. Respective mass (and percentage) removal rates for FWS and SSF systems were 0.17 kg/ha per day (34 percent) and 1.14 kg/ha per day (22 percent) (Kadlec and Knight, 1996) (Thomas and William, 2001). However, in many cases, wetlands do not provide the high level of efficient long-term removal for P that they provide for N. This is, in part, due to the lack of a gaseous sink, analogous to denitrification, for P removal (Haberl *et al*, 1995; Thomas and William, 2001; Siti *et al*, 2011; Rozema *et al*, 2016).

Li *et al* (2010) reported that P removal is mainly influenced by wetland substrate. The use of new technology and specialized media in the SFCW to improved P removal should be developed and demonstrated since P removal always shows worse performance in the wetlands (Siti *et al*, 2011). If significant P removal is a project requirement then a FWS wetland will probably be the most cost effective type of CW (US EPA, 2000).

Although none of the processes actually remove P from the wetland; they transform and/or store P in materials and compounds which may then re-release the P when conditions change (UNHSC and NEIWPCC, 2010). The principal P removal mechanisms in natural and CW systems are (Robert, 2010; UNHSC and NEIWPCC, 2010):

- Sorption within the bottom soils
- Precipitation of phosphates under elevated pH conditions
- Uptake by the macrophytic plants
- Storage
- Fixation by algae and bacteria

2.7.3.2.1 Sorption

Physical and biological processes are well recognized for removal of pollutants in CWs. Additionally, there are also a number of chemical processes involved in removing contaminants. Sorption is the most important chemical process in the wetland system to remove P. It is a broad term which is defined as the process of the transfer of ions or negatively or positively charged molecules from aqueous phase to the solid phase of the wetland (Thomas and William, 2001). In general, sorption is illustrated as a two-step chemical process (Kadlec and Wallace, 2009)

1. Adsorption:- is the first step by which P rapidly exchanges between the soil pore water and soil particles or mineral surfaces, and
2. Absorption:- is the second step by which P slowly penetrates into solid phases.

Numerous pollutants in wastewater occur as cations and cation exchange involves the physical attachment of cations to the surfaces of clay and organic matter particles in the soil by electrostatic attraction, which is a much weaker force than chemical bonding. So, cations are not permanently immobilized (Thomas and William, 2001). The cation exchange capacity of the soils usually increases with increasing organic matter and clay content. As chemisorption signifies more permanent and stronger bonding than cation exchange, many organic compounds and metals can be immobilized in the soil via chemisorptions with clays, organic matter, iron (Fe), and aluminum (Al) oxides. Phosphate can also bind with clays and Fe and Al oxides through chemisorptions (Thomas and William, 2001; Choi *et al*, 2012).

Dunne and Reddy (2005) as cited in Kadlec and Wallace (2009) revealed that desorption of P can also occur in a two step process. When water of a given concentration is added to a substrate, sorption occurs until the entire soil of the wetland is loaded to the solid phase concentration corresponding to concentration in the water. The time period to saturation can be short for solids with low sorption capacity, but can be long for soils with high sorption capacity.

The factors which control the binding of P through sorption are: particle size, sediment composition, and water chemistry such as ionic strength and concentration of P in the water (UNHSC and NEIWPCC, 2010). Moreover, the contents of different Fe and Al forms are also the main soil variables determining P sorption capacity in the soils. Fe and Al extractable in ammonium oxalate proved to be useful for indirect estimation of P sorption capacity (Borling, 2003). Likewise, the absorption and release of inorganic P was governed by the conversion of iron phosphate and aluminum phosphate to calcium phosphate (Fu *et al*, 2015). Vymazal (2010) pointed out that P removal is generally low unless special media with high sorption capacity are used.

2.7.3.2.2 Precipitation

Precipitation typically involves the reaction of phosphorus with metallic cations such as Fe, Al, Ca, or Mg, forming amorphous or poorly crystalline solids. These reactions typically occur at high concentrations of either phosphate or the metalloid cations. A variety of cations can precipitate phosphate under certain conditions. Some important mineral precipitates in the wetland environment are (Kadlec and Wallace, 2009).

Apatite	$Ca_5 (Cl, F)(PO_4)_3$
Hydroxylapatite	$Ca_5 (OH) (PO_4)_3$
Variscite	$Al (PO_4).2H_2O$
Strengite	$Fe (PO_4).2H_2O$
Vivianite	$Fe_3 (PO_4)_2.8H_2O$
Wavellite	$Al_3 (OH)_3(PO_4)_2 5H_2O$

Iron and Aluminum oxides can precipitate phosphate to form new mineral compounds: Fe phosphate and Al phosphate. These new forms are likely very stable in the soil and create long-term storage of P. Similarly, high concentration of calcium in wetlands can precipitate phosphate to form calcium phosphate, which is stable for long period of time (Thomas and William, 2001). Availability of co-precipitating compounds such as Fe, Al, and DO and water chemistry, especially pH governs precipitation (UNHSC and NEIWPCC, 2010).

There are three general conclusions about the tendency of P to precipitate with selected ions (Reddy *et al*, 1995; Scholz, 2006):

1. In acid soils, P is fixed as aluminum and iron phosphates;
2. In alkaline soils, P is bound by calcium and magnesium; and
3. The bioavailability of P is greatest at neutral to slightly acid pH.

2.7.3.2.3 Plant Uptake

Plants take up nutrients by using their root system and the estimate of net annual P uptake by emergent wetland plants ranges between 1.8 and 18 g P/m^2/y. The uptake and release of P occur similarly as that of the microbes, but the reactions require long period of time, possibly months to years. Uptake occurs during the growth phase of the plant and release occurs during plant senescence and death, followed by decomposition (US EPA, 2000).

Macrophytes and algae directly utilize only free orthophosphate form of P, which represents a major link between organic and inorganic P cycling in wetlands (Vymazal, 1995). Organically bound P constitutes 30 to 50% of the total P in most soils but it may range from as low as 5% to as high as 95% (Paul and Clark, 1996). Vymazal (2010) revealed that 30 to 70% of all the phosphate exists in an organic form in agricultural soils. Plants remove orthophosphate and release it as organic P in deposited plant litter, where orthophosphate is resulted as decaying proceeds (UNHSC and NEIWPCC, 2003).

The removal rate of P in vegetated wetlands is higher than the rate in wetlands with no vegetation (Menon and Holland, 2013). Adhikari *et al* (2011) states that P concentration was higher in the belowground parts of wetland plants, which suggests that harvest of the root system would be necessary for achieving maximum P removal (Adhikari *et al*, 2011). Assimilation of P in vegetation is usually short-term and decomposition of detrital plant tissue is usually rapid resulting in release of P. However, the undecomposed organic P accumulates in the system and becomes an integral part of the soil/sediment P pool (Reddy *et al*, 1995).

2.7.3.2.4 Wetland Phosphorus Storage

Compounds of P are a significant fraction of the dry weight of wetland plants, detritus, microbes, wildlife, and soils, although they are about ten times less than N compounds. The mass of these P storages varies in different wetland types, and with the season of the year. A general idea of the relative size of these various storage compartments is necessary to understand the P fluxs. Wetland soils and sediments contain the highest proportion of P fraction. Plants and litter comprise most of the remainder, while microbes, algae and water contain very little mass (Kadlec and Wallace, 2009).

Johannesson *et al* (2011) reported that the stores of P in the upper sediment layer of the wetland were substantially larger than the annual load of P to the wetland, and there is a risk for future release of P from those pools. Harvesting the emergent vegetation would be one way to reduce the risk for redox-induced release of soluble P and also to remove P from the wetland.

2.7.3.2.5 Microbial Phosphorus Removal

Uptake of phosphates by microorganisms, such as bacteria, algae, and duckweed, functions as a short-term, rapid-cycling means for soluble and insoluble forms. But most of the phosphate is returned back into the water column by cycling through the growth, death, and decomposition process. Some phosphate is lost in the process due to long-term accretion in newly formed sediments (US EPA, 2000).

Therefore, microbial communities of wetland soils are important both for the decomposition of organic material, and remobilization and cycling of nutrients (Prenger and Reddy, 2004). Reddy *et al* (2002) found that approximately 15–25% of the organic P in phosphorus treatment wetland soils and flocs was microbial. Some estimates place the proportion of P uptake by microflora and microfauna at about 50% (Richardson, 1985). As the life cycle of these small organisms is short, turnover is quick and it is likely that most of the uptake is returned as DOP and PP (Kadlec and Wallace, 2009). Since a larger percentage of P is in residual form, biological treatment could also take place at the bottom soil layer of the CW; however, the separation process is still limited because of the nature of P that exists with organic compounds (Choi *et al*, 2012).

Microbial uptake is very fast, but the amount stored is very low. The rate of uptake by micro-biota such as bacteria, fungi, algae, micro-invertebrates is rapid as the rate of growth and multiplication is very high. P uptake by micro-biota occurs on a time scale less than 1 hour. However, more than 90% was released within 6 hours (Vymazal, 2010; Chazarenc *et al*, 2015).

2.7.4 Fecal Coliform Removal

Many of the enteric bacteria and viruses cannot survive long once they are out of the host organisms. As a result, they start to die out and are removed by natural processes/hostile environment in the treatment wetland systems (Thomas and William, 2001). The die-off rates of fecal coliforms (FC) in the water and sediment were 0.256 log10 day^{-1} and 0.151 log10 day^{-1}, respectively (Karim *et al*, 2004; Russo, 2008). The other most important factors which play a crucial role in the removal of bacteria and viruses in natural systems are competition, predation, sedimentation, filtration, adsorption, pH extremes, and photolysis. The common groups of organisms that are used as fecal contamination indicators of surface waters are TC and FC (Thomas and William, 2001). FC is the most common indicator group in assessing water quality while the performance of wetlands is high for this group (Kadlec and Wallace, 2009).

Constructed wetlands were implemented to treat different types of municipal wastewaters with examples that include effective secondary and tertiary applications for the removal of pathogens such as FCs (Raymundo, 2008). Mahenge (2014) pointed out that there is a significant reduction in concentration of FCs in treatment wetlands and the removal rate of FCs reaches 98%. But adequate time (> 5-15 days) is required to allow the system to operate more in a steady state conditions for treatment of sewage to acceptable levels. In general, treatment wetlands show considerable potential for removing fecal bacteria from domestic wastewater (Sleytr *et al,* 2007; Fountoulakis *et al*, 2009; Vallejos *et al,* 2015).

Macrophytes-based systems turned out to be a good alternative for wastewater treatment concerning bacterial removal and water quality. In contrast, those systems without plants show lower efficiencies than their corresponding planted wetlands. It is also found that mean removal efficiencies and surface removal rates turn out to be significantly high in wetlands, and some increases in removal efficiencies are associated with warm season (Garcia *et al,* 2008; Foladori *et al*, 2015; Wu *et al*, 2016). A wetland tends toward a better performance during the maturity period reached by the system, noticeable through the presence of well-developed macrophytes (Zurita and Carreon-Alvarez, 2015).

According to Sharma and Brighu (2016), the major removal mechanism of microbes occurs due to the release of antimicrobial extract, especially from the rhizomatic part of the plant *Canna indica*. Moreover, increased surface area facilitated by increased and fibrous roots may help to result into higher filtration and adsorption mechanism of microbial removal.

The removal of FCs by using treatment wetlands is increased when the following conditions are met (Kadlec and Wallace, 2009).

- Longer nominal hydraulic residence time (t) or lower hydraulic loading
- Finer bed materials (sand), but only to the extent that the fine bed media does not impair hydraulic performance
- Warmer water temperatures
- Shallower bed depths

Tuncsiper *et al* (2012) revealed that the HRT and the loading rates are two of the most important factors in removing coliforms although the rate can be affected by a number of other conditions and environmental factors.

In tropical and subtropical climates, it is possible to remove harmful pathogenic organisms and to produce disinfected reclaimed wastewater without using expensive disinfectants, in poor areas where the reclamation of raw wastewater in agriculture is endangering human health (Zurita and Carreon-Alvarez, 2015). Hence, application of wetland system is especially suitable for small communities in developing countries, where the potential health benefits from pathogen removal are considerable (Shutes, 2001).

2.8 Reaction Kinetics

Envisaging the performance of treatment wetlands is based on the theory that the systems act as plug-flow reactors or attached-growth biological reactors, through which the wastewater flows in lock step. Plug flow evidently provides a more suitable description of the pattern of water flow in CWs. The model is first order in the forward direction while the reverse direction is zero order. Therefore, the removal performance equation can be described by employing first order plug flow kinetics (Kadlec and Knight, 1996; Thomas and William, 2001; Kadlec and Wallace, 2009; Azni *et al*, 2010).

Similarly, the removal of BOD_5 in SSF CW system can be explained with first-order plug flow kinetics and the soluble BOD_5 is removed as a result of microbial growth attached to the plant roots, stems, leaf litters and substrates. The removal rate of a particular contaminant is directly proportional to the remaining concentration at any point within the wetland bed and it is known as first-order kinetics (Kadlec and Knight, 1996; Kayombo, 2003).

The two idealized mixing theories that can be applied in first order kinetics are:

- Completely mixed reactor - the concentration is the same as the effluent concentration at any point in the reactor;
- Plug flow - the reactant concentration decreases along the length of the flow path through the reactor.

A number of individual processes such as mass transport, sedimentation, volatilization, and sorption that take place in CWs are therefore mainly first-order. So, it is rational to deduce that the processes can behave in a similar manner in combination, at least over some range of pollutant concentration. The local removal rate equation (Kadlec and Wallace, 2009) is:

$$J = k\,(C\text{-}C^{*}) \qquad \text{Eqn (2.12)}$$

Where:

J = removal per unit area, $g/m^2.d$

K= rate coefficient, m/d

C = concentration, g/m^3

C* = background constituent concentration, (g/m^3)

This rate equation is the most prevalent in treatment wetland literature. Then, combining the basic equation for a plug-flow model with the water mass balance, an exponential relation between inlet and outlet concentrations can be described by integration of the previous equation (Kadlec and Knight, 1996):

$$K = \frac{Q}{A} \ln \frac{(Ci-C*)}{(Ce-C*)} \qquad \text{Eqn. (2.13)}$$

Where:

As = surface area of a wetland (m^2)

Q = input discharge to a wetland (m^3/day) = $1.05m^3$

K= hydraulic loading rate (m/day) = 34 m/yr

C_i = inlet concentration (mg/L) = 200mg/L

C_e = outlet concentration (mg/L) =25mg/L, and

C* = background concentration (mg/L)

To overcome variability as a result of short term variation of inflow, areal removal rate constant is derived using time-averaged data. Despite this, Alley *et al* (2013) described that location and targeted constituent of wastewater can affect seasonal removal in a CW system and therefore seasonal factors such as temperature can be included in design features to maintain or enhance removal of targeted constituents.

The effluent concentration of targeted constituents is lower during the warm period as the performance is higher during this period compared to the winter period (Prochaska *et al*, 2007; Mietto *et al*, 2015). Wu *et al* (2014) pointed out that the operation of CWs at cold climate is a challenge, and consequently various adaptations are initiated through specific design and enhanced operation strategy. The variability among different systems is thus a fact for treatment wetlands which are closely related to their climate and surrounding environment (Kadlec and Wallace, 2009). In general, the effect of temperature on areal removal rate constant (*k*) can be modeled as a modified Arrhenis equation as:

$$k_T = k_{20}\, \Theta^{(T-20)} \qquad \text{Eqn. (2.14)}$$

Where:

k_T = rate constant at temperature T, d^{-1}

k_{20} = rate constant at 20 ^oC, d^{-1}

T = water temperature, ^oC

Θ = modified Arrhenius temperature factor, dimensionless

Where, k_T is the rate constant at temperature $T = T$°C and k_{20} is the rate constant at 20°C. Values of the temperature correction factor (θ) are estimated for data sets with adequate operational temperature data (Kadlec and Wallace, 2009).

Table 2-4: Kadlec and Knight K-C* model design parameters (Kadlec and Knight, 1996)

Parameters	**$K_{A,20}$**	**ϴ**	**C* (mg/L)**
BOD	34	1.00	3.5+0.053 C_i
TSS	1000	1.00	5.1+0.16 C_i
Organic-N	17	1.05	1.5
TN	22	1.05	1.5
TP	12	1.00	0.02
FC	75	1.00	300 cfu/100mL

Source: Treatment wetlands, by Kadlec and Knight, 1996, p. 217.

2.9 Application of Constructed Wetlands

Constructed wetlands with different designs have long been used primarily for the treatment of municipal or domestic wastewaters. However, because of the unique advantages of lower costs and additional benefits, the application of CWs is getting more attention and they are evolved into a dependable wastewater treatment system for the removal of a wide range of pollutants from a number of wastewater types during the last couple of decades of development. So, they are presently used for a wide variety of pollution, including agricultural and industrial wastewaters, various runoff waters, and landfill leachate (Vymazal, 2009; Wu *et al*, 2015b; Vymazal, 2014a; Vymazal and Kropfelova, 2009). Accordingly, the current literature was reviewed and the review on the applicability of constructed wetlands for various types of wastewater is presented in the following sections.

Table 2-5: Application of constructed wetlands for the treatment of different wastewater.

Type of wastewater	**Level of application**	**Location**	**References**
Pesticide polluted wastewater	Operational	South Africa	Schulz and Peall, 2001
Domestic wastewater	Experimental	Turkey	Korkusuz *et al*, 2004
Industrial wastewater	Experimental	Taiwan	Chen *et al*, 2006
Domestic wastewater	Experemintal	USA	Prochaska *et al*, 2007
Farmyard runoff	Operational	Ireland	Mustafa *et al*, 2009
Industrial wastewater	Operational	USA	Knox *et al*, 2010
Animal farm wastewater	Experimental	Ireland	Babatunde *et al*, 2011
Sugar factory wastewater	Experimental	Kenya	Odinga *et al*, 2011
Swine wastewater	Operational	Brazil	Sarmento *et al*, 2012
Tannery wastewater	Experimental	Bangladesh	Saeed *et al*, 2012
Textile wastewater	Experimental	India	Sivakumar *et al*, 2013
Winery wastewater	Operational	Spain	Varga *et al*, 2013
Pulp and paper mill wastewater	Experimental	India	Choudhary *et al*, 2011
Domestic wastewater	Operational	Tanzania	Mahenge, 2014
Acid mine drainage	Experimental	South Africa	Seadira *et al*, 2014
Storm water	Operational	Australia	Mangangka *et al*, 2015
Agricultural and urban runoff	Operational	USA	Pietro and Ivanoff, 2015
Diesel polluted wastewater	Experimental	Malaysia	Al-Baldawi *et al*, 2013
Landfill leachate	Experimental	Colombia	Madera-Parra *et al*, 2015
Food processing wastewater	Operational	France	Paing *et al*, 2015b
Dairy farm wastewater	Operational	USA	Tuncsiper *et al*, 2015
Municipal wastewater	Operational	Central Jordan	Albalawneh *et al*, 2016
Landfill leachate and domestic wastewater	Experimental	Iran	Mojiri *et al*, 2016
Leachate	Experimental	Iran	Bakhshoodeh *et al*, 2016

2.9.1 Application of Constructed Wetlands for Domestic Wastewater Treatment

Like many other treatment technologies, treatment wetlands were employed primarily for the treatment of wastewater from human dwellings and activities. The majority of applications of CW as a treatment method were associated with municipal and domestic wastewater, and the technology is still growing rapidly in many areas (Russo, 2008; Kadlec and Wallace, 2009). Many such systems are currently in use around the world, designed to treat domestic wastewater (Trivedy and Siddharth, 2010).

The number of CWs in use has very much increased in the most recent years. The use of CWs in the United States, New Zealand and Australia is gaining rapid interest. The systems are mainly used in towns and cities of those countries for tertiary treatment to remove nutrients of low concentration. However, CWs are usually used in European countries as a method of secondary treatment for domestic of village populations (Russo, 2008).

Likewise, the enormous potential for large-scale treatment and high demand for clean water in the tropical and subtropical areas become the driving forces for many developing countries in those locations to use the technology in order to solve pollution problems. Unfortunately, the available literatures regarding the application of CWs in those locations are comparatively low. It becomes visible that in some countries, basic researches are being conducted, while the technology reached pilot and full-scale levels for various applications in other countries (Kivasi, 2001; Zhang *et al*, 2015).

In recent times, researches related to the performance of CWs in treating domestic wastewater were carried out in developing countries such as Tanzania (Mashauri *et al*, 2000, Mahenge, 2014), Uganda (Kyambadde *et al*, 2004; Kyambadde *et al*, 2005), Malaysia (Katayon *et al*, 2008), Thailand (Konnerup *et al*, 2009), Kenya (Kelvin and Tole, 2011), Cameroon (Fonkou Theophile *et al*, 2011), Egypt (Abou-Elela and Hellal, 2012; Abou-Elala *et al*, 2014; Abdelhakeem *et al*, 2016), Taiwan (Hsueh *et al*, 2014) Morocco (Laffat *et al*, 2015) and Pakistan (Sehar *et al*, 2015).

Mashauri *et al* (2000) carried out a research on the performance of a horizontal flow constructed wetland applied for the treatment of wastewater effluent from waste stabilization ponds at Dar es Salaam University, Tanzania. The efficiency of the reed bed treatment system at low filtration rate (0.27 m/h) was 80% for TSS, 66% for COD, 90% for TC and 91% for FC. Similarly, in Uganda, Kyambadde *et al* (2005) assessed the feasibility of nutrient removal from wastewater using horizontal flow CWs. The wetland system was substrate-free and planted with two tropical plants: *Cyprus papyrus* and *Miscanthidium* *violaceum*. Accordingly, Results showed that the removal efficiencies for BOD, NH_4–N and TP fractions in *papyrus*-based CWs were 86.5%, 68.6%, and 69.7%, respectively, while for *Miscanthidium*-based system the values were 53.2%, 45.5%, and 46.7%, respectively. Therefore, wetlands planted with *Cyprus Papyrus* showed higher removal efficiency than *Miscanthidium* based system.

In Malaysia, three SSF CWs (two planted with *Lepironia articulate* and one unplanted cell), operated at four different retention time, were tested to evaluate the performance in treating a mild domestic wastewater. The CWs were able to remove about 56–77% of COD, 50–88% of TSS, 20–88% of TP, 27–96% of NH_4^+ and 99% of total coliforms. The removal rates of COD, TP, and NH_4^+ were affected by different hydraulic retention times, but the rates for TSS and TC were not affected by the retention time (Katayon *et al*, 2008).

Konnerup *et al* (2009) conducted a study on eight horizontal SSF constructed wetlands planted with *Canna* and *Helichonia* in Thailand. The system was designed to treat domestic wastewater at four different hydraulic loading rates: 55mm d^{-1}, 110mm d^{-1}, 220 mm d^{-1}, 440 mm d^{-1}. The result showed that the rates of mass removal for TSS, COD, TN and TP varied between 88-96%, 42-83%, 4-37%, and 6-35%, respectively, depending on the loading rates. Although the removal of TN in beds planted with *Canna* was higher than beds planted with *Helichonia*, the removals of both TN and TP were low in the pilot-scale wetlands. In another study done in Kenya by Kelvin and Tole (2011), the performance of the tropical constructed wetland was evaluated and comparison with the conventional treatment methods was made. In this study, the tropical CWs achieved the removal efficiency of 96.2% for BOD, 97.6% for COD, 21.4% for TP, 41% for ammonia and 99.99% for FC. Hence, the tropical CW system was found to perform much better than the commonly used waste stabilization ponds.

The performance of *Cyprus papyrus* in HSSF and HSF constructed wetlands for the treatment of domestic wastewater was evaluated in a study carried out in Cameroon (Fonkou *et al*, 2011). From the study, the reductions of several physico-chemical parameters and FC in vegetated systems were not significantly different as compared with the non vegetated wetland system (Fonkou *et al*, 2011). In Egypt, Abou-Elela and Hellal, (2012) tested a pilot scale vertical flow constructed wetland for treating primary treated municipal wastewater. The wetland unit was employed with the surface area of 457.56m^2 and influent flow rate of 20m^3/day. The average removal efficiencies for BOD, COD, and TSS were 90%, 88% and 92%, respectively.

2.9.2 Application of Constructed Wetlands for Storm Water Treatment

Urbanization creates large impervious areas that increase the quantity and peak rate of runoff. Rainfall then washes deposited materials directly into surface waters, causing stream pollution (Robert, 2001). The composition of storm water varies greatly, depending on the surrounding land use. For example, urban runoff may contain soil particles, dissolved nutrients, heavy metals, oil, and grease. Residential and agricultural runoff may also contain organic matter and pesticides (US EPA, 2000; Thomas and William, 2001). Pollutant concentrations and loads generally range from low levels from undeveloped and park lands, to low density residential and commercial, to higher density residential and commercial, and finally to high density commercial and industrial land uses (Kadlec and Wallace, 2009).

The application of CWs for treatment of combined sewer overflow (CSO) is considered a sustainable approach with relevant potential and it can be interesting in an urban context (Amaral *et al*, 2013). The use of wetland retention basins for treatment of storm water runoff becomes relatively common (Thomas and William, 2001; Mangangka *et al*, 2015), and FWS wetlands are the nearly exclusive choice for the treatment of urban, agricultural, and industrial storm waters, because of their ability to deal with pulse flows and changing water levels. In contrast to other applications, there is basically no pretreatment for urban stormwaters, if the forebay settling basin is considered part of the wetland (Kadlec and Wallace, 2009).

Gan (2004) stated that wetlands are an effective storm water treatment measure for the removal of fine SSs and associated contaminants, as well as soluble contaminants. These systems utilise a combination of physical, chemical and biological processes in removing storm-water pollutants. They are used as “end-of-pipe” or at “source control measures”. Mangangka *et al* (2015) highlighted the importance of ensuring that the inflow into the wetland has low turbulence in order to achieve consistent treatment performance for both, small and large rainfall events.

2.9.3 Constructed Wetlands for Industrial Wastewater Treatment

Industrial wastewater is here a loosely defined category, including wastewaters that are not from domestic, municipal, animal, or food product processing (Kadlec and Wallace, 2009). Constructed wetlands were increasingly applied in treating various industrial wastewaters with specific characteristics (Al-Baldawi *et al*, 2013; Vymazal, 2014a; Wu *et al*, 2015). A reliable and stable effluent can be obtained using the CW system for treating industrial wastewater treatment (Chen *et al*, 2006).

Vymazal (2014) revealed that all types of CWs were used with most systems being either free water surface CWs with emergent vegetation or horizontal subsurface flow CWs. The use of vertical flow CWs for treatment of industrial wastewaters has been less frequent so far. However, vertical flow CWs were successfully used for treatment of olive mills wastewaters and also in hybrid CWs. The treatment technology of CWs has evolved in a reliable technology which is nowadays successfully used for many types of industrial effluents.

The application of CWs system for the treatment of various types of industrial wastewaters is listed below.

- Acid mine drainage (Johnson and Hallberg, 2005; Seadira *et al*, 2014)
- Control of landfill lecheat (Madera-Parra *et al*, 2015; Mojiri *et al*, 2016)
- Electroplating wastewater (Sudarsan *et al*, 2015)
- Industrial park wastewater (Chen *et al*, 2006)
- Tannery wastewater (Leta *et al*, 2004)
- Textile wastewater (Sivakumar *et al*, 2013)
- Refinery effluent (Robert and Kadlec, 1996)
- Pulp and paper wastewater (Choudhary *et al*, 2011)
- Winery wastewater (Rozema *et al*, 2016)

2.9.4 Application of Constructed Wetlands for Agro-Industrial Wastewater Treatment

Agro-industrial wastewaters can be very strong in terms of pollutant concentrations and hence can contribute significantly to the overall pollution load imposed on the environment. Examples of agro-industrial wastewaters include those arising from industrial-scale animal husbandry, slaughterhouses, fisheries, and seed oil processing (Jern, 2006).

It has, however, been noted that wastewaters with COD: BOD_5 ratios of 3 or lower can usually be successfully treated with biological processes. COD: BOD_5 ratios of 3 or lower are encountered in many of the agricultural and agro-industrial wastewaters. But agro-industrial wastewaters need not always have such low COD: BOD_5 ratios. For instance, tobacco processing wastewater can have a COD: BOD_5 ratio of about 6:1. This is a strong wastewater which can be difficult to treat to meet COD discharge limits because residual organics following biological treatment are resistant to further biological treatment (Jern, 2006).

Wastewater from the dairy industries that does not receive some form of biological pre-treatment contain almost twice the content of soluble organics as a percentage of the total load compared with municipal wastewater (Milne and Gray, 2013). Hu *et al*, (2011) described in his study that significant variety of livestock wastewater in biodegradability during long-time storage was observed and the laboratory results showed that fresh livestock wastewater was readily biodegradable, while it turned to be non-biodegradable after long-time storage.

Despite of these, CWs efficiently reduced BOD_5 and TSS in dairy effluent. The BOD_5 and TSS reduction efficiencies were significantly greater during the best growing seasons of plants and the best seasons of microbiological activity. CWs have the potential as a recommendable practice for the treatment of BOD_5 and TSS contained in dairy farm effluents under cold climate conditions. The BOD_5 and the TSS treatments by CWs were enhanced by connecting two CWs in-series (Tuncsiper *et al*, 2015). Cortes-Esquivel *et al* (2012) also indicated that CWs can be a very useful tool for the removal of heavy metals like Zn and Cu in swine wastewater. Similarly, wetlands may be used effectively for treatment of animal and aquaculture wastes (Thomas and William, 2001).

2.9.5 Constructed Wetlands for Leachate Treatment

Treatment and disposal of liquid leachates is one of the most difficult problems associated with the use of sanitary landfills for disposal of solid waste. Leachate is produced when rains fall and percolate groundwater combine with inorganic and organic degraded waste. The highly variable nature of solid waste, differences in age and decomposition, and the diversity of chemical and biological reactions that take place in landfills result in a wide range of chemical quality of leachates (Mulamoottil *et al.*, 1998) (Kadlec and Wallace, 2009).

Madera-Parra *et al* (2015) revealed that CWs effectively remove COD, BOD_5 and nutrients (TKN, NH_4^+–N, PO_4^{3-}-P) from pretreated landfill leachate and except for NH_4^+–N achieved concentrations below the provisional standard. Additionally, they attain reduction levels similar to those obtained with highly mechanized systems. Hence, the development of these types of CWs at a full scale is an attractive technology for landfill leachate treatment in countries with low resources and high necessities to protect the environment and public health.

It is reported that the levels of particular contaminants in municipal landfill leachate exceed the allowable discharge restrictions for colour, COD, ammonia, Ni, and Cd. Pollutants from landfill leachate wastewater can be removed by using a CW system. CWs show removal efficiency of 90.3%, 86.7%, 99.1%, 86.0%, and 87.1% for colour, COD, ammonia, Ni, and Cd, respectively. Removal efficiencies decrease as leachate ratio in the leachate and wastewater mixture increased (Mojiri *et al,* 2016; Bakhshoodeh *et al*, 2016).

2.9.6 Constructed Wetlands for Acid Mine Drainage Treatment

The major producer of acid mine drainage is the mining industry. Waters draining active and, in particular, abandoned mines and mine wastes are often net acidic. Such waters typically pose an additional risk to the environment by the fact that they often contain elevated concentrations of metals (iron, aluminium and manganese, and possibly other heavy metals and metalloids of which arsenic is generally of greatest concern) (Johnson and Hallberg, 2005).

Treatment wetlands typically operate at neutral pH for influents that are not strong acids or bases. This is true for both FWS and SSF CWs, but low influent pH levels are the norm in case of acid mine drainage. Some HSSF wetlands are designed with reactive media in the bed material such as zeolite that can produce high pH effluents. Although CWs often operate at pH 6.5, acid mine drainage wetlands function with incoming pH less than 5, which is commonly regarded as a lower limit for aquatic resource protection (Kadlec and Wallace, 2009).

Microbiological processes that generate net alkalinity are mostly reductive processes and include denitrification, methanogenesis, sulfate reduction, and iron and manganese reduction. Ammonification is also an alkali-generating process. Due to the relative scarcity of the necessary materials (e.g., nitrate), some of these processes tend to be of minor importance in acid mine drainage-impacted environments. However, in as much as both ferric iron and sulfate tend to be highly abundant in acid mine drainage, alkali genesis resulting from the reduction of these two species has a potentially major significance in acid mine drainage-impacted waters. Photosynthetic microorganisms, by consuming a weak base (bicarbonate) and producing a strong base (hydroxyl ions), also generate net alkalinity (Johnson and Hallberg, 2005)

2.9.7 Constructed Wetlands for Agricultural Runoff Treatment

The use of CWs to improve the quality of wastewater or the water from mining exploitations and agriculture is a technology under way of development (Muresan, 2012).Concurrent high removal rate of

COD, ammonia, and phosphorous can be obtained in a two-stage CW system, demonstrating its potential use for cost-effective reduction of pollution load of agricultural wastewaters (Babatunde *et al*, 2011).

The efficiency of a FWSF CW in treating agricultural discharges was investigated during storm and non-storm events. Overall, the results indicated that the design of the CW system could feasibly function for the retention of typical non-point source pollutants like suspended solids, excess nutrients and organic matters. The hydraulic fluctuations and increase in pollutant concentrations during storm events made the system more efficient in addition to the moderate temperature (greater than 15^0C) during the storm seasons. Although the overall mass was not removed, the levels of pollutants were reduced to appreciable levels. More importantly, CWs contributed to the improvement of stream water quality thereby reducing the potential impact of pollutants downstream (Maniquiz *et al*, 2012).

The long-term assessment demonstrates that the example integrated CWs system can be considered an effective and sustainable wastewater treatment option for agricultural runoff rich in nutrients (Mustafa *et al*, 2009). CWs should be considered an option for current agricultural wastewater applications. Many have adopted this view, and, as a result, there are a large amount of full scale, functional CWs found throughout North America, and the world, that are being used to treat various types of wastewater. There is no one CW design that is the most effective for agricultural wastewater, but, rather, each design has strengths and weaknesses so hybrid designs may prove to be the most practical (Rozema *et al*, 2016).

Scholz *et al* (2010) described that integrated constructed wetlands (ICW) are capable of treating farmyard dirty water and that they provided a sustainable management option to effectively reduce nutrient and contaminant loss from farmyards to watercourses. Significant concentration reductions in suspended organic matter, nutrients and faecal bacteria between ICW influents and effluents were observed. Surface discharges from the ICW sites had seasonal patterns. None of the farm ICW had surface discharges during most summer months.

To treat low C/N ratio wastewaters, such as nitrate-rich agricultural runoff and polluted groundwater, the carbon source only from the root exudates of macrophytes is not sufficient to maintain a high performance of nitrate removal. Denitrification can be enhanced by the external supply of electron donors via direct organic carbon addition using organic filtration media and/or step feeding operation. However, the potential secondary pollution should be considered. The promotion of autotrophic denitrification, especially via the pathway of microbial anammox, could be a potential promising strategy (Wu *et al*, 2014).

2.9.8 Constructed Wetlands for Pesticide Treatment

CWs have become the best management practice for pesticide mitigation from non-point source agricultural runoff and drainage in many countries. So far, CWs with free water surface have been used mostly, while the SSF CWs have recently been adopted as well. As both aerobic and anaerobic processes are involved in pesticide removal, hybrid CWs may offer efficient solution. Current survey indicated that removal of pesticides is generally effective, but the efficiency varies widely among pesticides and also among systems for a particular pesticide (Vymazal and Brezinova, 2015).

There are many processes which are responsible for pesticide mitigation such as hydrolysis, photolysis, sedimentation, adsorption, microbial degradation or plant uptake, however, the extent of these processes depends on local conditions, and it is difficult to single out the most important ones (Vymazal and Brezinova, 2015).

There is strong evidence to suggest that the presence of vegetation enhances pesticide retention. The results of the survey revealed that the highest pesticide removal was achieved for pesticides of the organochlorine, strobilurin/strobin, organosphosphate and pyrethroid groups while the lowest removals were observed for pesticides of the triazinone, aryloxyalkanoic acid and urea groups (Vymazal and Brezinova, 2015).

Organophosphorus pesticides were found in the outlet suspended-particle samples, highlighting the retention capability of the wetland. A toxicological evaluation employing a Chironomus bioassay in situ at the wetland inlet and outlet revealed an 89% reduction in toxicity below the wetland during runoff (Schulz and Peall, 2001).

Macrophyte vegetated wetlands have the potential to contribute to aqueous-phase pesticide risk-mitigation. It can be concluded that the conservation and management of vegetation in small drainage channels may be an effective tool to avoid agricultural pesticide contamination of larger receiving water bodies (Schulz *et al*, 2003).

A long water residence time improves the effectiveness of CWs, favouring relatively fast processes, such as sedimentation and pesticide sorption, and subsequently removing pesticides from the water phase.

Thus, these processes result in pesticide storage in the solid phases and can reduce the effectiveness of CWs by saturating the sorption sites. Pesticides can be degraded from the solid phase in CWs during periods without water flow, which reduces their accumulation and risks decreasing their effectiveness over time. In addition, the effectiveness of CWs may be improved by increasing the water residence time (Romain *et al*, 2015).

2.10 Studies on Performance of Constructed Wetlands in Ethiopia

Even though the potential of the application of CWs for treating various wastewater types is tremendous, they are not commonly used in Ethiopia. Only few institutions were employed operational CWs for treating domestic wastewater, and obviously some other industries such as tanneries, breweries, and coffee mills are starting to implement the system in recent times (Birhanu, 2007; Asaye, 2009; Kenatu, 2011; Connie, 2012).

In the mean time, many studies to evaluate the performance of pilot-scale and in few cases full-scale CWs for different wastewaters were carried out. Birhanu (2007) conducted a study to evaluate the removal performance of horizontal SSF CWs system constructed at JWBO, for treating domestic wastewater. Based on the results of his study, the average removal efficiency of the treatment system were 99.3% for BOD_5, 89% for COD, 85% for TSS, 28.1% for $NH4^+$-N, 64% for $NO3^-$-N, 61.5% for TN, 28% for PO_4^+, 22.7% for TP, 77.3% for SO_4^+, 99% for S^{2-}, 94.5% for TC and 93.1% for FC. The result also mentioned that wetland beds planted with *Cyprus papyrus* showed higher removal efficiency for NO_3^--N, NH_4^+-N, TN, PO_4^{3-}, and TSS than the other wetland cells, while beds planted with *Phoenix canariensis* showed higher removal efficiency for TP, S^{2-}, BOD_5, COD, TC and FC.

Asaye (2009) conducted a study to evaluate selected plant species (*Cyprus papyrus*, *Typha domingensis*, *Cyperus alopcuroides*, *Schenoplectus corymbosus*, *Sesbania sesban*, *Aeschynomene elaphroxylon*) for the treatment of tannery wastewater. Accordingly, he reported that the wetland cell planted with *Cyperus Papyrus* showed high removal efficiencies for NO_3^-(73.2%) and NH_4^+ (26.2%). High removal efficiencies for total Cr (98.4%), COD (68.7%) and S^{2-} (59.2%) resulted in the cell planted with *Schenoplectus corymbosus,* cell cell planted with *Sesbania sesban* showed removal efficiencies for SO_4^{2-} (96.3%), BOD_5 (84.7%) and TN (58.3%).

In another experimental study, Kenatu (2011) carried out a study to evaluate the performance of SSF constructed wetlands planted with *Canna indica, Phragmite karka, and colocacia gigantean* for the treatment of wastewater from Breweries. The result showed that the average removal efficiencies of planted CWs for BOD_5, COD, TN, NH_4^+-N, NO_3-N, TP, PH_4^{3-}, SO_4^{2-}, S^{2-}, TSS, TDS, and EC were: 74.4%, 78.9%, 77.4%, 62.8%, 55.2%, 68 %, 81.4%, 36.1%, 97.3%, 61.7%, 54.1%, and 27.4%, respectively. She also reported that the values of some of the effluent parameters complied with the provisional emission standard limits.

Similarly, the performance of HSSF CWs planted with *Phragmites australis* in removing heavy metals from landfill leachate was evaluated. The experimental study revealed that the removal efficiencies were *99.33%* for *Fe*, *93.67%* for *Mn*, *89.24%* for *Pb*, *96.14%* for *Cu* and *98.33%* for *Zn.* The result also showed that heavy metal uptake by the root system is higher than the uptake by the stem and leave (Mesele, 2013).

Likewise, in another study, Kassa and Mengistou (2014) evaluated nutrient uptake efficiency and growth of *Cyperus papyrus* and *Phragmites karaka*. The results of the study showed that *Cyperus papyrus* showed higher rate of biomass accumulation as evidenced by increase in shoot and root weights (83.93 gm) compared to *Phragmites karka*. *Cyperus papyrus* showed higher accumulation of TP in the root system and TN in leaves than *Phragmites karaka*. The mean removal efficiencies of the cell planted with *C. papyrus* were 56.37% for NO_3^--N, and 84.04% for PO_4^{3-}, while removal efficiencies were by the cell planted with *P. karaka* were 58.37% for NO_3^--N and 65.18% for PO_4^{3-}.

Most of the studies were conducted to evaluate the performace of laboratory-scale or pilot-scale HFCW systems in treating domestic or industrial wastewaters. In this study, in addition to the HFCW system, both the vertical flow and hybrid of the horizontal and vertical flow CWs, which are the state of art in recent times, were employed in Kotebe WWTP for the treatment of domestic wastewater. The performance of the three CW cells was monitored for relatively long periods of time /one year/ to check the sustainability of the removal efficiency of the system and also the seasonal performance of the systems.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Description of the Study Area

The study was conducted in Addis Ababa, the capital city of Ethiopia. Addis Ababa covers a total area of about 540 km^2. According to the 2007 census, the population of Addis Ababa is 2.738 million, of which 51.6% are females while 48.4% are males. The city lies between 2,200 and 2,500 meters above sea level and the lowest and the highest annual average temperatures are 9.89 and 24.64 °C, respectively (Dillon, 2005; AAWSA, 2002; FDREPCC, 2008). Being a tropical area, the classification of seasons in Ethiopia has virtually opposite features to temperate regions. In most cases, the period from December to February is known to be the dry season, while the period from March to May is characterized by high temperature with occasional showers. On the other hand, the time from June to August is the heavy rainy season while the time from September to November is the spring season.

According to Daniel *et al* (2010), Addis Ababa generates an estimated annual volume of 45Mm^3 domestic wastewater. However, the existing sewerage system is not adequate and access to piped sewerage system is limited to about 10% (World Bank, 2015). Most of the wastewater from housing units connected to the city's sewer system is conveyed to Kality wastewater treatment plant that has not improved since 1993 (AAWSA, 2002). About 4,500 m^3/day wastewater is transported and treated by Kality wastewater treatment plant (design capacity of 7,600m^3/d). In addition to this, about 1,200m^3/day transported by truck and treated by the drying beds.

Kotebe treatment plant is the other centralized treatment plant which was initially designed to receive and treat sludge from vacuum trucks that empty dry pit latrines and septic tanks, with annual capacity of approximately 150,000 m^3 and it consist of 20 drying beds and 10 lagoons (AAWSA, 2002). But, the treatment plant has been under continuous expansion and connected to the sewerage system to receive and treat wastewater using stabilization ponds starting from 2011. Kotebe wastewater treatment plant is located in the north-eastern outskirts of Addis Ababa. It has a capacity of 2,000 m^3 per day and serves primarily condominiums which are equivalent to about 5000 households (World Bank, 2015).

The pilot-scale subsurface flow constructed wetland systems used in this study were implemented at Kotebe WWTP and the specific area was situated at a longitude of 38°51'9.67'' E, a latitude of 8°58' 14.73'' N and an altitude of 2,266m.

Several field visits were conducted to the site to gather appropriate and detailed information before the application of the pilot scale constructed wetland systems for this study.

3.2 Construction of the wetland systems

The pilot scale constructed wetland system was located at about 165 m down to the sedimentation tank of the main sewerage system which was convenient to use gravity flow instead of using a pump system to transport the wastewater from the main sedimentation tank of Kotebe Wastewater Treatment Plant (KWWTP) to the pilot scale wetland systems without any difficulty.

The pilot scale constructed wetland systems had three parallel SSF treatment cells. Each wetland contained inlet piping, outlet piping, plastic liner to protect ground water pollution and water loss from the wetland system through infiltration, the same fill media (gravel) and emergent vegetation (*Cyprus papyrus*).

3.2.1 Determination of the size of wetland cells

The wetland system was applied at pilot scale level with a capacity of 70 population equivalents (PE). According to Daniel *et al*. (2010), a person in Addis Ababa can produce approximately 0.045 m^3 wastewater. Based on this figure, the flow rate or the calculated volume of wastewater produced by 70 PE per day was 3.15 m^3, the total volume treated by the three wetland cells each day. The size of the pilot scale constructed wetland was determined based on published first order plug flow kinetic model of Kadlec and Kright method considering the BOD removal according to the "Constructed Wetlands Manual" UN-Habitat (2008b).

$$As = \frac{Q}{K} (\ln Ci - \ln Ce) \qquad \text{Eqn (3.1)}$$

Where: A_s = surface area of a wetland (m^2)

Q = input discharge to a wetland system (m^3/day) = 3.15m^3

K = Areal removal rate (m/year) = 34m/year

C_i = inlet concentration (mg/L) = 200mg/L

C_e = outlet concentration (mg/L) =25mg/L

The total area required to treat the wastewater flow rate of 3.15m^3/d was calculated as 72m^2.

3.2.2 Hydraulic retention time of the wetland systems

The time that the domestic wastewater resides in the wetland systems was estimated using Darcy's formula (US EPA, 1993) as follows:

$$HRT = \frac{nLWd}{Qav};$$ …………………….. Darcy's law Eqn (3.2)

Where: n = effective porosity of the media, % as a decimal (0.23 - 0.38% for gravel media); 32%

L = Length of the wetland cell (m); 6m

W = Width of the wetland bed (m); 4m

d = average depth of liquid in the wetland (m); 0.55m

Qav = Average of the inflow and outflow; $\frac{(Qin + Qout)}{2}$, (m^3/d);

$= \frac{(1.05+0.85)}{2}$ (m^3/d) = 0.95m^3/d

$= \frac{0.32*6m*4m*0.55m}{0.95\ m3/d}$

$= \frac{4.224}{0.95}$ days

= 4.5 days

3.2.3 Layout and configuration of the pilot-scale constructed wetlands

The layout, configuration and other characteristics of the constructed wetland systems are shown in Figure 3.1 and summarized in Table 3.1. Regarding the configuration of the wetlands system, the pilot-scale CWs system had three subsurface flow constructed wetland (SSF CW) cells. All the three CW cells are parallel and the hybrid CW system had two cells (HFCW and VFCW) connected in series. The only difference in designing the three SSFCW was the wastewater flow pattern. The types of wastewater flows into the wetland system were continuous horizontal flow /horizontal CW/, vertical flow /vertical CW/ and hybrid of horizontal and vertical flow /hybrid CW/. The area of each wetland cell was 24m^2 with a dimension of 4m width and 6m length. In line with this, the area of the horizontal flow bed in the case of the hybrid system (having horizontal and vertical flow paths arranged in series) was chosen as $8m^2$ (2m x 4m), which is half of the area of the vertical flow wetland, $16m^2$.

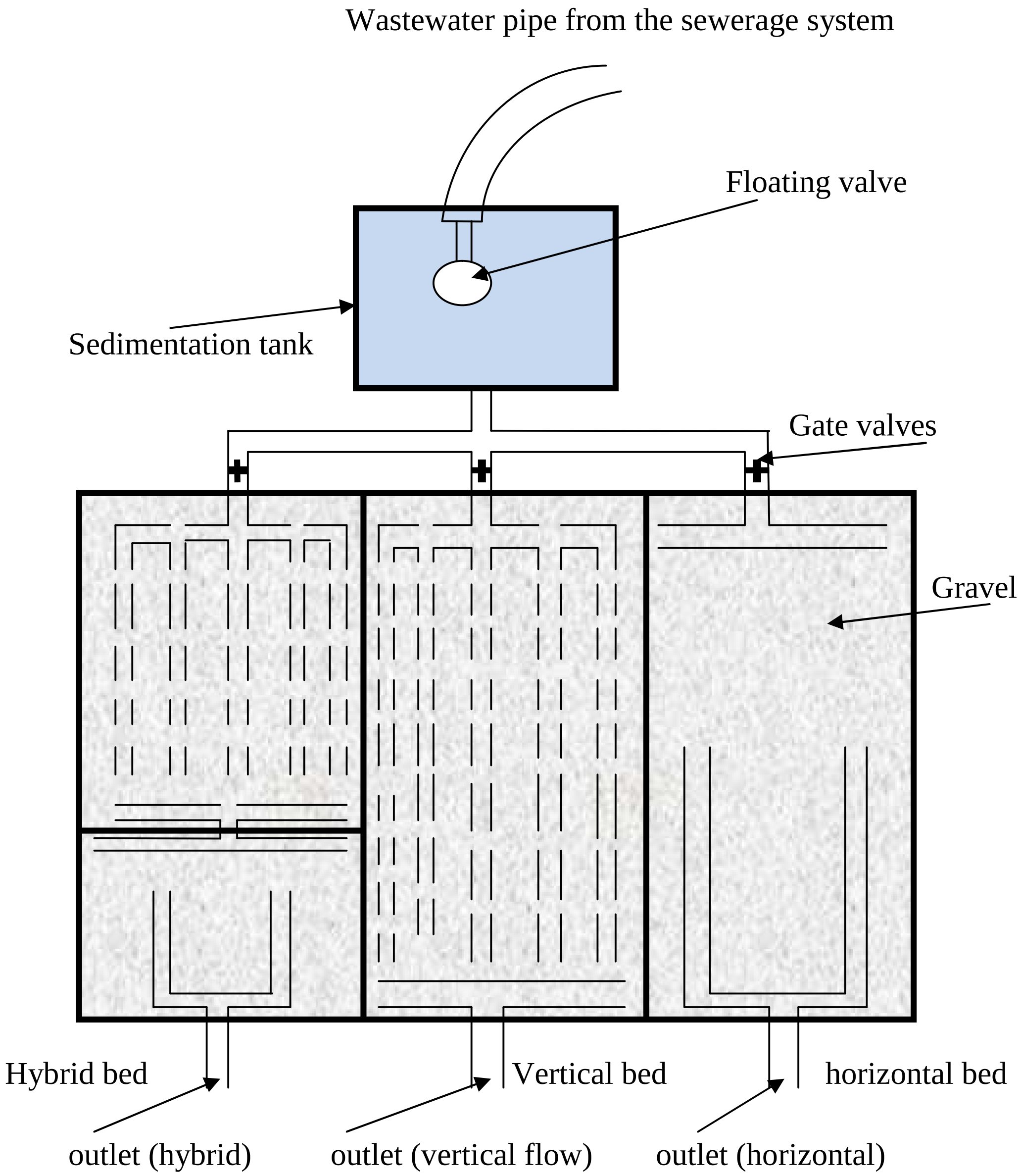

Figure 3.1: Sketch map of the configuration of the pilot scale constructed wetland systems

3.2.4 Type of filter media used

The filter media used for the pilot scale CW systems applied was gravel with different diameter. The size of the filter media was in the range of 10 and 30 mm, the recommended size by US EPA (US EPA, 2000). The effective depth of the filter media applied for the pilot-scale constructed wetlands of the study was 0.6m.

3.2.5 Plant species used in the study and planting procedures

Cyprus papyrus was the wetland plant type used in the pilot scale constructed wetlands in this study. The rhizomes of *Cyprus papyrus* with their shoots were collected from the natural stock/natural wetlands and transported to the study area in vehicles. All the three constructed wetland cells were planted at 0.5m interval between each planted rhizomes; a density of 9/nine/ rhizomes/m^2. The planting was carried out in June, the early time of the rainy season/summer/. Immediately after the completion of the plantation, the wetland beds were filled with tap water and watering of the wetland with rain water and tap water went on for six months, until the wetland plants adapted to the environment and grew well.

3.2.6 Sedimentation tank

A sedimentation tank with dimensions of 3.2 m (length), 2.2 m (width) and 0.90 m (height) was constructed 0.5 m above the surface of the wetlands to primarily treat wastewater before it entered into the wetland cells by the help of gravitional force. It had a total volume of $6.3m^3$ and an effective volume of about $4.9m^3$. The sedimentation tank was used primarily for storing wastewater and settling of solids. The raw domestic wastewater was kept for 3 - 4 hours in the sedimentaion tank. The internal wall and the bottom surface were plastered to prevent any water leakage from the tank.

A PVC pipe of 50mm diameter which fed the wetlands was connected to the tank at 10 cm above the bottom surface. The first 10 cm height from the bottom surface was left for the accumulation of wastewater sludge/settled solids and the sedimentation tank was provided with an outlet/PVC pipe at the bottom to remove the accumulated sludge. Then, the sludge was regularly removed every two months using the outlet pipe and stored in a pit prepared at 25m far from the tank.

3.2.7 Installation of inlet and outlet pipes

To transport the wastewater from the main sedimentation tank of the treatment plant to the sedimentation tank of the wetland system, High-density Polyethylene (HDPE) pipe with 2 inch diameter and 10 m length was installed to the outlet of the main sedimentation tank. Then the HDPE pipe was tightly fitted at its tip /inlet end/ with a 5L jerrican, having more or less uniform holes around its surface to allow the flow of wastewater into the pipe, but to prevent larger debris or solids from getting into the HDPE pipe. Then the pipe with large diameter was reduced to ¾ inch pipe and the wastewater was transported using gravity to the wetland system, constructed downwards at a distance of about 165m from the main sewerage system.

To regulate the wastewater flow of the wetland system, a floating valve was fitted at the edge of the HDPE pipe which fed the sedimentation tank of the pilot scale constructed wetland system. When the tank was full, to the height of about 0.8 m, the wastewater in the sedimentation tank picked the floating valve up to close the pipe and avoid overflow of the wastewater. When the level of the wastewater in the sedimentation tank was lowered, the floating valve turned down to open the HDPE pipe that fed the sedimentation tank. However, the level of wastewater in the sedimentation tank was limited to the depth of about 0.2 m during the operation period in order to ensure 3 - 4 hours of detention time.

The influent from the sedimentation tank entered into the wetland system using PVC pipe installed at 10 cm above the bottom of the sedimentation tank. It was divided into three branches using T, pipe fitting, to feed the three wetlands. The pipe line at the inlet of each of the wetland was fitted with a gatevalve to regulate the flow of wastewater. As a result, the amount of wastewater that entered into each wetland cell was adjusted using a stop watch and a graduated glass container; i.e. the flow rate was adjusted to be about 0.73 liter/min (1.05 m^3/day) for each wetland or 3.15 m^3/d for the three wetland systems. All the pipes, valves, and the wetland system in general were monitored twice per day to maintain proper wastewater flow rate and functioning of the system.

Polyvinyl Chloride (PVC) pipes of 50 mm diameter were used for the flow of wastewater into the CW system and to collect the effluent. The PVC pipes were drilled at every 25 cm distance and each hole had 7 mm diameter. This was made to ensure an equal wastewater flow distribution at the inlet of the horizontal CW and all over the surfaces of the vertical CW. Therefore, the perforated pipes were installed at the inlet region in case of the horizontal CW while they were installed to run from the inlet to the outlet direction above the surface of the vertical CW. Here, welded metallic rods and blocks were used to hold the pipes at about 20cm above the top surface in case of the vertical flow constructed wetlands.

In the horizontal flow wetlands, the wastewater was fed at the inlet and run slowly through the porous fill media under the surface of the bed in a more or less horizontal path until it reached the outlet zone. But the wastewater was fed from the top to gradually percolate down through the bed and to be collected by a drainage network at the base in case of vertical flow wetlands. Generally, PVC/perforated PVC pipes with 50 mm diameter were used to feed the treatment wetlands or to collect the effluent.

3.2.8 Lining of the wetland beds

In order to prevent contamination of wastewater infiltration, synthetic geomembrane was selected to be applied as a sealing material on the floor of the media. The bottom of each constructed wetland system was made to have 1% slope, to allow easier water circulation within the bed during the operation phase of the system.

Table 3-1: Summary of the characteristics of the pilot scale constructed wetland systems applied in Addis Ababa, Ethiopia.

Design factors	**Types of constructed wetlands**			
			Hybrid of VFCW and HFCW	
	HFCW	**VFCW**	**VFCW** *	**HFCW** *
Wastewater flow type	HF	VF	VF	HF
hydraulic loading (m^3/day)	1.05	1.05	1.05	1.05
Width (m)	4	4	4	4
Length (m)	6	6	4	2
Area (m^2)	24	24	16	8
Wetlands fill media	Gravel	Gravel	Gravel	Gravel
Depth of fill media (m)	0.6	0.6	0.6	0.6
Hydraulic loading rate (m/d)	0.044	0.044	0.066	0.131
Wetland plants	*C. papyrus*	*C. papyrus*	*C. papyrus*	*C. papyrus*

* Refers to the component cells of the hybrid wetland bed constructed in series.

3.3 Monitoring of performance (removal efficiency) of the constructed wetland systems

The performance of the horizontal, vertical and the hybrid of both horizontal and vertical subsurface flow constructed wetlands were monitored from December 15, 2015 to November 30, 2016. In the meantime, the wastewater flow rate and the proper functioning of the wetland system were monitored regularly in the morning and in the afternoon times. The most common activities during the monitoring period were checking of the piping system and valves whether they are blocked with solids or not, removing of weds which grew on the wetland beds and protecting the wetlands system from animals. After the monitoring period of 12 months, the pilot scale constructed wetlands system was handed over to Addis Ababa Water and Sewerage Authority as it was applied in the compound of Kotebe WWTP which has been owned by the Authority.

3.4 Sampling and laboratory analyses

For water quality monitoring, triplicate grab samples were collected every two weeks from the influent and the three outlets /effluents/ of the wetland systems. In addition to this, to determine the nutrient content of the wetland plants, the above-ground and below-ground plant parts/samples/ were taken from three randomly selected quadrants with dimentions of 0.25m x 0.25m on each wetland cells in May and November, during the monitoring period. The water quality parameters such as BOD_5, COD, TSS, NO_3^--N, NH_4^+-N, TN, orthophosphate and TP were analyzed at Addis Ababa Environmental Protection Authority (AA EPA) laboratory based on the standard methods for the examination of water and wastewater (APHA, 1999). Temperature, pH, electrical conductivity and DO were measured onsite. Likewise, the analyses of triplicate samples of Fecal Coliform (FC) and the N and P content of wetland plants were done at the laboratory of Water and Energy Design and Supervision Works Sector, Ethiopian Construction Design and Supervision Works Corporation.

Table 3-2: Summary of the laboratory methods and instruments used to measure wastewater parameters during the monitoring period.

Parameters	Laboratory methods/instruments
Temperature, T^o	Portable pH/EC/TDS/^{o}C Meters, HI 9811-5
pH	Electrometeric method Portable pH/EC/TDS/^{o}C Meters, HI 9811-5
Electrical Conductivity, EC	Portable pH/EC/TDS/^{o}C Meters, HI 9811-5
Dissolved Oxygen, DO	Membrane Electrode DO meter, ExStik* DO600
Biochemical Oxygen Demand, BOD	Pressure sensor
Chemical Oxygen Demand, COD	Spectrophotometer, DR 3900, Hack method
Total Suspended Solids, TSS	Spectrophotometer, DR 3900, Hack method
Nitrate Nitrogen, NO_3^{-}–N	Spectrophotometer, DR 3900, Hack method
Ammonium Nitrogen, NH_4^+ - N	Spectrophotometer, DR 3900, Hack method
Total Nitrogen, TN	Spectrophotometer, DR 3900, Hack method
Orthophosphate, PO_4^{3-} - P	Spectrophotometer, DR 3900, Hack method
Total Phosphorous, TP	Spectrophotometer, DR 3900, Hack method
Fecal Coliforms, FC	Membrane filter technique
Wetland plant P content	DTPA extraction, KH_2PO_4 extraction, Olsen, Kjeldahl digestion Walklay black, Ammonium acetate and instrumental

* GARMIN GPSmap 62 - was used to take GPS readings.

The concentration based removal efficiencies of the pilot scale constructed wetland systems for each of the wastewater parameters were calculated by using the following formula:

$$\text{Removal Efficiency (\%)} = \frac{(Ci - Ce)}{Ci} * 100 \qquad \text{Eqn (3.3)}$$

Where:

C_i = the concentration of the wastewater parameter in the influent, and

C_2 = the concentration of the wastewater parameter in the effluent

The loading rates for each of the wastewater parameters were calculated using the concentrations of the influents and hydraulic loading rates by using the formula:

$$\text{Pollutant loading rates } (g/m^2.d) = (Q_{in}/A) \times C_{in} \qquad \text{Eqn (3.4)}$$

Where:

C_{in} = concentration of wastewater parameter within the influent, and

Q_{in}/A = hydraulic loading rate

In line with this, the areal removal rate for each wastewater parameter was computed based on the formula, by assuming that Total Q_{in} = Total Q_{out}.

$$\text{Areal removal rate } (g/m^2.d) = [(Q_{in} \times C_{in}) - (Q_{out} \times C_{out})]/A$$

$$= [Mass_{in} - Mass_{out}]/A$$

$$= HLR \times (C_{in} - C_{out})$$

Where:

Q_{in} = influent concentration (mg/L)

C_{out} = effluent concentration (mg/L)

Q_{in} = wastewater flow rate (m^3/d)

HLR = hydraulic loading rate (m/d)

$Mass_{in}$ = mass of the pollutant within the influent (g)

$Mass_{out}$ = mass of the pollutant within the effluent (g)

A = area of the wetland system (m^2)

In general, for the calculations given above, an area of 24 m^2, a constant inflow and outflow rate of 1.05 m^3/d and a hydraulic loading rate of 0.044 m/d for the subsurface flow pilot scale constructed wetlands were taken into account.

3.5 Data analysis

The data obtained from the study were analyzed by using MS Office Excel 2007 and IBM SPSS Statistics version 21 software package. Mean, standard error, removal efficiencies, linear correlation, and analysis of variance (ANOVA) tests were done using these packages. The statistical results of one-way ANOVA were presented in the following form: (ANOVA; $F_{0.95}$ (X, Y); N) where $F_{0.95}$ = 95% confidence limit; X and Y = degrees of freedom and N = obtained value of F. If N is larger than the critical F-value at (X, Y) degrees of freedom and $P < 0.05$, the result is significant (reject the null hypothesis).

3.6 Total cost of construction

The total construction cost for the implementation of the pilot-scale constructed wetland systems was $ 2,504.15 and the breakdown is indicated in table 3.3. The cost for maintenance of the system and performance monitoring was not included in the cost summary. Most of the costs used in conducting this project was covered by the fund obtained from Wollo University (WU) and Bursary Research Fund (BRF).

Table 3.3: Cost summary of the construction of the pilot scale constructed wetland systems applied at Kotebe WWTP, Addis Ababa, Ethiopia.

Description	Quantity	Unit cost ($)	Total cost ($)
Excavation, m^3	60	5.25	315.00
Stone, m^3	11	10.90	119.90
Blocks, each	400	0.35	140.00
Sand, m^3	12	15.20	182.40
Cement, bag	40	18.90	435.00
Geo-synthetic membrane, m^2	120	0.90	108.00
Pipes – HDPE, 2 inch diameter, m	15	3.05	45.75
- HDPE, ¾ inch diameter, m	200	0.35	70.00
- PVC, 50mm diameter with 6m length, each	14	4.35	60.90
Gravel 4mm diameter, m^2	8	16.50	132.00
Gravel 3mm diameter, m^3	14	16.50	231.00
Gravel 2mm diameter, m^3	8	15.00	120.00
Gravel 1mm, m^3	14	11.30	158.20
Manpower: – Mason, contractor			260.00
- 3 laborer, days	12	3.50	126.00
		Total	***2,504.15***

3.7 Sources of meteorological data of the study area

Meteorological data such as daily average rainfall and temperature for the surroundings of the study area were obtained from the nearby office of the National Meteorological Agency (NMA). Therefore, the daily average values of rainfall and temperature for the sampling dates are presented in table 4.3. However, the amount of water loss from the CWs through evapo-transpiration was not available and therefore the water budget calculation for the system applied at Kotebe WWTP was not done.

Based on the meteorological data obtained from NMA, the daily average rainfall values of each season during the monitoring period were 0.75 ± 2.67 mm for winter (December, January and February), 4.31 ± 8.55 mm for autumn (September, October and November), 6.62 ± 8.18 mm for summer (June, July and August) and 1.64 ± 4.48 mm for spring (March, April and May). The highest daily average of rainfall was observed in summer and the lowest value was observed in winter. Although autumn is known to be light rainy season, the daily average value of rainfall was to some extent high in 2016.

Similarly, the daily average values of temperature were 24.2 ± 0.95 °C for winter, 24.3 ± 2.00 °C for autumn, 22.3 ± 1.95 °C for summer and 23.4 ± 1.36 °C for spring. The highest seasonal average temperature value was seen in spring while the lowest value was observed in autumn.

Table 3-4: Daily average data of rainfall and ambient air temperature for the sampling dates.

Sr. No	**Sampling dates**	**Average daily rainfall (mm)**	**Average daily Ambient air temperature (°C)**	**Remark**
1	15/12/15	0.0	24.3	
2	29/12/15	0.0	24.5	
3	13/01/16	1.0	23.4	
4	28/01/16	2.6	23.0	
5	11/02/16	0.0	25.2	
6	25/02/16	9.0	22.8	
7	10/03/16	0.0	24.6	
8	28/03/16	0.0	25.2	
9	12/04/16	2.1	26.4	
10	27/04/16	19.6	24.0	
11	12/05/16	0.0	23.6	
12	26/05/16	2.8	23.0	
13	09/06/16	0.0	25.2	
14	23/06/16	4.1	23.0	
15	11/07/16	2.5	20.6	
16	26/07/16	6.6	23.0	
17	10/08/16	8.2	20.4	
18	25/08/16	8.7	22.4	
19	08/09/16	0.0	21.5	
20	22/09/16	0.0	24.8	
21	10/10/16	1.3	25.6	
22	26/10/16	0.0	24.5	
23	10/11/16	0.0	23.0	
24	30/11/16	0.0	22.2	

Source: National Meteorology Agency of Ethiopia (NMA), data delivery and dissemination case team, Addis Ababa.

CHAPTER FOUR

RESULT AND DISCUSSION

4.1 Characteristics of the Domestic Wastewater

The operational parameters of the raw domestic wastewater (influent) in the primary treatment, sedimentation tank, and the treated wastewater (effluent) discharged into the pilot scale constructed wetland systems are shown in Table 4.1. The data did not show significant difference in the removal of pollutants except, TSS and FCs by the sedimentation tank, after a decrease of 43% and 28% of the same after primary treatment. The removal of BOD and COD were 9% and 11%, respectively and this might be attributed to the removal of TSS by sedimentation. However, the concentration of NO_3^- - N, PO_4^{3+} - P, and TP was increased after primary treatment. The increament in the concentration of PO_4^{3+} and TP may be attributed to the breaking down of high-molecular-weight polyphosphates to low-molecular-weight phosphates by the process of hydrolysis (Korkusuz *et al*, 2004).

The effect of biological breakdown by the actions of microorganisms or chemical processes seem to have been insignificant during primary treatment since these processes require relatively longer period of time to play their role in wastewater treatment while the detention time of the wastewater in the sedimentation tank was 3 - 4 hours. As a result, the removal of wastewater pollutants was not significant.

Table 4-1: Characteristics of domestic wastewater before and after primary treatment and removal efficiencies (%) of the sedimentation tank, Addis Ababa.

Description	Wastewater Parameters (mg/L)								
	BOD	COD	TSS	NO_3^--N	NH_4^+-N	TN	PO_4^{-3}	TP	FC (CFU/100 ml)
Influent (St. dev.)	187 ± 20	413 ± 142	311 ± 133	2.7 ± 1.8	53 ± 24.8	67 ± 20	4.1 ± 2.5	6.4 ± 3.0	156667 ± 67885
Effluent (St. dev.)	170 ± 34	369 ± 92	178 ± 65	4.9 ± 3.4	50 ± 17.6	68 ± 25	4.4 ± 2.3	6.9 ± 2.8	112333 ± 21008
RE (%) after primary treatment	9 %	11 %	43	-82 %	NS	NS	NS	NS	28 %

The characteristics of the domestic wastewater (influent) treated using CWs in this study in comparison with reports from various countries are summarized in Table 4.2. The concentration of BOD_5 in this study was more or less similar to the concentration reported in most countries. But it was higher than the values reported in Mexico (Zurita *et al*, 2009) and Australia (Sleytre *et al*, 2007) and lower than the values in Brazil (Lana *et al*, 2013) and Ireland (Kayranli *et al*, 2010). The domestic wastewater used in this study was relatively comparable in its COD concentration with the values reported in Egypt (Abdelhakeem *et al*, 2016) and Kenya (Mburu *et al*, 2013) although it was by far higher than the value reported in Colombia (Caselles-Osorio *et al*, 2011) and Mexico (Zurita *et al*, 2009). The ratio of BOD_5: COD in this study was 0.45 and therefore it could be concluded that the domestic wastewater was biodegradable and could be classified as “low strength” wastewater.

Table 4-2: The physico-chemical characteristics of domestic wastewater (influent) of some other countries used for treatment in constructed wetland systems.

Countries	Wastewater parameters (mg/L)									References
	BOD_5	COD	TSS	NO_3^--N	NH_4^+-N	TN	PO_4^{-3}	TP	FC (CFU/ 100ml)	
Austria	150	367	-	-	42	-	-	6.6	-	Sleytr *et al*, 2007
Brazil	279	465	293	0.1	26.4	-	-	3.9	-	Lana *et al*, 2013
China	103-207	213-381	-	-	48-112	71-104	-	4.8-12.1	-	Lu *et al*, 2015
Colombia	-	132	-	5.3	23	-	5	-	87677	Caselles-Osorio *et al*, 2011
Egypt	181-253	383-624	180-281	5.2-6.7	30-42	-		2.5-2.9		Abdelhakeem *et al*, 2016
Greece	-	458	-	0.01	-	49	8.18	-	-	Prochaska *et al*, 2007
Ireland	761.1	1279.3	2183.8	4.8	32.1	-	3.7	-	-	Kayranli *et al*, 2010
Kenya	232	424	118	39	4	-	-	-	-	Mburu *et al*, 2013
Mexico	115.5	247.5	57.5	9.3	15.7	28.7		8.3		Zurita *et al*, 2009
Thailand	-	93-136	47-65	1.1-1.4	11.4-21.1	17.4-27	4.3-7.1	6.7-9.8	-	Konnerup *et al*, 2009
Turkey	200	343	333	-	-	55.6	-	7.06	19823	Tuncsiper *et al*, 2012
UK	104-221	186-352	122-379	0.8-12	23-69.4	-	7.2-18.7	-	-	Al-Isawi *et al*, 2015
USA	230-392	-	418-2102	-	-	-	-	-	32900 -90400	Karathanasis *et al*, 2003
This study	186	417	206	7.51	50.8	76.6	5.84	18.27	95292	

4.2 Temperature, pH, Electrical Conductivity and Dissolved Oxygen

The temperature, pH, electrical conductivity and dissolved oxygen of the influent and effluent in this study are shown in Table 4.3. Accordingly, the average temperatures of the influent were in the range between 16.1°C - 22.4°C. Similarly, the values for effluent from the HFCW, VFCW and HyFCW were 13.5°C - 21.0°C, 14.0°C - 21°C and 13.1°C - 21.2°C, respectively. In general, the average temperature was high in winter (December - February) and spring (March - May), and low in summer (June - August) and autumn (September - November).

One-way ANOVA showed that the average temperatures of the influent and effluents of the HFCW, VFCW and the hybrid flow CW were significantly different from season to season. Accordingly, the ANOVA results were $F_{0.95}$ (3, 20) = 99.27; $P < 0.05$ for the influent, $F_{0.95}$ (3, 20) = 57.27; $P < 0.05$ for HFCW effluent, $F_{0.95}$ (3, 20) = 64.46; $P > 0.05$ for VFCW effluent and $F_{0.95}$ (3, 20) = 98.58; $P < 0.05$ for hybrid flow CW effluent.

With regards to pH, it was slightly basic with the pH of the influent in the range of 7.2 and 7.6; whereas the pH of the effluent ranged between 7.2 -7.7 without showing significant difference amongst the sampling months or types of constructed wetlands. The average values of EC in the influents were in the range of 906 - 1012 µs, while the values within the effluents of the HFCW, VFCW and HyFCW were 790 µs - 932 µs, 778 µs - 925 µs and 798 µs - 923 µs, respectively.

Similarly, the DO of the influent was between 0.4 mg/L and 1.2 mg/L and the average values of the effluent varied between 0.8 mg/L - 2.8 mg/L for HFCW, 1.1 mg/L - 3.3 mg/L for VFCW and 1.0 mg/L - 3.2 mg/L for hybrid flow CW systems. This shows that the DO of the effluent was slightly higher at the VF and hybrid flow systems than the DO of the HF wetland system, indicating better oxygen supply/aeration in the former two systems. Similarly, DO concentration was observed to be higher in summer (June - August) and lower in winter (December - February). This might have resulted from the seasonal temperature difference that affects the solubility of oxygen in water as the solubility of oxygen in water decreases when temperature increases.

Table 4-3: Mean values of temperature, pH, dissolved oxygen and electrical conductivity in the influent and effluents of the subsurface HFCW, VFCW and HyFCW of the pilot scale CW systems.

Parameters	**Sample types**	**December-February**	**March-May**	**June-August**	**September-November**
T° (^{O}C)	Influent	21.6 ± 1.1	22.4 ± 0.8	18.3 ± 1.4	16.1 ± 1.4
	HFCW Effluent	21.0 ± 0.7	20.8 ± 0.5	16.6 ± 2.0	13.5 ± 1.9
	VFCW Effluent	20.1 ± 0.6	21.0 ± 0.2	15.7 ± 1.7	14.0 ± 1.8
	HyFCW Effluent	20.4 ± 0.4	21.2 ± 0.8	16.3 ± 1.5	13.1 ± 1.4
pH	Influent	7.4 ± 0.3	7.6 ± 0.3	7.2 ± 0.3	7.3 ± 0.4
	HFCW Effluent	7.4 ± 0.5	7.6 ± 0.4	7.4 ± 0.4	7.4 ± 0.3
	VFCW Effluent	7.7 ± 0.3	7.4 ± 0.1	7.5 ± 0.6	7.6 ± 0.4
	HyFCW Effluent	7.7 ± 0.4	7.4 ± 0.4	7.3 ± 0.4	7.5 ± 0.3
DO (mg/L)	Influent	0.4 ± 0.1	0.9 ± 0.5	1.2 ± 0.2	0.6 ± 0.2
	HFCW Effluent	0.8 ± 0.3	1.7 ± 0.6	2.8 ± 0.3	1.4 ± 0.3
	VFCW Effluent	1.1 ± 0.5	2.1 ± 0.6	3.3 ± 0.5	1.9 ± 0.5
	HyFCW Effluent	1.0 ± 0.3	2.2 ± 0.6	3.2 ± 0.3	1.7 ± 0.4
EC (µs)	Influent	959 ± 85	921 ± 121	906 ± 67	1012 ± 51
	HFCW Effluent	925 ± 119	932 ± 64	790 ± 34	802 ± 15
	VFCW Effluent	918 ± 128	925 ± 98	788 ± 60	778 ± 30
	HyFCW Effluent	913 ± 121	923 ± 68	798 ± 85	901 ± 57

4.3 Removal of Biochemical Oxygen Demand (BOD_5)

As it is indicated in Table 4.4 and Figure 4.1, the BOD_5 of the influent (at the inlet) of the CW systems at different seasons was within the range of 163.5 ± 25.1 mg/L - 198.3 ± 21.1 mg/L, with the average value of 186.4 ± 15.7 mg/L. During the monitoring period, the average BOD_5 concentrations within the effluents of the horizontal, vertical and hybrid systems were 20.3 ± 9.0 mg/L, 14.2 ± 7.0 mg/L and 12.0 ± 6.1 mg/L, respectively. The highest BOD_5 of 31.5 mg/L was recorded within the effluent of the horizontal CW in September - November, while the lowest value was 5.8 mg/L in the effluent of the hybrid system in March - May.

Table 4-4: The mean BOD_5 concentration (mg/L) and removal efficiencies (%) of the HF, VF and hybrid flow pilot scale CW systems.

Seasons of the year	Horizontal flow CW			Vertical flow CW			Hybrid flow CW		
	Influent BOD_5	Effluent BOD_5	RE (%)	Influent BOD_5	Effluent BOD_5	RE (%)	Influent BOD_5	Effluent BOD_5	RE (%)
Dec-Feb	198.3 ± 21.1	15.6 ± 6.5	92.2	198.3 ± 21.1	9.5 ± 3.9	95.2	198.3 ± 21.1	8.1± 2.0	95.9
Mar-May	194.4 ± 25.6	10.9 ± 4.0	94.4	194.4 ± 25.6	7.3 ± 2.6	96.2	194.4 ± 25.6	5.8± 2.2	97.0
Jun-Aug	163.5 ± 25.1	23.2 ± 8.4	85.8	163.5 ± 25.1	17.6 ± 7.6	89.3	163.5 ± 25.1	15.8± 5.7	90.4
Sep-Nov	189.2 ± 15.0	31.5 ± 10.5	83.4	189.2 ± 15.0	22.3 ± 8.2	88.2	189.2 ± 15.0	18.5± 6.6	90.2
Annual	186.4 ± 15.0	20.3 ± 9.0	89.1	186.4 ± 15.0	14.2 ± 7.0	92.2	186.4 ± 15.0	12.0± 6.1	93.4

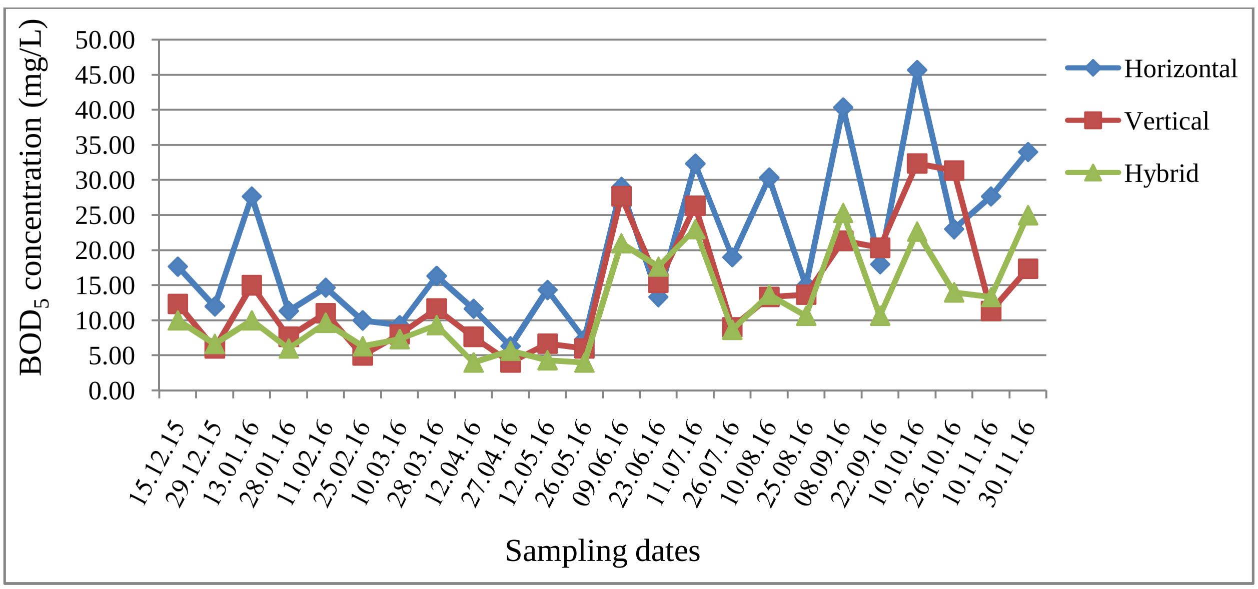

Figure 4.1: Concentration of BOD_5 within the effluents of the horizontal, vertical and hybrid pilot scale constructed wetland systems vs the sampling dates.

Accordingly, the BOD_5 removal efficiency of the pilot scale CW systems was 89.1% for the HFCW, 92.2% for the VFCW and 93.4% for the HyFCW system. The result showed that the highest percent reduction was obtained from the hybrid system and the lowest was from the horizontal flow CW system.

The one-way ANOVA demonstrated that the performance of the horizontal, vertical and hybrid systems (CW's) in removing BOD_5 was significantly different from one another statistically (one-way ANOVA; $F_{0.95}$ (2, 69) = 5.843; P < 0.05). Hence, it could be described that the difference in the wastewater flow into each of the three CWs beds resulted in significant effect on the performances of the wetlands in removing BOD_5. Because the wetland systems were supposed to function under similar conditions except for the differences in wastewater flow.

The seasonal removal efficiency of the BOD_5 of the HFCW was the lowest of 83.4% in September - November and the highest of 94.4% in March - May. Similarly, the lowest BOD_5 removal of 88.2% was recorded in September - November and the highest removal efficiency of 96.2% was obtained in March - May at the VFCW systems, whereas the lowest and the highest percent BOD_5 reduction of the HyFCW was 90.2% in Sep. - Nov and 97.0% in March - May, respectively.

The performances of the HF, VF and Hybrid flow constructed wetland systems differed significantly from season to season in removing BOD_5; one-way ANOVA results of the horizontal, vertical and hybrid systems were $F_{0.95}$ (3, 20) = 13.595; P < 0.05, $F_{0.95}$ (3, 20) = 12.496; P < 0.05 and $F_{0.95}$ (3, 20) = 18.445; P < 0.05, respectively. However, Vymazal (2014) reported that there was no significant difference between average outflow concentrations of wastewater pollutants in summer and winter periods.

Song *et al* (2006) reported that removal efficiencies for BOD_5 and COD vary from season to season. So, it could be possible to explain that the degradation of organic matter by microorganisms is influenced by climatic conditions and consequently the rate of degradation in tropical areas is higher than temperate or cold areas (Anish *et al*, 2012). The higher removal percentage of BOD_5 was then recorded particularly in winter (December - February) and spring (March - May), when higher water temperature was recorded than summer (June - August) and autumn (September - November).

Azni *et al* (2010) stated that the process of breaking down of organic carbon to CO_2 by microorganisms to obtain energy for growth requires dissolved oxygen (Azni *et al*, 2010). In this study, the performance of HyFCW/VFCW was better than HFCW for the removal of BOD_5 and this could be resulted from the higher DO concentration in the effluents of VF/HyF systems than the HF system. The higher DO concentration might be mainly occurring as the unsaturated flow condition in VF bed presented more oxygen for the oxidation-reduction potential to take place in vertical subsurface flow wetland (Pandey *et al*, 2013).

Both aerobic and anaerobic decomposition processes can take place in the removal of organic matter. However, the nature of biochemical reactions depends on the conditions created as a result of the rate of oxygen transfer in the wetland system. If there is adequate oxygen supply, aerobic decomposition is so rapid that the accumulation of organic matter in the wetland is small. But if the rate of oxygen transfer cannot meet the oxygen requirements, the removal process becomes anaerobic decomposition which results in the accumulation of organic matter in the wetland (Wallace, 2004; Garcia *et al*, 2005). Hence, it could be explained that there was better oxygen transfer in the vertical flow and hybrid flow CW systems by which the higher BOD_5 removal was observed.

In general, the effluent BOD_5 from the three subsurface flow CW systems met the provisional discharge standards, 80 mg/L, which was set by Ethiopian Environmental Protection Authority (EEPA, 2003). Hence, the result showed that the BOD_5 of the effluent was below the discharge limit values indicating the system was effective in fulfilling the regulatory limits to discharge the effluent into the environment /water bodies.

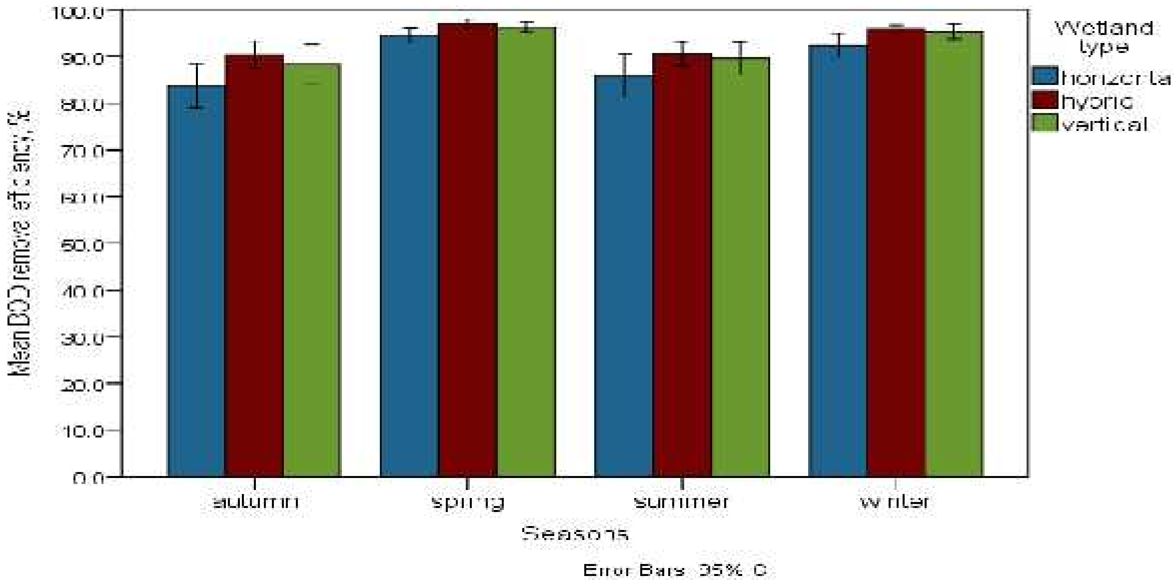

Figure 4.2: Seasonal BOD_5 removal efficiencies of the horizontal, vertical and hybrid pilot scale constructed wetland systems.

Some of the BOD_5 removal percentages of subsurface flow constructed wetland systems in other countries are shown in Table 4.5. The CWs system in this study showed higher performance compared to the study conducted in Egypt (Abdelhakeem *et al*, 2016), India (Prashant *et al*, 2013), Jordan (Albalawneh *et al*, 2016) and Nepal (Pandey *et al*, 2013). However, other reports from both temperate and tropical regions also showed comparable results in removing BOD_5 (Laaffat *et al*, 2015; Vymazal, 2014b; Lu *et al*, 2015; Vergeles *et al*, 2015).

Table 4-5: Comparison of current BOD_5 removal efficiencies to subsurface flow constructed wetlands applied in other countries

Country	Removal Efficiency, %	References
Czech Republic	96	Vymazal, 2014b
Morocco	92	Laaffat *et al*, 2015
China	87.9	Lu *et al*, 2015
Egypt	84	Abdelhakeem *et al*, 2016
Ukraine	82.6	Vergeles *et al*, 2015
Nepal	41.4 – 89.3	Pandey *et al*, 2013
India	40 - 75	Prashant *et al*, 2013
Jordan	55	Albalawneh *et al*, 2016
Pakistan	50	Mustafa, 2013
Ethiopia	89.1 -93.4	This study

For the period of performance monitoring, the loading rate of BOD_5 ranged between 5.8 and 9.9 $g/m^2.d$ (8.2 ± 1.092 $g/m^2.d$) (Fig 4.3). The correlation between loading rate and removal rate by the horizontal, vertical and hybrid system revealed that the BOD_5 removal rate was highly dependent on the BOD_5 loading rate; and the removal rates for the hybrid ($R^2 = 0.871$) and the vertical ($R^2 = 0.840$) systems, were slightly more dependent on the loading rate compared to the horizontal system ($R^2 = 0.747$).

The difference in the removal efficiency of the three CWs system could be related to the differences in the oxygen supply, which in turn affects the number of heterotrophs since the rate of aerobic biodegradation is faster than anaerobic biodegradaton. Ye *et al* (2012) stated that there is a positive correlation between the levels of DO and the biofilm mass, showing the presence of other sources of oxygen supply in addition to the oxygen in the influent, particularly in the upper part of the wetland bed. Ye *et al* (2012) also supported the assumption that the major source of oxygen for VFCWs is atmospheric re-oxygenation, and the contribution of atmospheric re-oxygenation in the process of domestic wastewater treatment is more than 99.9% of the total oxygen supply to the VFCWs. Therefore, the vertical and hybrid systems could have improved capacity to hold and perform better as the BOD_5 concentration increases.

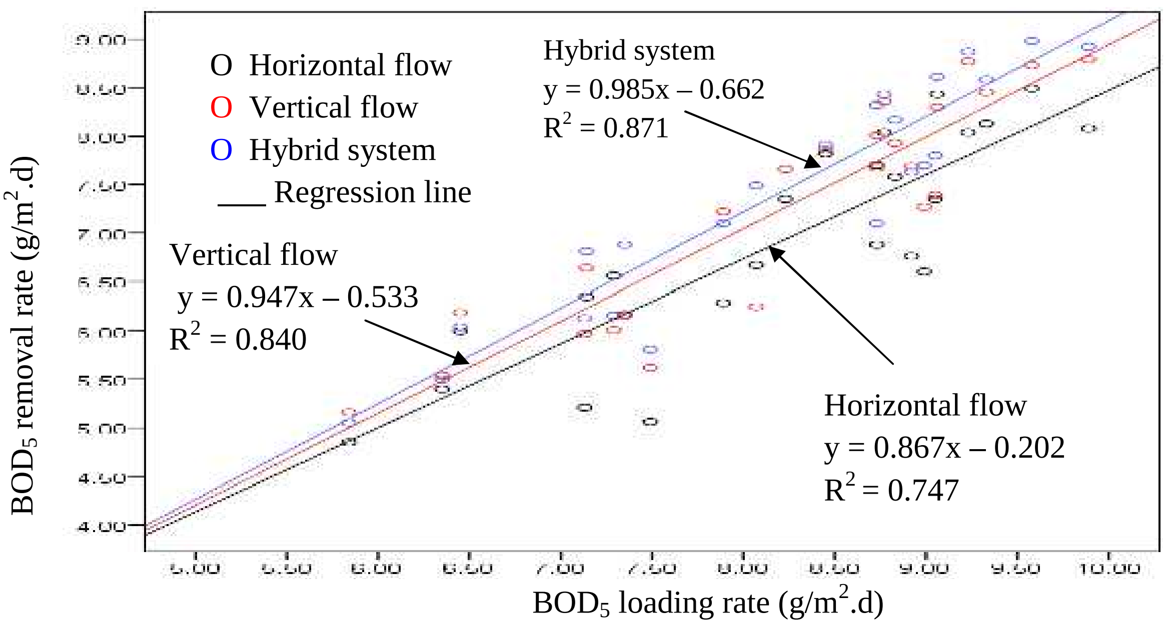

Figure 4.3: BOD_5 loading rate ($g/m^2.d$) against BOD_5 removal rate ($g/m^2.d$) of the pilot scale constructed wetland systems.

4.4 Removal of Chemical Oxygen Demand (COD)

The influent COD concentrations of the wastewater discharged into the horizontal, vertical and hybrid CWs WWTP at different seasons ranged between 374.1 – 448.0 mg/L, with an average of 402.8 ± 31.84 mg/L (Table 4.6). The highest average COD concentration (448.0 ± 65.06 mg/L) within the influent was obtained in December - February (dry season) while the lowest value (374.1 ± 44.08 mg/L) was recorded within the influent in June - August (rainy season). The seasonal variation observed in the concentrations of COD within the influent showed similar trends with the seasonal variation in the concentration of BOD_5 within the influent. The seasonal average values of both parameters were slightly high during the dry season and low during the rainy season. This could be attributed to the dilution effect on the wastewater because of surface runoff or the change in water consumption habits of the residents.

Similarly, the COD values of the effluents were 61.0 - 102.1 mg/L (mean 77.2 ± 19.8 mg/L) for the horizontal bed, 51.4 - 91.3 mg/L (mean 71.3 ± 17.4 mg/L) for the vertical bed and 47.8 - 84.3 (63.7 ± 16.4 mg/L) for the hybrid bed (Fig 4.4 and Table 4.6). The lowest COD (mean 47.8 ± 10.14 mg/L) of the effluent of the hybrid constructed wetland system was recorded in March - May.

Table 4-6: The mean COD concentration (mg/L) and removal efficiencies (%) of the HF, VF and hybrid flow pilot scale CW systems.

Seasons of the year	**Horizontal flow CW**			**Vertical flow CW**			**Hybrid flow CW**		
	Influent COD	Effluent COD	RE (%)	Influent COD	Effluent COD	RE (%)	Influent COD	Effluent COD	RE (%)
Dec-Feb	448.0 ± 65.1	61.0 ± 14.3	86.4	448.0 ± 65.1	63.9 ± 15.8	85.7	448.0 ± 65.1	53.8 ± 12.6	88.1
Mar-May	399.1 ± 39.3	61.6 ± 14.4	84.6	399.1 ± 39.3	51.4 ± 10.8	87.1	399.1 ± 39.3	47.8 ± 10.1	88.0
Jun-Aug	374.1 ± 44.1	84.2 ± 13.5	77.5	374.1 ± 44.1	78.7 ± 11.5	79.0	374.1 ± 44.1	69.0 ± 12.7	81.6
Sep-Nov	390.1 ± 52.0	102.1 ± 18.2	73.8	390.1 ± 52.0	91.3 ± 26.4	76.6	390.1 ± 52.0	84.3 ± 21.2	78.4
Annual	402.8 ± 31.8	77.2 ± 19.8	80.6	402.8 ± 31.8	71.3 ± 17.4	82.1	402.8 ± 31.8	63.7 ± 16.4	84

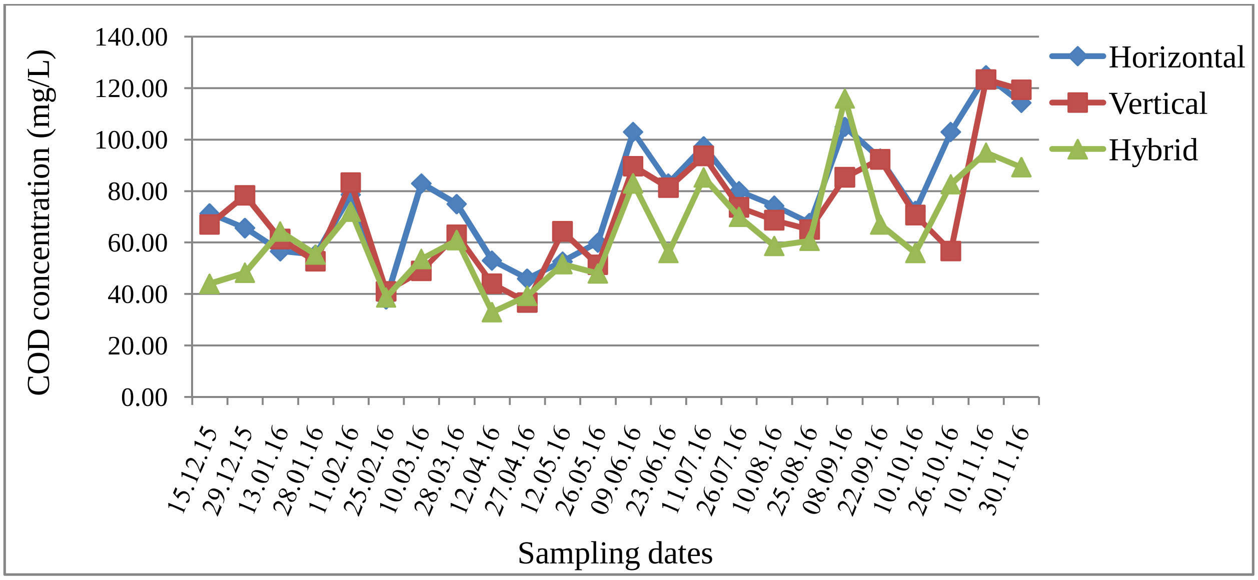

Figure 4.4: COD concentration within the effluents of the horizontal, vertical and hybrid pilot scale constructed wetland systems vs the sampling dates.

The COD removal efficiencies of the horizontal flow, vertical flow and hybrid wetland systems were 80.6 ± 5.9%, 82.1 ± 5.1% and 84.0 ± 4.8%, respectively, with the lowest removal percentage for the horizontal bed and the highest removal percentage for the hybrid bed (Fig. 4.5 and Table 4.6). Unlike $BOD_{5,}$ the performances of the horizontal, vertical and hybrid systems did not differ significantly from one another (one-way ANOVA; $F_{0.95}$ (2, 69) = 2.552; $P > 0.05$). According to the result, the difference in flow type did not affect the removal of COD.

Concerning the seasonal COD removal efficiency of the pilot scale constructed wetland systems, the lowest and the highest mean percent reductions of COD by the HFCW system were 73.8% in September - November and 86.4% in December - February, respectively. Likewise, the VFCW system showed the lowest mean removal efficiency of 76.6% in September - November and the highest mean removal efficiency of 87.1% in March - May. For the hybrid system, the lowest COD mean removal efficiency was 78.4% in Sep - Nov and the highest value was 88.0% in March- May.

The performance of the COD removal of the three CWs system was significantly different from season to season. The results of one-way ANOVA were $F_{0.95}$ (3, 20) = 45.556; $P < 0.05$ for the horizontal system, $F_{0.95}$ (3, 20) = 23.020; $P < 0.05$ for the vertical system and $F_{0.95}$ (3, 20) = 19.294; $P < 0.05$ for the hybrid system. Valsero *et al* (2012) revealed that pollutant removal efficiency of constructed wetlands is marked by seasonal difference. In addition to this, Wu *et al* (2014) described the influence of seasonal difference on the performance of CW system as the operation of CWs at cold climate is a big challenge.

Zou *et al* (2012) described that COD is mainly removed by filtration of suspended organic substances and quick biodegradation of the soluble ones by microbes in the upper 15 cm filter layer in all operational phases. Therefore, it could be concluded that the relatively low COD concentration within the effluent in March - May might be attributed to the increased temperature which in turn can create favorable conditions for microbial activities.

The COD concentrations within the outlets of Kotebe CWs were in compliance with the provisional effluent discharge standards, 250 mg/L which was set by Ethiopian Environmental Protection Authority (EEPA, 2003), indicating the Kotebe consucted wetland sytem was effective in treating domestic wastewater.

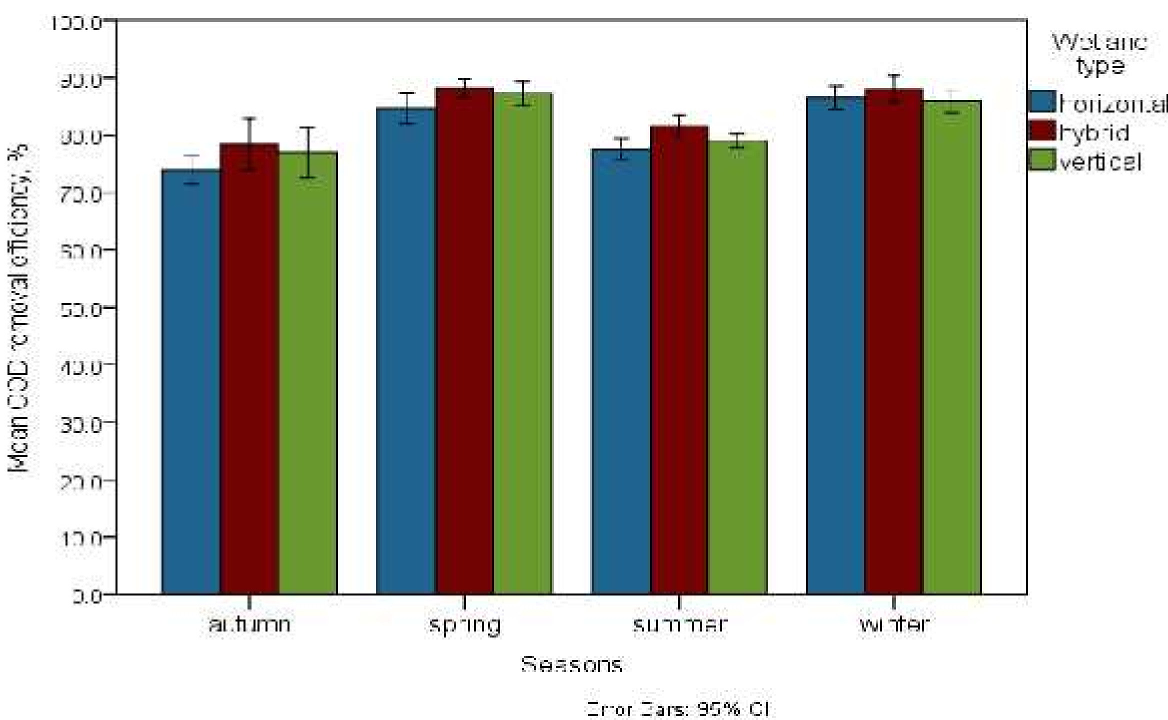

Figure 4.5: Seasonal COD removal efficiencies of the horizontal, vertical and hybrid pilot scale constructed wetland systems.

The mean percent reductions of COD in some other countries are presented in Table 4.7. The COD removal efficiency obtained in this study was higher than the result reported in Egypt

(Abdelhakeem *et al*, 2016), UK (Al-Isawi *et al*, 2017), Jordan (Albalawneh *et al*, 2016), Pakistan (Mustafa, 2013) and China (Yang *et al*, 2014). Likewise, the COD removal percentage was in agreement with the result reported in Ukraine (Vergeles *et al*, 2015) and India (Deeptha *et al*, 2015). However, relatively better percent reductions were reported in Kenya (Khisa and Tole, 2011) and Tunisia (Ghrabi *et al*, 2011).

Table 4-7: Comparison of current COD removal efficiencies to subsurface flow constructed wetlands applied in other countries.

Country	**COD Removal Efficiency, %**	**References**
Kenya	96.2	Kelvin and Tole, 2011
Tunisia	95	Ghrabi *et al*, 2011
India	86	Deeptha *et al*,2015
Ukraine	77	Vergeles *et al*, 2015
Egypt	75	Abdelhakeem *et al*, 2016
UK	58.6 – 70.8	Al-Isawi *et al*, 2017
Jordan	51	Albalawneh *et al*, 2016
China	48.9	Yang *et al*, 2014
Pakistan	44	Mustafa, 2013
Ethiopia	80.6 – 84.0	This study

Based on the results of performance monitoring study of the three CWs system, the COD loading rates ranged between 13.79 $g/m^2/d$ and 22.23 $g/m^2.d$ (17.72 ± 2.433 $g/m^2.d$) (Fig 4.6). The linear correlation between COD loading rate and COD removal rate was relatively similar among the three wetland systems. The values of linear correlation were (R^2 = 0.872) for hybrid system, (R^2 = 0.841) for the horizontal system and (R^2 = 0.838) for the vertical system. Even though, the correlation between COD loading rate and removal rate was not significantly different among the three design types, the capacity to hold and remove COD was slightly better in the hybrid system as the concentration increased.

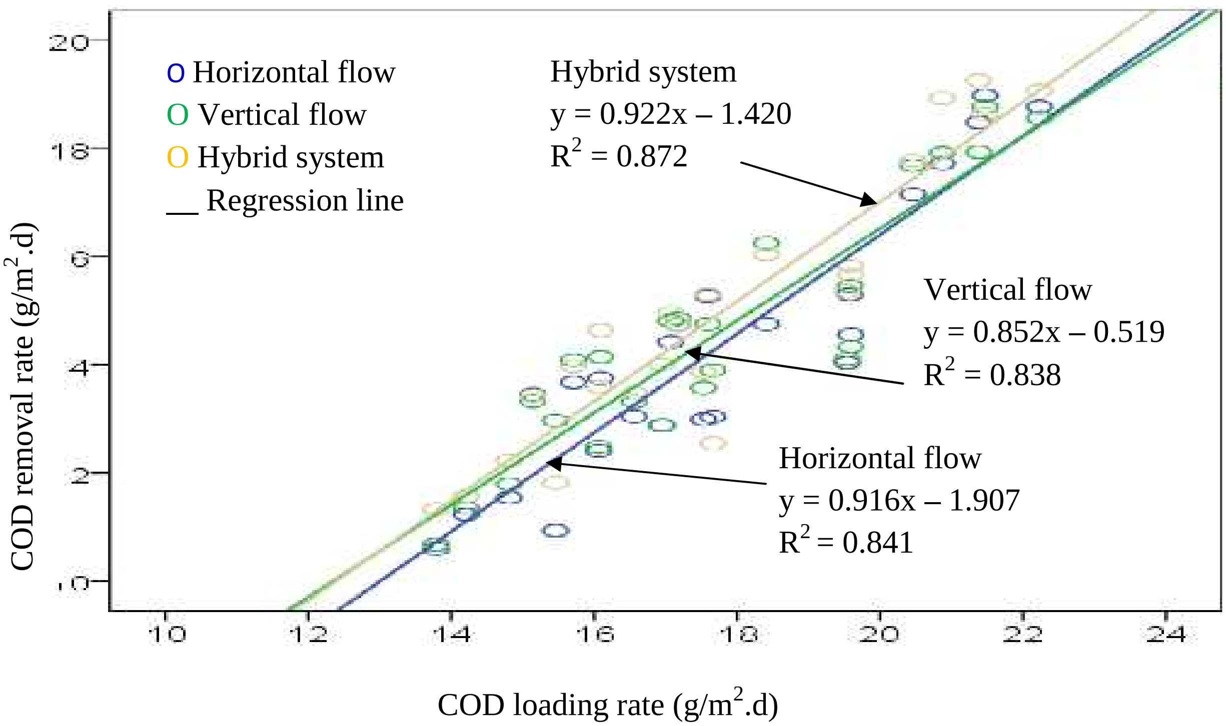

Figure 4.6: COD loading rate (g/m^2.d) against COD removal rate (g/m^2.d) of the pilot scale constructed wetland systems.

4.5 Removal of Total Suspended Solids (TSS)

The TSS of the influent, effluent and its percent reduction are presented in Table 4.8, and Figures 4.7 and 4-8. Accordingly, the average TSS in the influent of the horizontal, vertical and hybrid pilot scale CWs were in the range of 186.9 - 230.3 mg/L (206.1 ± 21.4 mg/L). The lowest TSS of the influent was recorded in December - February, while the highest value was obtained in June - August. The inflow of TSS varied from season to season (Sani *et al*, 2013), and unlike the BOD_5 and COD of the influent, the average TSS value was higher during the rainy season and low during the dry seasons. The high TSS might be due to addition of suspended solids from surface runoff following rainfall.

The TSS in the effluent was in the range of 15.8 - 36.6 mg/L (22.7 ± 9.4 mg/L), 28.4 - 46.2 mg/L (33.6 ± 8.5 mg/L) and 23.5 - 39.8 mg/L (30.7 ± 7.7 mg/L) for the HFCW, VFCW and hybrid flow systems, respectively. The lowest mean TSS was observed in the effluent of the HFCW system in December - February, while the highest value was recorded in the effluent of the VFCW system in June - August.

Table 4-8: The mean TSS concentration (mg/L) and removal efficiencies (%) of the HF, VF and hybrid flow pilot scale CW systems.

Seasons of the year	**Horizontal flow CW**			**Vertical flow CW**			**Hybrid flow CW**		
	Influent TSS	Effluent TSS	RE (%)	Influent TSS	Effluent TSS	RE (%)	Influent TSS	Effluent TSS	RE (%)
Dec-Feb	186.9 ± 41.5	15.8 ± 3.8	91.5	186.9 ± 41.5	28.4 ± 7.9	84.8	186.9 ± 41.5	23.5 ± 5.9	86.6
Mar-May	189.1 ± 74.0	18.6 ± 7.4	90.0	189.1 ± 74.0	30.9 ± 9.8	83.7	189.1 ± 74.0	25.4 ± 10.1	85.3
Jun-Aug	230.3 ± 20.5	36.6 ± 2.8	84.1	230.3 ± 20.5	46.2 ± 5.4	80.0	230.3 ± 20.5	39.8 ± 3.3	82.7
Sep-Nov	217.9 ± 43.1	20.0 ± 4.1	90.9	217.9 ± 43.1	28.8 ± 7.5	86.8	217.9 ± 43.1	34.2 ± 8.3	84.3
Annual	206.1 ± 21.4	22.7 ± 9.4	89.1	206.1 ± 21.4	33.6 ± 8.5	83.8	206.1 ± 21.4	30.7 ± 7.7	84.7

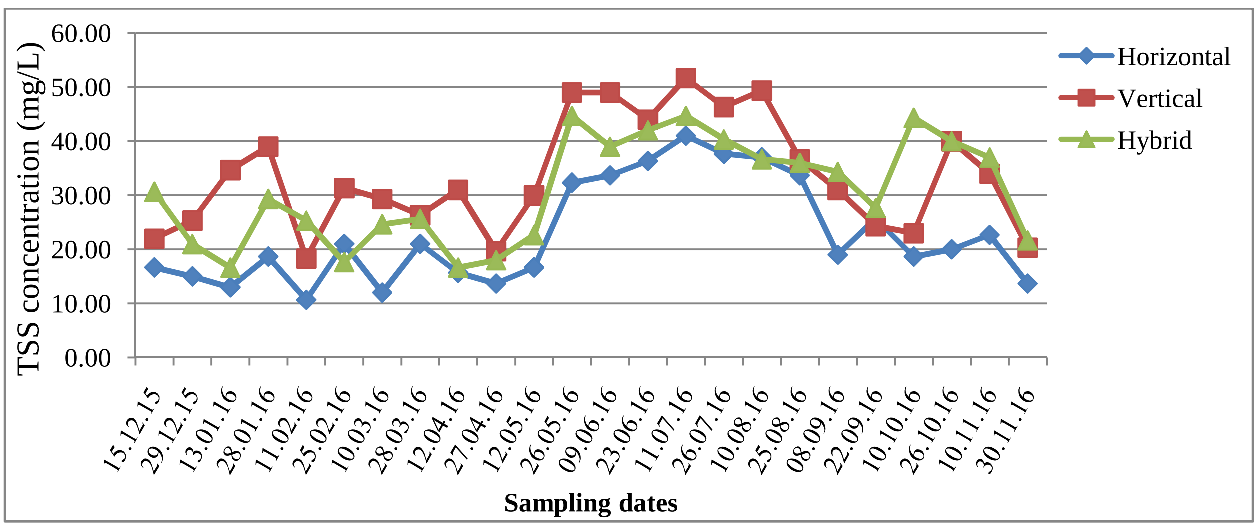

Figure 4.7: Concentration of TSS within the effluents of the horizontal, vertical and hybrid pilot scale constructed wetland systems vs the sampling dates.

The removal efficiency of TSS was 89.1± 3.4% for the HFCW, 83.8 ± 2.9% for the VFCW, and 84.7 ± 1.7% for the HyFCW systems (Table 4-8), that was significantly different from one another (one-way ANOVA; $F_{0.95}$ (2, 69) = 10.127; $P < 0.05$).

The data also showed that the removal efficiency of TSS by the horizontal, vertical and hybrid systems at different seasons of the year was in the range of mean percent reduction of 84.1% - 91.5%, 80.0 % - 86.8% and 82.7 % - 86.6 %, respectively. The lowest TSS removal efficiency was obtained in June - August for all the three CW systems. The highest TSS percent removal was recorded in December - February in the case of both the HFCW and HybFCW systems and in March - May for the VFCW system.

The performances in the vertical and hybrid systems did not differ significantly from season to season in removing TSS; one-way ANOVA results of the vertical and hybrid systems were, $F_{0.95}$ (3, 20) = 1.105; $P > 0.05$ and $F_{0.95}$ (3, 20) = 1.910; $P > 0.05$, respectively. However, the performance of the horizontal flow CW system was different from season to season, $F_{0.95}$ (3, 20) = 9.362; $P < 0.05$. The seasonal variation in the horizontal flow system might be caused by unexpected operational consequences such as inlet zone flooding during summer/rainy season (Kadlec and Wallace, 2009).

Although, there was anticipation that vertical flow wetlands are more successful in removing TSS (Kadlec and Wallace, 2009), higher removal percentage, 89.1%, was recorded from the horizontal bed, while the VFCW and HyFCW systems showed almost comparable percent reduction, 83.8 % and 84.7 %, respectively. But Haghshenas-Adarmanadabi *et al* (2016) pointed out that the removal efficiency of HFCWs was significantly higher than those of the VFCWs. This might be as a result of the more rapid biological processes taking place internally in the wetland system that increased more solids into the effluents in the vertical and hybrid flow beds.

Thomas and William (2001) reported that the settling rate of particles depends on a number of factors such as, the wetland length that affects the efficiency in removing SSs. So, the slight increase in the TSS removal percentage of the horizontal flow CW might be related with its length-through which the wastewater flows. In the case of the VFCW, the influent enters into the bed vertically all over the bed.

The TSS of the effluent from the subsurface flow CW system of this study hence met the discharge standards to water bodies (EEPA, 2003). The discharge standard for TSS is 100 mg/L.

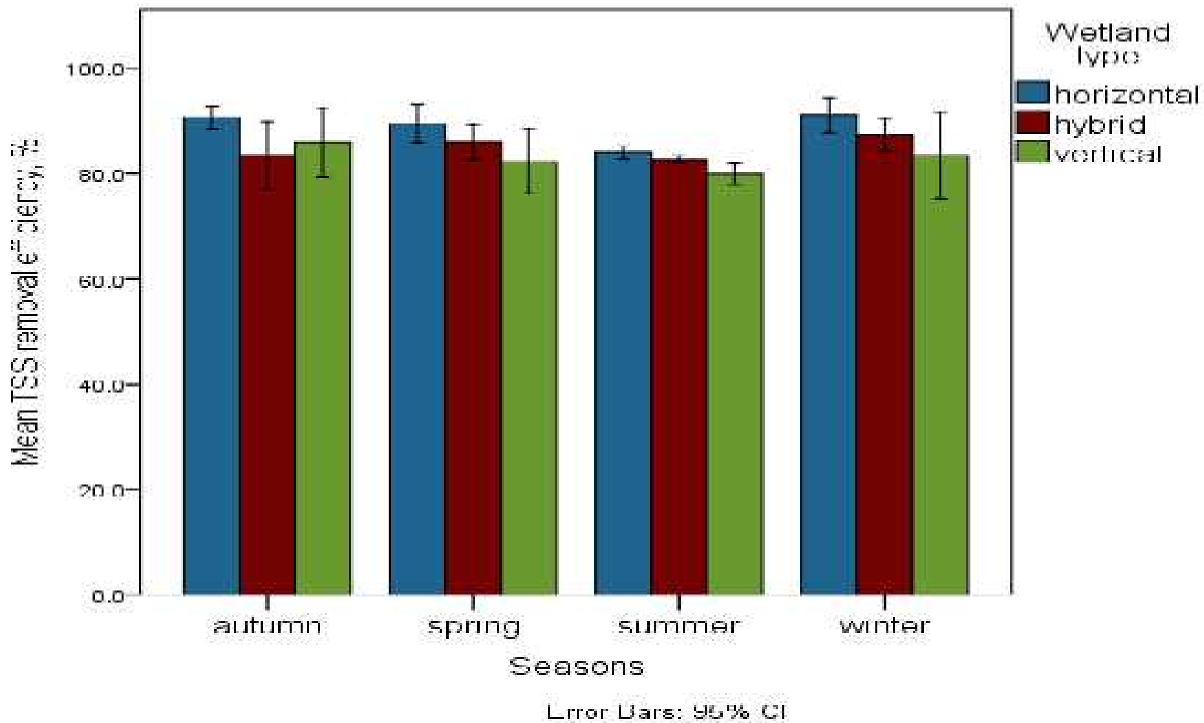

Figure 4.8: Seasonal TSS removal efficiencies of the horizontal, vertical and hybrid pilot scale constructed wetland systems.

The percent reductions of TSS by the subsurface flow CW systems applied in some other countries are presented in Table 4.9. Based on the result, the average TSS removal efficiency of the HFCW, VFCW and HyFCW systems applied at Kotebe WWTP was higher than the performance reported in Cameroon (Fonkou *et al*, 2011); but similar to the mean percent reduction reported from Kenya (Kelvin and Tole, 2011), Greece (Gikas and Tsihrintzis, 2010), and UK (Al-Isawi *et al*, 2017). However, the removal efficiency obtained in this study was lower than the results obtained from Tunisia (Ghrabi *et al*, 2011) and Italy (Masi *et al*, 2013) as shown in Table 4.9.

Table 4-9: Comparison of current TSS removal efficiencies to subsurface flow constructed wetlands applied in other countries.

Countries	TSS Removal Efficiency, %	References
Italy	97	Masi *et al*, 2013
Tunisia	97	Ghrabi *et al*, 2011
Egypt	92	Abou-Elela *et al*, 2014
UK	91.3 – 92.4	Al-Isawi *et al*, 2017
Kenya	84.3	Kelvin and Tole, 2011
Greece	79.3	Gikas and Tsihrintzis, 2010
Pakistan	78	Mustafa, 2013
Cameroon	65.1	Fonkou *et al*, 2011
Ethiopia	83.8 -89.1	This study

During the one year performance monitoring period, the loading rate of TSS was in the range of 4.14 and 12.79 g/m^2.d (9.1 ± 2.164 g/m^2.d) (Fig 4.9). The linear correlation between loading rate and the removal rate by the horizontal, vertical and hybrid system showed that the TSS removal rate was dependent on the TSS loading rate. The value of the linear correlation for the HFCW, VFCW and hybrid systems were (R^2 = 0.973), (R^2 = 0.968), and (R^2 = 0.963), respectively. The values showed that all the systems had strong capacity to hold and remove TSS as the concentrations increased.

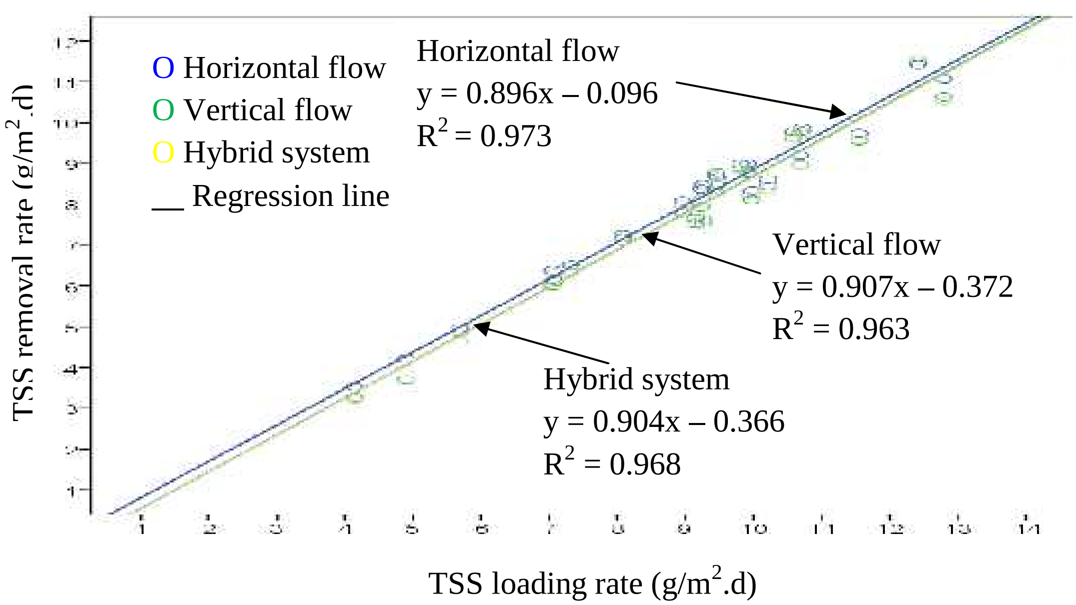

Figure 4.9: TSS loading rate (g/m^2.d) against TSS removal rate (g/m^2.d) of the pilot scale constructed wetland systems.

4.6 Removal of Ammonium (NH_4^+)

In this study, the average NH_4^+ content in the influent varied between 39.6 - 58.5 mg/L (50.8 ± 8.39 mg/L). The highest average value of NH_4^+ (58.5 ± 6.07 mg/L) of the influent was obtained in December - February; whereas the influent had the lowest average concentration of NH_4^+ (39.6 ± 2.35 mg/L) in the rainy season between June - August.

Likewise, the average concentration of NH_4^+ within the effluents of the HFCW, VFCW and hybrid systems was in the range of 20.7 ± 2.2 mg/L, 17 ± 2.0 mg/L and 17.3 ± 2.6 mg/L, respectively (Figure 4.10 and Table 4.10). The lowest mean NH_4^+ of 14.6 ± 3.6 mg/L was obtained in the effluent of the vertical flow CW in June - August, while the higher NH_4^+ value of 23.4 ± 5.0 mg/L, was recorded in the effluent of the horizontal flow CW system in September - November.

Table 4-10: The mean NH_4^+ concentration (mg/L) and removal efficiencies (%) of the HF, VF and hybrid flow pilot scale CW systems.

Seasons of the year	**Horizontal flow CW**			**Vertical flow CW**			**Hybrid flow CW**		
	Influent NH_4^+	Effluent NH_4^+	RE (%)	Influent NH_4^+	Effluent NH_4^+	RE (%)	Influent NH_4^+	Effluent NH_4^+	RE (%)
Dec-Feb	58.5 ± 6.1	20.3 ± 5.2	65.3	58.5 ± 6.1	16.6 ± 4.2	71.7	58.5 ± 6.1	15.2 ± 1.9	74.0
Mar-May	55.8 ± 7.6	21.0 ± 2.7	62.3	55.8 ± 7.6	17.2 ± 4.5	69.2	55.8 ± 7.6	18.3 ± 3.7	67.1
Jun-Aug	39.6 ± 2.4	18.1 ± 3.2	54.3	39.6 ± 2.4	14.6 ± 3.6	63.1	39.6 ± 2.4	15.1 ± 2.4	61.9
Sep-Nov	49.2 ± 13.3	23.4 ± 5.0	52.4	49.2 ± 13.3	19.4 ± 5.8	60.6	49.2 ± 13.3	20.4 ± 5.2	58.7
Annual	50.8 ± 8.4	20.7 ± 2.2	58.6	50.8 ± 8.4	17.0 ± 2.0	66.2	50.8 ± 8.4	17.3 ± 2.6	65.4

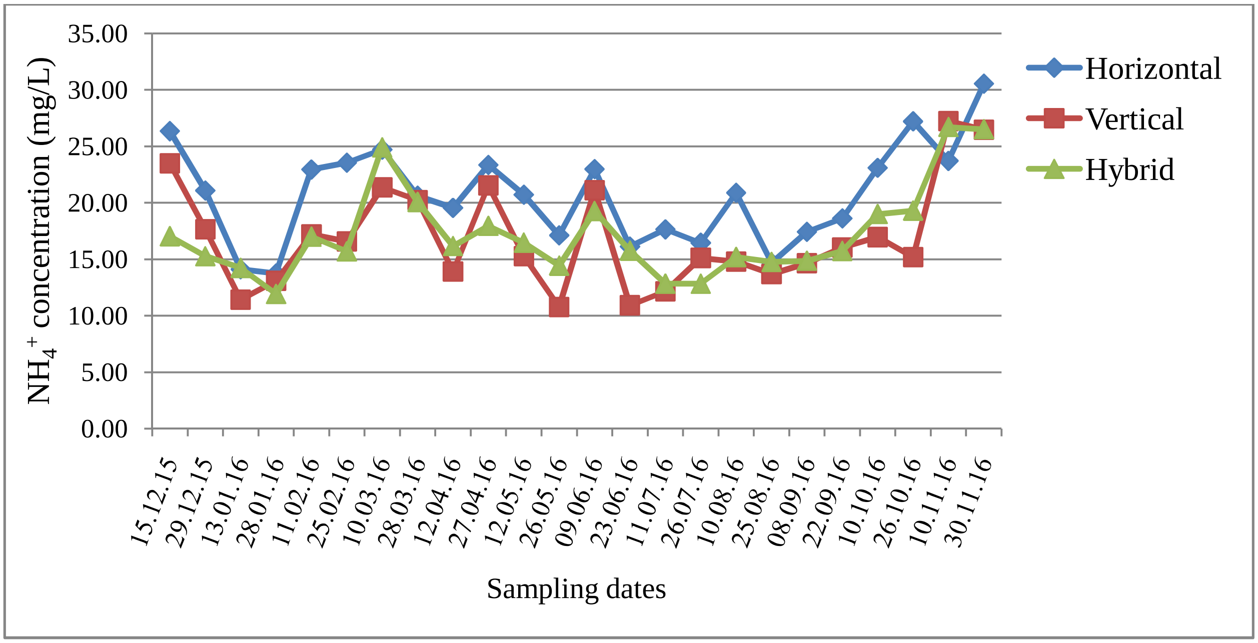

Figure 4.10: Concentration of NH_4^+ within the effluents of the horizontal, vertical and hybrid pilot scale constructed wetland systems vs the sampling dates.

The removal efficiency of the pilot scale CW systems in removing NH_4^+ was 58.6 % for the HFCW, 66.2 % for the VFCW and 65.4 % for the hybrid flow constructed wetland systems (Figure 4.11 and Table 4.10). The data showed that high variability among the three wetland beds, in that the highest average and lowest removal efficiencies were recorded in the vertical flow CW and horizontal flow CW systems, respectively. The performance of the horizontal, vertical and hybrid flow systems in removing NH_4^+ was significantly different from one another (one-way ANOVA; $F_{0.95}$ (2, 69) = 7.952; $P < 0.05$).

Regarding the seasonal performance of NH_4^+ removal of the CW, the HFCW system showed the lowest and highest performance of 52.4 % in September - November and 65.3 % in December - February, respectively. For the VF system the lowest percentage removal was 60.6 % in September - November and the highest value was 71.7 % in December - February. Similarly, the hybrid system showed the lowest percentage removal of 58.7 % in Sep - Nov and the highest NH_4^+ removal of 74.0 % in December - February. The one-way ANOVA with $F_{0.95}$ (3, 20) = 8.095; $P < 0.005$, $F_{0.95}$ (3, 20) = 5.104; $P < 0.05$ and $F_{0.95}$ (3, 20) = 21.344; $P < 0.05$ for the horizontal, vertical and hybrid systems, respectively showed the significant removal difference of NH_4^+ from season to season. The highest NH_4^+ removal during dry seasons could be related to an increase in water temperature that enhanced high nitrification rate (Tuncsiper, 2009; Dong *et al*, 2013)

Unlike most of the other parameters considered in this study, the NH_4^+ content of the effluent coming out of the three CWs was high, that did not fulfill the standard limit of 5 mg/L set by Ethiopian Environmental Protection Authority to discharge wastewater into the environment/water bodies (EEPA, 2003).

The concentration of NH_4^+ was relatively high in the domestic wastewater used in this study and this might be related with the lower per capita water consumption by the residents. The low per capita water consumption could increase the concentration of animal and plant tissue and excreted urea in the wastewater. The assumption is strengthened by the fact that the concentration had lower values during the rainy season when dilution of the wastewater occurred because of increased water consumption (using of rain water at least for domestic purposes) and infiltration into the sewerage system. NH_4^+ is formed, under aqueous conditions, by the rapid hydrolysis of ammonia formed from the breaking down of N containing organic compounds (Wallace, 2004).

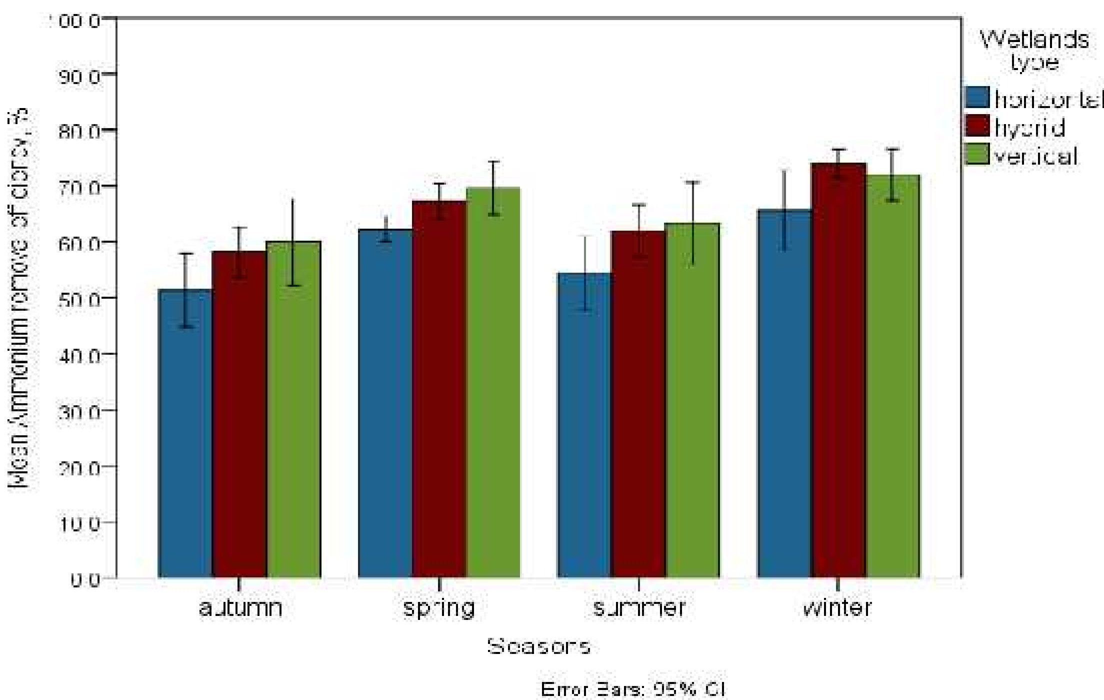

Figure 4.11: Seasonal NH_4^+ removal efficiencies of the horizontal, vertical and hybrid pilot scale constructed wetland systems.

However, the average removal percentage of NH_4^+ obtained in this study was higher than the NH_4^+ percent reduction reported in Pakistan (Mustafa, 2013), Egypt (Abou-Elela *et al*, 2014) and China (Guo *et al*, 2014). The NH_4^+ percent reduction was similar to the results reported in Turkey (Tuncsiper, 2009), Ireland (Kayronli *et al*, 2010) and Colombia (Caselles-Osorio *et al*, 2011). However, relatively better NH_4^+ removal efficiency of the pilot-scale subsurface flow constructed wetlands was reported in Singapore (Zhang *et al*, 2012) and UK (Al-Isawi *et al*, 2017).

Tuncsiper *et al* (2009) pointed out that HSSF wetland system shows lower NH_4^+ removal. This might be due to the reason that the horizontal flow system gives suitable environmental conditions for denitrification though the condition for nitrification is limited. Contrasting to the HSSF, the VSSF has better aeration condition and hence it provides a higher NH_4^+ removal efficiency.

Table 4-11: Comparison of current NH_4^+ removal efficiencies to subsurface flow constructed wetlands applied in other countries.

Countries	**NH_4^+ removal efficiency, %**	**References**
UK	91.3 – 92.4	Al-Isawi *et al*, 2017
Singapore	80.4	Zhang *et al*, 2012
Colombia	69	Caselles-Osoria *et al*, 2011
Turkey	63	Tuncsiper, 2009
Ireland	58.2	Kayranli *et al*, 2010
Pakistan	49	Mustafa, 2013
Egypt	45	Abou-Elela *et al*, 2014
China	38.7	Guo *et al*, 2014
Ethiopia	58.6 – 66.2	This study

The loading rate of NH_4^+ to the CWs system was varied in the range between 1.436 and 3.062 $g/m^2.d$ (2.234 ± 0.472 $g/m^2.d$) (Fig 4.12). The correlation between NH_4^+ loading rate and removal rate achieved by the the pilot scale constructed wetland systems in this study was still reasonably strong although the linear correlation values were low compared to the values of BOD_5, COD and TSS. The NH_4^+ removal rate of the horizontal system (R^2 = 0.587) and the vertical flow system (R^2 = 0.560) showed more or less similar trends while both the horizontal and the vertical system revealed strong correlation between NH_4^+ loading rate and removal rate than the hybrid system (R^2 = 0.432).

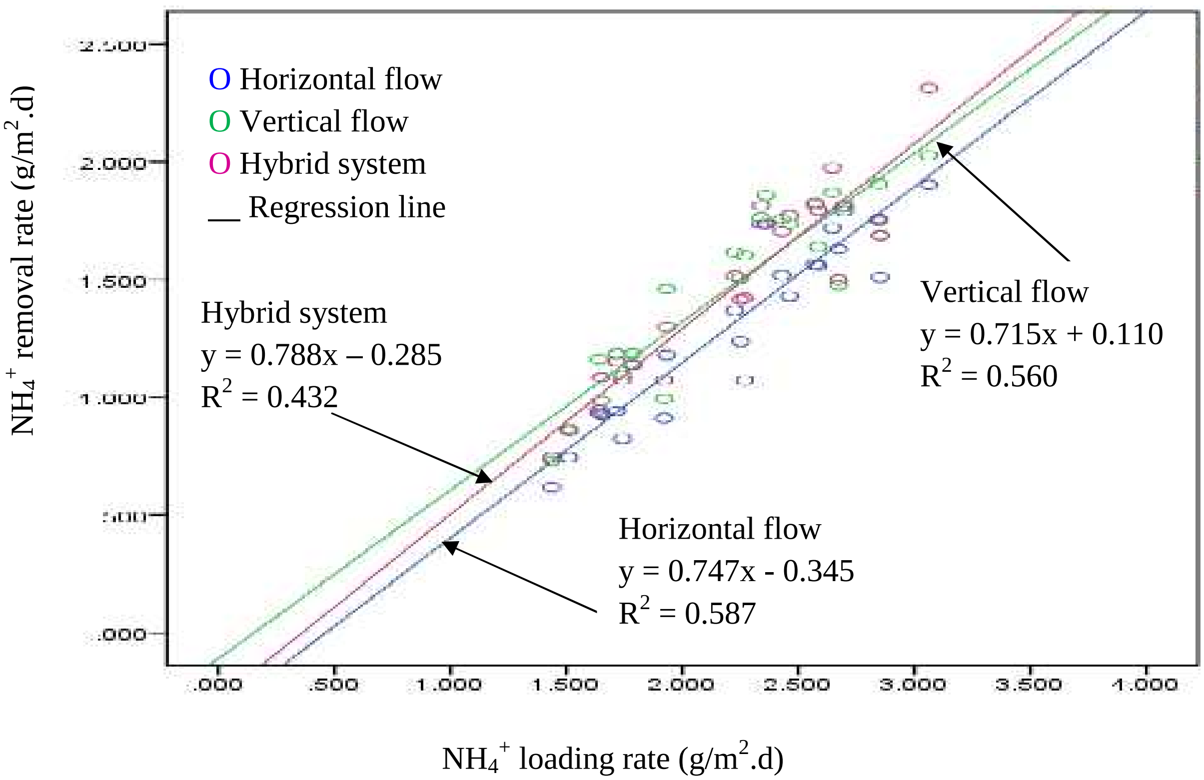

Figure 4.12: NH_4^+ loading rate (g/m^2.d) against the NH_4^+ removal rate (g/m^2.d) of the pilot scale constructed wetland systems.

4.7 Removal of Nitrate (NO_3^-)

The NO_3^- removal efficiency of the CW systems is shown in Figure 4.13 and Table 4.12. Based on the result, the influent nitrate concentrations that fed the horizontal flow, vertical flow and hybrid of the two systems were in the range of 5.83 mg/L - 10.05 mg/L (7.51 ± 1.97 mg/L).

The average NO_3^--nitrogen concentrations in the effluents were 1.83 – 4.90 mg/L (mean 2.82 ± 1.44 mg/L) for the HFCW, 1.43 - 3.57 mg/L (mean 2.19 ± 0.95 mg/L) for the VFCW and 1.56 - 3.21 (mean 2.02 ± 0.80 mg/L) for the hybrid flow CW systems. The minimum and maximum NO_3^- concentrations within the effluents of all the three constructed wetland cells were recorded during December - February and September - November, respectively (Figure 4.13 and Table 4.12).

Table 4-12: The mean NO_3^- concentration (mg/L) and removal efficiencies (%) of the HF, VF and hybrid flow pilot scale CW systems.

Seasons of the year	Horizontal flow CW			Vertical flow CW			Hybrid flow CW		
	Influent NO_3^-	Effluent NO_3^-	RE (%)	Influent NO_3^-	Effluent NO_3^-	RE (%)	Influent NO_3^-	Effluent NO_3^-	RE (%)
Dec-Feb	6.09 ± 2.2	1.83 ± 0.7	70.0	6.09 ± 2.2	1.43 ± 0.6	76.6	6.09 ± 2.2	1.56 ± 0.6	74.5
Mar-May	8.08 ± 2.9	2.65 ± 0.8	67.2	8.08 ± 2.9	1.92 ± 0.8	76.2	8.08 ± 2.9	1.69 ± 0.7	79.0
Jun-Aug	5.83 ± 2.6	1.88 ± 1.1	67.7	5.83 ± 2.6	1.82 ± 1.0	68.8	5.83 ± 2.6	1.61 ± 0.6	72.5
Sep-Nov	10.05 ± 2.9	4.90 ± 2.4	51.2	10.05 ± 2.9	3.57 ± 0.9	64.5	10.05 ± 2.9	3.21 ± 2.0	68.1
Annual	7.51 ± 1.97	2.82 ± 1.44	64	7.51 ± 1.97	2.19 ± 0.95	71.5	7.51 ± 1.97	2.02 ± 0.80	73.5

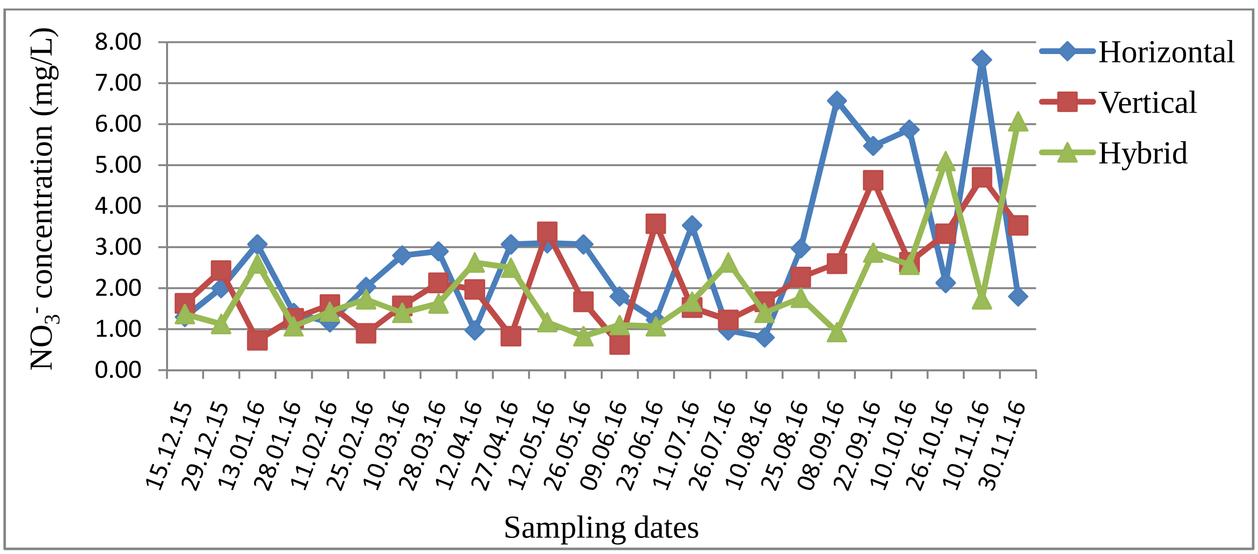

Figure 4.13: Concentration of NO_3^- within the effluents of the horizontal, vertical and hybrid pilot scale constructed wetland systems vs the sampling dates.

The data showed that the removal efficiency of the horizontal, vertical and hybrid CW systems in removing NO_3^--N during the monitoring perid was 64.0%, 71.5%, and 73.5%, respectively (Figure 4.14 and Table 4.12). Although the VFCW and the hybrid flow CW systems showed slightly better performance, the three systems did not show significant difference in removing NO_3^- (one-way ANOVA; $F_{0.95}$ (2, 69) = 2.155; P > 0.05).

The seasonal performance of the CW showed that, the lowest removal efficiency (51.2%) was obtained in September - November, while the highest removal efficiency (70%) was recorded in December - February. Likeswise, the lowest (64.5%) and highest (76.6%) performance of the VFCW system was obtained in September - November and December - February, respectively. The hybrid system showed the lowest performance (68.1%) in September - November and the highest value in March - May. However, the different systems did not show significant difference in nitrate removal from season to season. The one-way ANOVA test of the horizontal, vertical and hybrid systems were $F_{0.95}$ (3, 20) = 0.772; P > 0.05, $F_{0.95}$ (3, 20) = 0.580; P > 0.05 and $F_{0.95}$ (3, 20) = 0.655; P > 0.05, respectively.

The effluent concentration values met the provisional discharge standard value set by Ethiopian Environmental Protection Authority (EEPA, 2003). The discharge standard for nitrate was 20 mg/L.

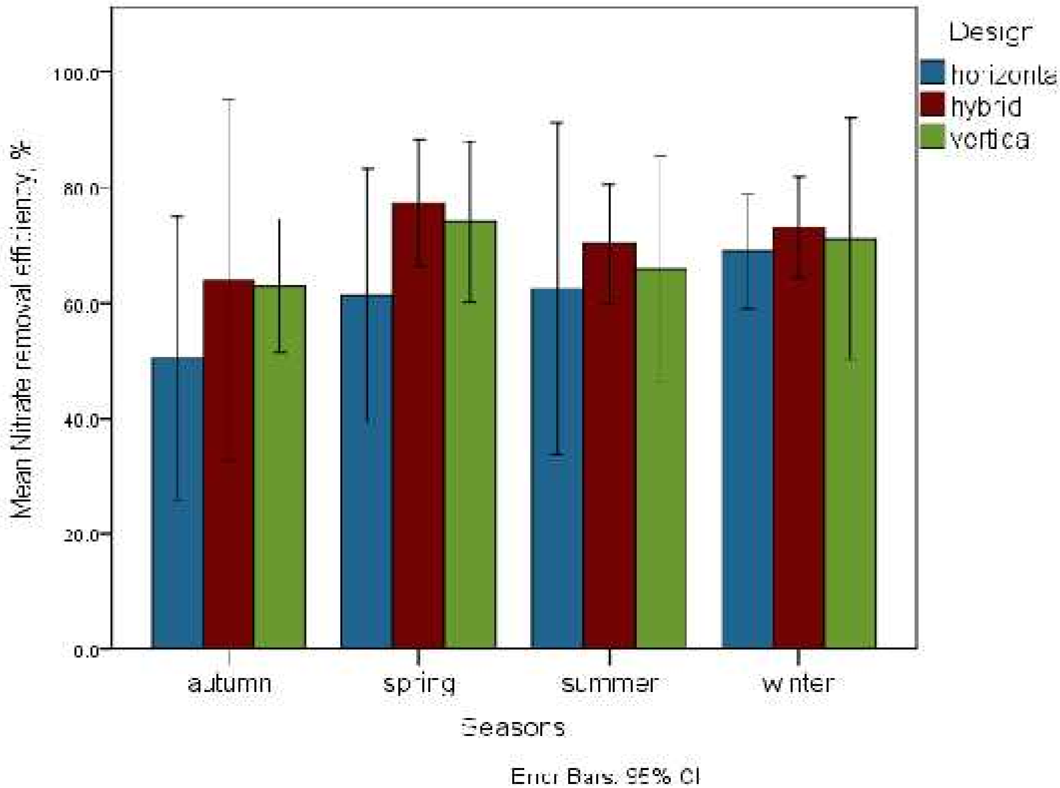

Figure 4.14: Seasonal NO_3^- removal efficiencies of the horizontal, vertical and hybrid pilot scale constructed wetland systems.

Table 4.13 indicates the percent reduction of NO_3^- by the subsurface flow constructed wetland systems applied in other countries. The result of this study was in agreement with NO_3^- removal percentages reported in Pakistan (Sehar *et al*, 2015), India (Rai *et al*, 2015) and Malaysia (Siti *et al*, 2011). Despite this, negative removal efficiency, NO_3^- concentration increament in the effluents was reported in Ireland (Kayranli *et al*, 2010) and UK (Al-Isawi *et al*, 2017).

Table 4-13: Comparison of current NO_3^- removal efficiencies to subsurface flow constructed wetlands applied in other countries.

Country	**NO_3^- Removal Efficiency, %**	**References**
Pakistan	70.3 - 80.4	Sehar *et al*, 2015
India	63.1 - 71.1	Rai *et al*, 2015
Malaysia	65	Siti *et al*, 2011
Ireland	-11.8	Kayranli *et al*, 2010
UK	Negative	Al-Isawi *et al*, 2017
Ethiopia	64 - 73.5	This study

The concentrations of nitrate in the influent of the CW systems used in this study were relatively high although the condition in the sewerage system was expected to be anaerobic. This might have occurred as a result of the surface runoff from the surrounding agricultural area, which creates the chance for nitrates to get into the sewerage system. The other possible reason could be the occurrence of natural re-aeration in the sewerage system while the domestic wastewater flows through it.

Environmental factors are known to influence denitrification rates and temperature is one of the key factors within the CWs system (Robert, 2004). As in other biological processes, growth rates in aquatic plant systems depend on temperature and the vegetated system show a much better performance during the warmer months of the year (Wang and Li, 2014). Lee *et al* (2013) also pointed out that the dependence of removal efficiency on temperature is significant due to plant uptake, which plays a significant role in nutrient removal. Likewise greater bacterial activity is shown during the warmer season than the colder one (Chon and Cho, 2015). So, warmer climate improves performances, especially for nitrification (Masi *et al*, 2013; Molle *et al*, 2015).

Despite this fact, the percent reduction of NO_3^- did not show significant difference as of the change of seasons in this study, although the temperature was significantly different from season to season. This could be related with the fact that the range of mean water temperature during the study period (13.1^oC - 22.4 ^oC) usually exceeds, the minimum temperature required for nitrate production. Prochaska *et al*, (2007) described that the minimum temperatures for nitrates production are 4 - 5 ◦C.

The correlation between the loading rate of NO_3^- and removal rate is presented in Figure 4.15. Based on the monitoring result, the loading rate of NO_3^- varied between 0.15 and 0.64 g/m^2.d (0.33 ± 0.133 g/m^2.d). The correlation between loading rate and removal rate of the vertical system (R^2 = 0.855) and the hybrid system (R^2 = 0.826) showed strong correlation than the horizontal system (R^2 = 0.641). So, it is possible to conclude that the vertical and the hybrid CWs had stronger capacity to hold and treat nitrate as the concentration increases.

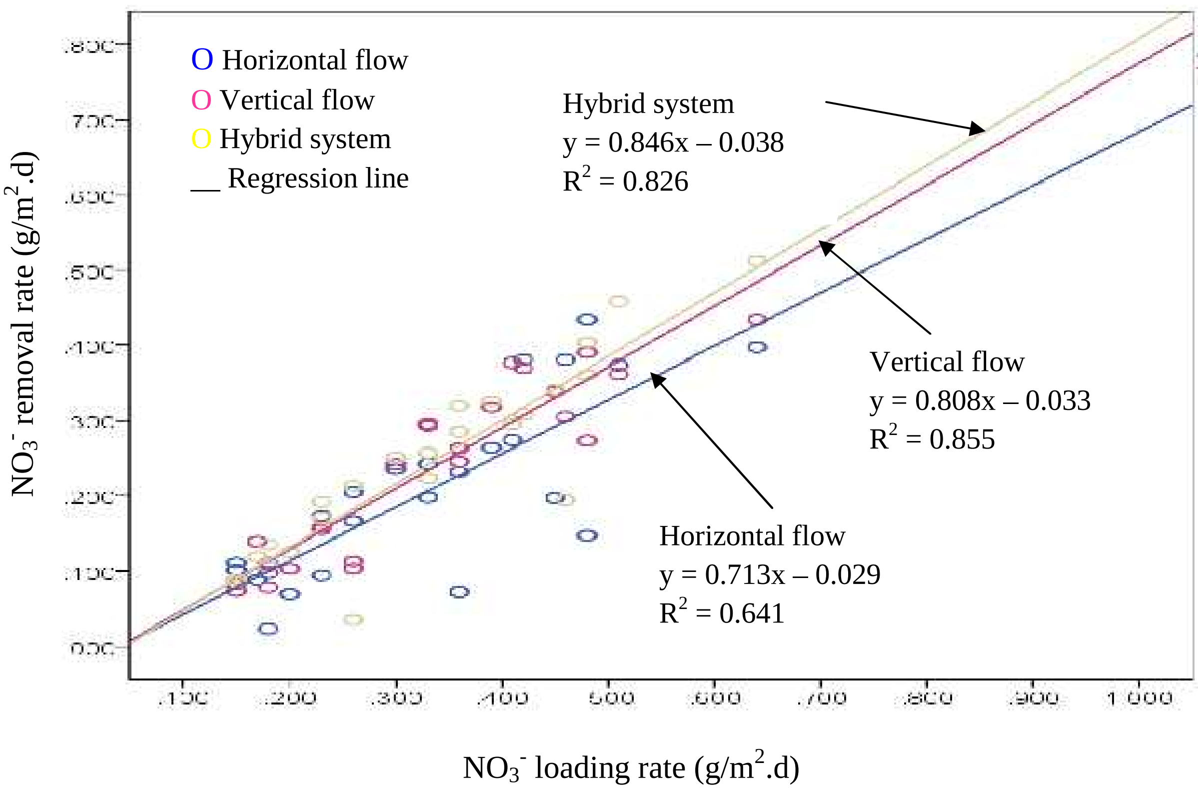

Figure 4.15: NO_3^- loading rate (g/m^2.d) against NO_3^- removal rate (g/m^2.d) of the pilot scale constructed wetland systems.

4.8 Removal of Total Nitrogen (TN)

As is shown in Figure 4.16 and Table 4.14, the average TN values in the influent of the horizontal, vertical and hybrid CW systems were in the range of 67.3 ± 3.0 mg/L - 87.2 ± 4.0 mg/L, with the average value of 76.6 ± 9.16 mg/L. Similarly, the average concentrations of TN in the outlets during the monitoring period were 35.2 - 43.2 mg/L (mena 38.8 ± 3.4 mg/L) for the horizontal, 32.5 – 36.2 mg/L (mean 34.2 ± 1.7 mg/L) for the vertical and 29.5 – 32.9 mg/L (mean 31.4 ± 1.5 mg/L) for the hybrid flow constructed wetland systems, respectively. The lowest TN concentration (29.5 mg/L) was recorded in the effluent of the hybrid flow CW system in March - May, while the HFCW system showed the highest concentration (43.2 mg/L) of TN in December - February.

Table 4-14: The mean TN concentration (mg/L) and removal efficiencies (%) of the HF, VF and hybrid flow pilot scale CW systems.

Seasons of the year	**Horizontal flow CW**			**Vertical flow CW**			**Hybrid flow CW**		
	Influent TN	Effluent TN	RE (%)	Influent TN	Effluent TN	RE (%)	Influent TN	Effluent TN	RE (%)
Dec-Feb	87.2 ± 4.0	43.2 ± 1.6	50.5	87.2 ± 4.0	36.2 ± 3.7	58.5	87.2 ± 4.0	32.2 ± 3.1	63.1
Mar-May	80.8 ± 12.0	37.6 ± 5.3	53.4	80.8 ± 12.0	32.5 ± 5.0	59.8	80.8 ± 12.0	29.5 ± 2.2	63.5
Jun-Aug	67.3 ± 3.0	35.2 ± 6.2	47.8	67.3 ± 3.0	33.4 ± 5.5	50.4	67.3 ± 3.0	30.8 ± 6.8	54.3
Sep-Nov	71.1 ± 4.4	39.3 ± 4.3	44.8	71.1 ± 4.4	34.9 ± 4.5	50.9	71.1 ± 4.4	32.9 ± 3.6	53.7
Annual	76.6 ± 9.1	38.8 ± 3.4	49.1	76.6 ± 9.1	34.2 ± 1.7	54.9	76.6 ± 9.1	31.4 ± 1.5	58.7

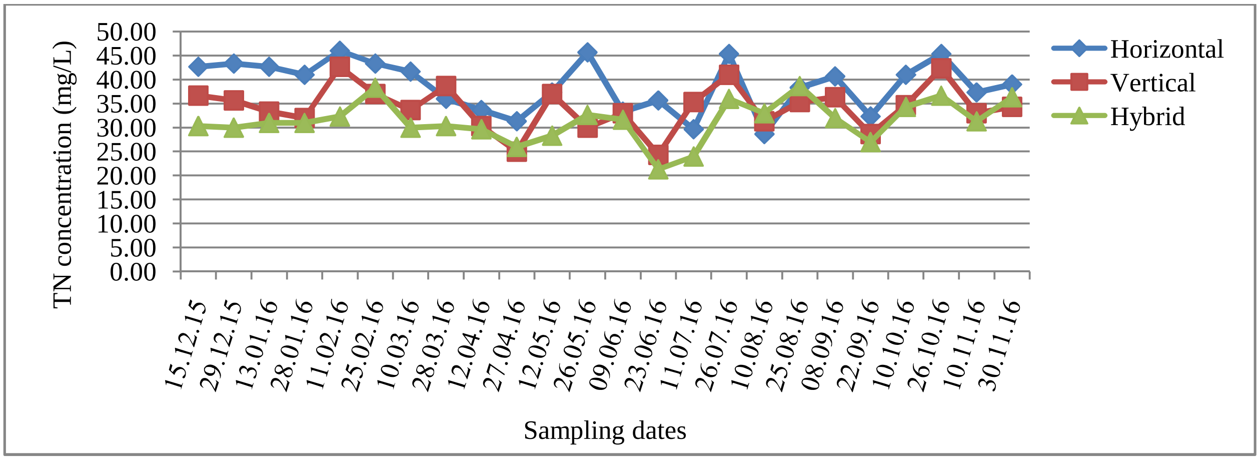

Figure 4.16: Concentration of TN of the effluents of the horizontal, vertical and hybrid pilot scale constructed wetland systems vs the sampling dates.

The average TN removal efficiencies of the pilot scale horizontal, vertical and the hybrid flow CW systems based on the concentration values were evaluated and the results were in the range of 44.8 - 53.4 % (49.1 %), 50.4 - 59.8 % (54.9 %) and 53.7 - 63.5% (58.7 %), respectively. The lowest performance was shown by the horizontal flow CW sytem while the hybrid flow CW sytem showed the highest performance (Figure 4.17 and Table 4.14). The result of one-way ANOVA also indicated that the removal efficiencies of the systems in removing TN were significantly different from one another statistically (one-way ANOVA; $F_{0.95}$ (2, 69) = 7.543; $P < 0.05$).

As to the seasonal average percent reduction of TN, the lowest value of the HFCW system was 44.8%, recorded in September - November while the highest value, 53.4% was obtained in March - May. In case of the VFCW system, the lowest, 50.4% and the highest, 59.8% removal efficiencies were recorded in June - August and March - May, respectively. Similarly, the hybrid flow system showed the lowest TN removal percentage, 53.7% in September - November and the highest removal percentage, 63.5% in March - May. None of the three systems showed significant difference from season to season in TN removal with one-way ANOVA results of the horizontal, vertical and hybrid systems were $F_{0.95}$ (3, 20) = 0.772; $P > 0.05$, $F_{0.95}$ (3, 20) = 0.580; $P > 0.05$ and $F_{0.95}$ (3, 20) = 0.655; $P > 0.05$, respectively.

The TN concentration values within the effluents of the three CWs system, indicating the effluent quality of the pilot-scale constructed wetland systems fulfills the discharge limit (60 mg/L) to the environment or water bodies (EEPA, 2003).

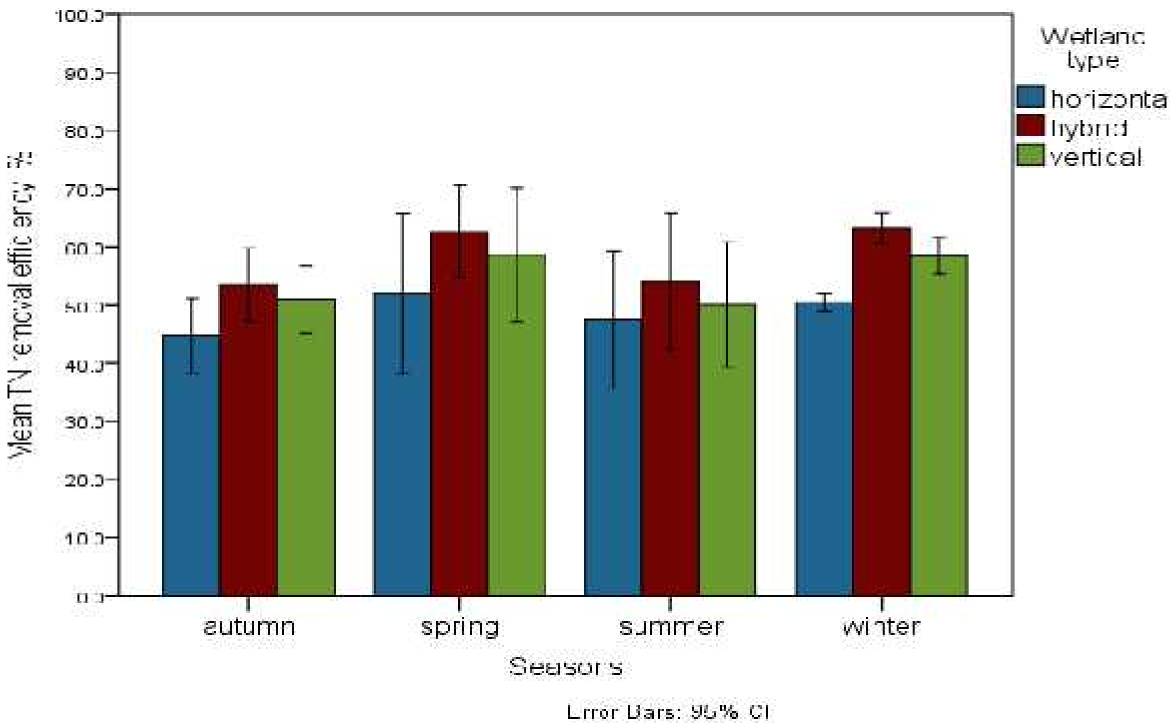

Figure 4.17: Seasonal TN removal efficiencies of the horizontal, vertical flow and hybrid flow pilot scale constructed wetland systems.

The average TN removal percentage was compared with similar works in other countries (Table 4-15). Accordingly, the result was higher than the removal efficiencies reported in Kenya (Mburu *et al*, 2013) and USA (Chavan *et al*, 2008), whereas the result was lower than the removal efficiencies reported in India (Deeptha, 2015) in South Korea (Kim *et al*, 2016) and Tunisia (Ghrabi, 2011). The performance of the pilot scale constructed wetland systems applied in this study was relatively similar with the value of reported in China (Meng *et al*, 2015).

In the same way, the percent reduction of the CW systems applied at Kotebe WWTP was low during summer (the rainy season) and autumn although the seasonal performance differences were not significant statistically. In a study done in South Korea, Kim *et al* (2016) pointed out that the TN removal efficiency of CWs in winter season (66.4%) was generally lower than that in summer (75.2%), autumn (72.6%), and spring (73.4%).

Table 4-15: Comparison of current TN removal efficiencies to subsurface flow constructed wetlands applied in other countries.

Country	**TN Removal Efficiency, %**	**References**
India	76	Deeptha, 2015
South Korea	66.4 – 75.2	Kim *et al*, 2016
Tunisia	71	Ghrabi *et al*, 2011
China	63.4	Lu *et al*, 2015
China	43	Guo *et al*, 2014
China	39 – 54	Meng *et al*, 2015
USA	24 – 47	Chavan *et al*, 2008
Kenya	8	Mburu *et al*, 2013
Ethiopia	49.1 – 58.7	This study

The loading rate of TN were found to be in the range of 2.54 and 4.08 $g/m^2.d$ (3.37 ± 0.452 $g/m^2.d$) (Fig 4.18) during the monitoring period. The linear correlation of TN loading rate and TN removal rate for the hybrid system (R^2 = 0.855) was relatively the same with the correlation by the vertical flow system (R^2 = 0.823), while the correlation by the horional flow CW system (R^2 = 0.752) was slightly lower than the vertical flow and hybrid flow CW system. So, as the loading rate increases, the hybrid flow and the vertical flow system could have relatively better capacity to hold and remove TN in the constructed wetland systems.

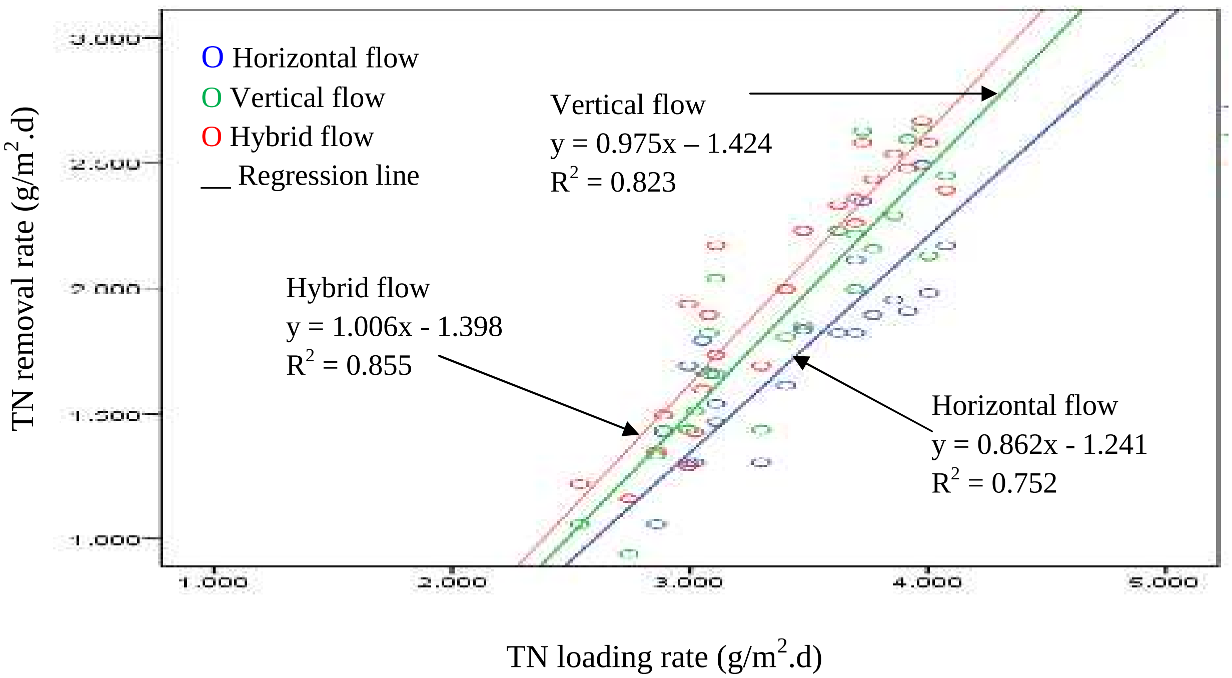

Figure 4.18: TN loading rate (g/m^2.d) against the TN removal rate (g.m^2.d) of the pilot scale constructed wetland systems.

4.9 Removal of Phosphate (PO_4^{3-})

The average inlet and outlet PO_4^{3-} concentrations of the pilot scale constructed wetland systems are presented in Figure 4.19 and Table 4.16. Accordingly, the average inlet PO_4^{3-} concentration of the horizontal, vertical and hybrid constructed wetland systems varied between 4.09 - 7.67 mg/L (mean 5.84 ± 1.76 mg/L). The concentration of PO_4^{3-} in the influent was relatively high in December - February and March - May, while it was low in June - August and September - November.

The PO_4^{3-} concentrations of the effluents were found to be in the range of 1.96 - 4.55 mg/L (mean 3.24 ± 1.26 mg/L) from the horizontal bed, 1.84 - 4.29 mg/L (mean 2.98 ± 1.24 mg/L) from the vertical bed and 1.81 - 4.33 mg/L (mean 3.1 ± 1.33 mg/L) from the hybrid bed, respectively.

Table 4-16: The mean PO_4^{3+} concentration (mg/L) and removal efficiencies (%) of the HF, VF and hybrid flow pilot scale CW systems.

Seasons of the year	**Horizontal flow CW**			**Vertical flow CW**			**Hybrid flow CW**		
	Influent PO_4^{3-}	Effluent PO_4^{3-}	RE (%)	Influent PO_4^{3-}	Effluent PO_4^{3-}	RE (%)	Influent PO_4^{3-}	Effluent PO_4^{3-}	RE (%)
Dec-Feb	7.00 ± 0.9	4.06 ± 0.9	42.0	7.00 ± 0.9	3.77 ± 0.8	46.1	7.00 ± 0.9	4.16 ± 0.9	40.6
Mar-May	7.67 ± 0.5	4.55 ± 0.7	40.6	7.67 ± 0.5	4.29 ± 0.4	44.0	7.67 ± 0.5	4.33 ± 0.6	43.5
Jun-Aug	4.09 ± 1.6	2.40 ± 0.5	41.4	4.09 ± 1.6	2.01 ± 0.7	51.0	4.09 ± 1.6	2.10 ± 0.4	48.6
Sep-Nov	4.60 ± 1.3	1.96 ± 0.8	57.4	4.60 ± 1.3	1.84 ± 0.9	59.9	4.60 ± 1.3	1.81 ± 0.9	60.8
Annual	5.84 ± 1.76	3.24 ± 1.26	45.4	5.84 ± 1.76	2.98 ± 1.24	50.3	5.84 ± 1.76	3.10 ± 1.33	48.4

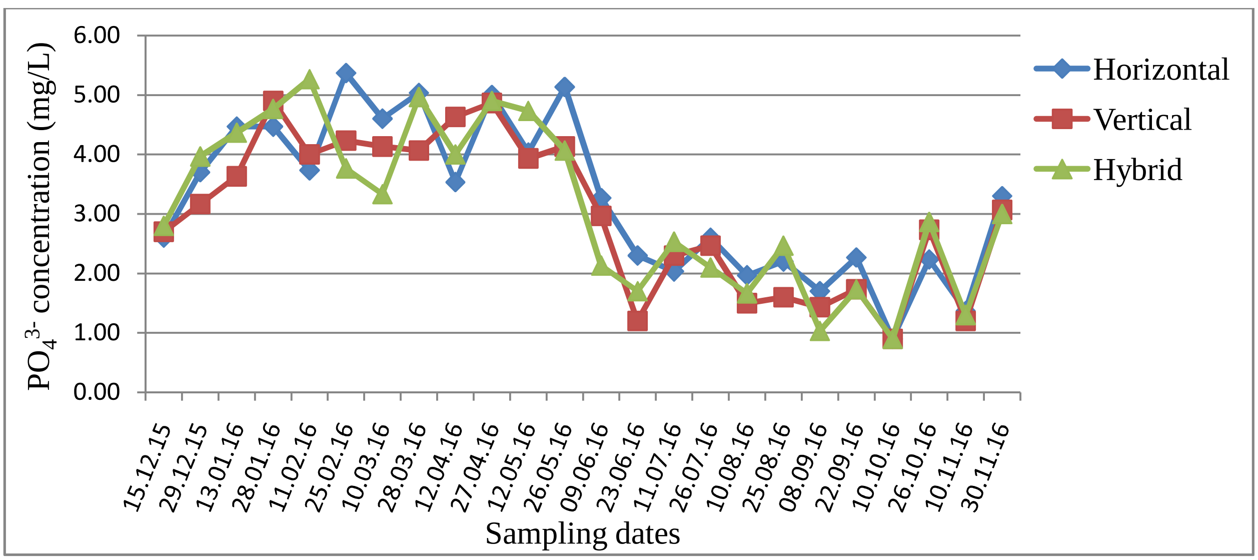

Figure 4.19: Concentration of PO_4^{3-} of the effluents of the horizontal, vertical and hybrid pilot scale constructed wetland systems vs the sampling dates.

The average concentration based PO_4^{3-} removal efficiencies of the horizontal, vertical and hybrid CW systems used for the treatment of domestic wastewater were 45.4 ± 8.0 %, 50.3 ± 7.1 % and 48.4 ± 8.9 %, respectively. The the lowest removal percentage for the HF and the highest removal percentage for the VF system were obtained during the monitoring period (Figure 4.20 and Table 4.16). In the vertical flow CW system, better macrophytes growth was seen during the monitoring period. However, the performances of the HF, VF and hybrid flow systems in removing PO_4^{3-} were not significantly different from one another statistically (one-way ANOVA; $F_{0.95}$ (2, 69) = 0.855; $P > 0.05$).

With regard to seasonal performance, the PO_4^{3-} removal efficiencies of HFCW system showed the minimum and maximum percent reduction of 40.6% in March - May and 57.4% in September - November, respectively. Similarly, the minimum and maximum values in case of the VFCW system were 44.0% in March - May and 59.9% in September - November, respectively. The hybrid flow system showed the minimum and maximum PO_4^{3-} percent reductions of 40.6% in December - February and 60.8% in September - November, respectively.

The removal efficiencies of the three constructed wetlands were compared using one-way ANOVA; and the results of the horizontal, vertical and hybrid systems were $F_{0.95}$ (3, 20) = 2.069; $P > 0.05$, $F_{0.95}$ (3, 20) = 1.746; $P > 0.05$ and $F_{0.95}$ (3, 20) = 2.381; $P > 0.05$, respectively. Even though, the performance of the CWs system did not show significant difference with change of seasons, Prachaska *et al* (2007) revealed that seasons have an effect on orthophosphates removal.

The concentration values of PO_4^{3-} were in compliance with the provisional discharge standards (5 mg/L) set by Ethiopian Environmental Protection Authority (EEPA, 2003).

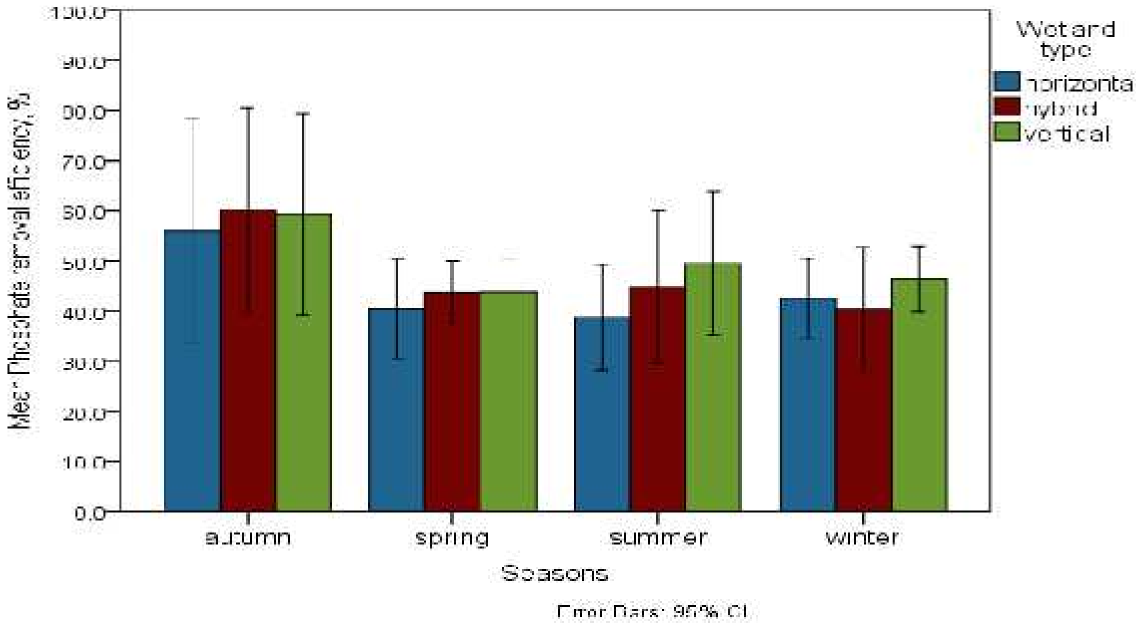

Figure 4.20: Seasonal PO_4^{3-} removal efficiencies of the horizontal, vertical and hybrid pilot scale constructed wetland systems.

The PO_4^{3-} removal efficiencies of subsurface flow constructed wetland systems achievable in some other countries are presented in Table 4.17. The removal percentage obtained in this study was lower than the removal efficiencies reported in Colombia (Caselles-Osorio *et al*, 2011), Tunisia (Ghrabi *et al*, 2011), India (Rai *et al*, 2015) and Pakistan (Sehar *et al*, 2015). But similar percent reductions of PO_4^{3-} were reported in Pakistan (Mustafa, 2013) and Egypt (Abou-Elela *et al*, 2014).

Table 4-17: Comparison of current PO_4^{3-} removal efficiencies to subsurface flow constructed wetlands applied in other countries.

Country	**PO_4^{3-} removal efficiency, %**	**References**
Ireland	91.8	Mustafa, 2013
Colombia	85	Caselles-Osorio *et al*, 2011
Tunisia	82	Ghrabi *et al*, 2011
India	58.3 - 77.5	Rai *et al*, 2015
Pakistan	58.8 - 73.7	Sehar *et al*, 2015
UK	59.8 - 64.7	Al-Isawi *et al*, 2017
Pakistan	52	Mustafa, 2013
Egypt	44	Abou-Elela *et al*, 2014
Ethiopia	45.4 - 50.3	This study

The PO_4^{3-} loading rate at the time of performance monitoring varied in the range between 1.31 and 0.364 g/m^2.d (0.257 ± 0.083 g/m^2.d) (Fig 4.21). The correlation between loading rate and removal rate by the horizontal, vertical and hybrid system showed that the PO_4^{3-} removal rate was dependent on the PO_4^{3-} loading rate. The removal rate for vertical flow system (R^2 = 0.605) was slightly more dependent on loading rate than the horizontal system (R^2 = 0.543) and the hybrid system (R^2 = 0.490); while the correlation between removal rate and loading rate in the horizontal system is slightly stronger than the hybrid system.

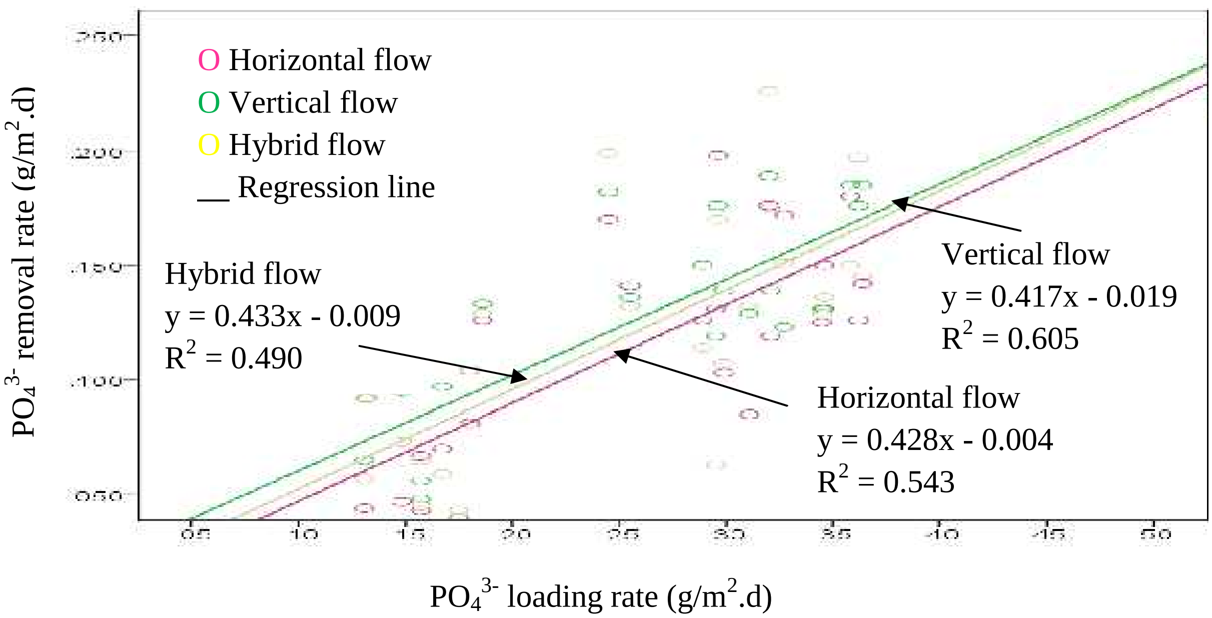

Figure 4.21: PO_4^{3-} loading rate (g/m^2.d) against the PO_4^{3-} removal rate (g/m^2.d) of the pilot scale constructed wetland systems.

4.10 Removal of Total Phosphorous (TP)

The average TP concentrations in the influent and effluents and the average removal efficiencies of the pilot scale constructed wetland systems are given in Figure 4.22 and Table 4.18. During the monitoring period, the average concentrations of the influent which fed the pilot scale constructed wetland systems ranged from 4.77 mg/L - 10.41 mg/L (mean 7.42 ± 2.46 mg/L). The highest concentration of TP in the influent was obtained in March - May, while the lowest concentration was recorded in June - August.

The average TP concentrations in the effluents of the horizontal, vertical and hybrid flow systems were in the range of 2.07 - 3.92 mg/L (mean 3.06 ± 0.77 mg/L), 2.39 - 4.83 mg/L (mean

3.52 ± 1.01 mg/L) and 2.16 - 4.63 mg/L (mean 3.35 ± 1.02 mg/L), respectively. The lowest concentration of TP was obtained in the effluent of the HFCW system in June - August, while the highest concentration value was recorded in the effluent of the VFCW system in March - May.

Table 4-18: The mean TP concentration (mg/L) and removal efficiencies (%) of the HF, VF and hybrid flow pilot scale CW systems.

Seasons of the year	**Horizontal flow CW**			**Vertical flow CW**			**Hybrid flow CW**		
	Influent TP	Effluent TP	RE (%)	Influent TP	Effluent TP	RE (%)	Influent TP	Effluent TP	RE (%)
Dec-Feb	8.28 ± 2.7	3.26 ± 1.0	60.6	8.28 ± 2.7	3.59 ± 1.4	56.6	8.28 ± 2.7	3.49 ± 1.4	58.0
Mar-May	10.41 ± 2.2	3.92 ± 0.8	62.4	10.41 ± 2.2	4.83 ± 1.6	53.6	10.41 ± 2.2	4.63 ± 1.1	55.5
Jun-Aug	4.77 ± 1.2	2.07 ± 0.8	56.7	4.77 ± 1.2	2.39 ± 0.7	49.9	4.77 ± 1.2	2.16 ± 0.7	54.7
Sep-Nov	6.21 ± 1.6	2.97 ± 1.2	52.2	6.21 ± 1.6	3.28 ± 1.7	47.2	6.21 ± 1.6	3.13 ± 1.4	49.6
Annual	7.42 ± 2.46	3.06 ± 0.77	58	7.42 ± 2.46	3.52 ± 1.01	51.8	7.42 ± 2.46	3.35 ± 1.02	54.5

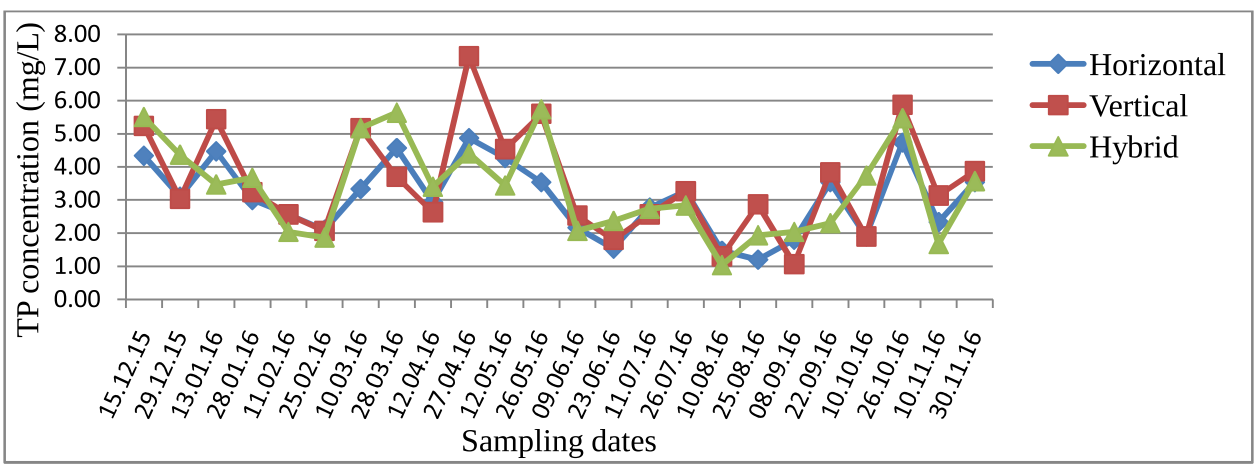

Figure 4.22: Concentration of TP of the effluents of the horizontal, vertical and hybrid pilot scale constructed wetland systems vs the sampling dates.

The concentration based removal efficiencies of the horizontal, vertical and hybrid CWs in removing TP were 58.0 ± 4.53%, 51.8 ± 4.13% and 54.5 ± 3.53%, respectively. The performance of the horizontal, vertical and hybrid flow systems in removing TP was not significantly different from one another (one-way ANOVA; $F_{0.95}$ (2, 69) = 2.436; P > 0.05).

The seasonal TP removal efficiencies of the pilot scale constructed wetland systems showed the minimum and maximum percent reduction of TP by the HFCW system were 52.2% in September - November and 62.4% in March - May, respectively. The VFCW system showed a minimum removal efficiency of 47.2% in September - November and the maximum removal efficiency was 56.6% in December - February, whereas the minimum and maximum removal efficiencies obtained within the effluents of the hybrid flow CW system were 49.6% in September - November and 58.0% in December - February, respectively. However, they did not differ significantly from season to season in removing TP; one-way ANOVA results of the horizontal, vertical and hybrid systems were $F_{0.95}$ (3, 20) = 0.882; P > 0.05, $F_{0.95}$ (3, 20) = 0.417; P > 0.05 and $F_{0.95}$ (3, 20) = 0.657; P > 0.05, respectively.

The TP of the effluents of the pilot scale constructed wetland systems met the provisional discharge standards of 10 mg/L which was set by Ethiopian Environmental Protection Authority (EEPA, 2003) to discharge the effluents to the environment or water bodies.

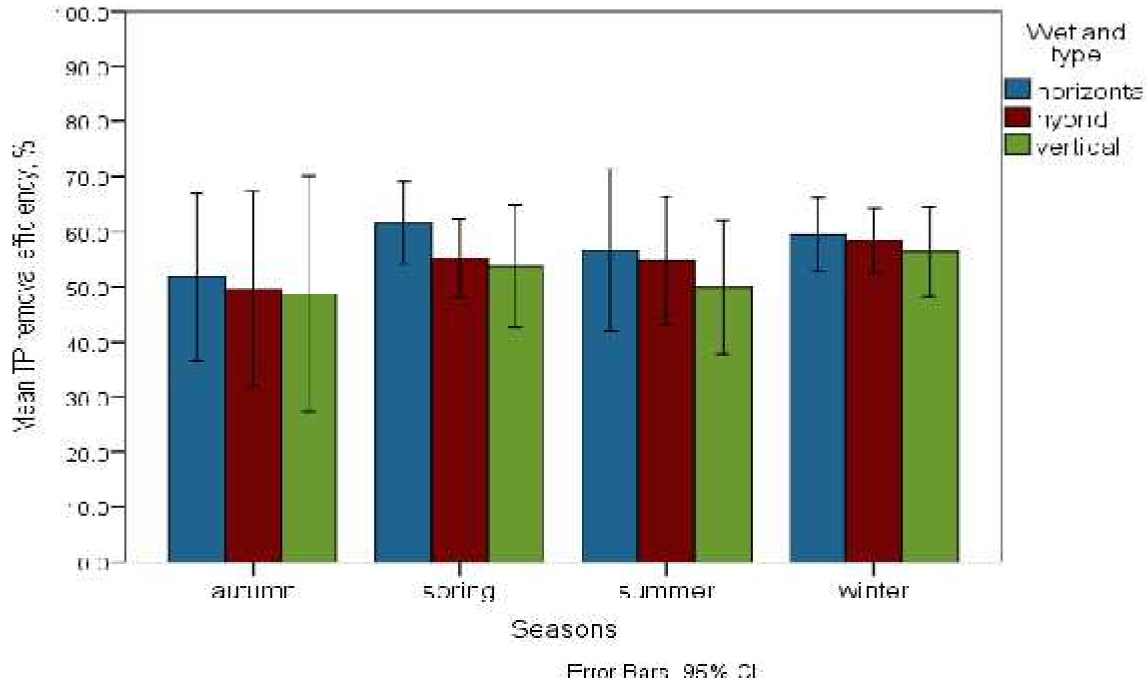

Figure 4.23: Seasonal TP removal efficiencies of the horizontal, vertical and hybrid pilot scale constructed wetland systems.

Table 4.19 indicates the TP removal efficiencies of the subsurface flow constructed wetland systems applied in some countries. Accordingly, the removal efficiency of the CW system in this study was lower than the ones reported from China (Guo *et al*, 2014; Lu *et al*, 2015), Colombia (Caselles-Osorio *et al*, 2011), India (Deeptha *et al*, 2015) and USA (Menon and Holland, 2013). However, the removal percentage was higher than the values reported in China (Meng *et al*, 2015), Kenya (Mburu *et al*, 2013) and Turkey (Ayaz *et al*, 2012).

The removal of TP varied between 40 and 60% in all types of CWs depending on the type of CWs and inflow loading and P removal is mainly influenced by wetland substrate (Vymazal, 2007; Li *et al*, 2010). Siti *et al* (2011) described that the use of specialized media in CWs to improve P removal should be developed and demonstrated since P removal always shows worse performance in the wetlands. The relatively low removal efficiency in this study could be related with the application of gravel as fill media since gravel could not be considered as a good P-adsorption wetland media (Vymazal, 2005). Removal mechanisms such as filtration, plant uptake and biological assimilation could also be the main ones in removing phosphorous in this study.

Table 4-19: Comparison of current PO_4^{3-} removal efficiencies to subsurface flow constructed wetlands applied in other countries.

Country	TP Removal Efficiency, %	References
China	92.6	Lu *et al*, 2015
Colombia	85	Caselles-Osorio *et al*, 2011
India	73	Deeptha *et al*, 2015
USA	70	Menon and Holland, 2013
China	70	Guo *et al*, 2014
China	36 - 43	Meng *et al*, 2015
Kenya	26	Mburu *et al*, 2013
Turkey	90 (VFCW), and < 20 (HFCW)	Ayaz *et al*, 2012
Ethiopia	51.8 - 58	This study

During the monitoring period, the loading rate of TP ranged between 0.145 and 0.573 $g/m^2.d$ (0.326 ± 0.127 $g/m^2.d$) (Fig 4.24). The correlation between loading rate and removal rate by the horizontal, vertical and hybrid system showed that the TP removal rate was dependent on the TP loading rate; and the correlation for horizontal system (R^2 = 0.902) was slightly stronger than the hybrid system (R^2 = 0.837). On the other hand, the correlation for both the horizontal and the hybrid system were to some extent stronger than the vertical system (R^2 = 0.743). The difference might be attributed to the type of wastewater flow into each wetland cell and the hydraulic condition of the system as the pilot scale CWs were made to function at similar situations.

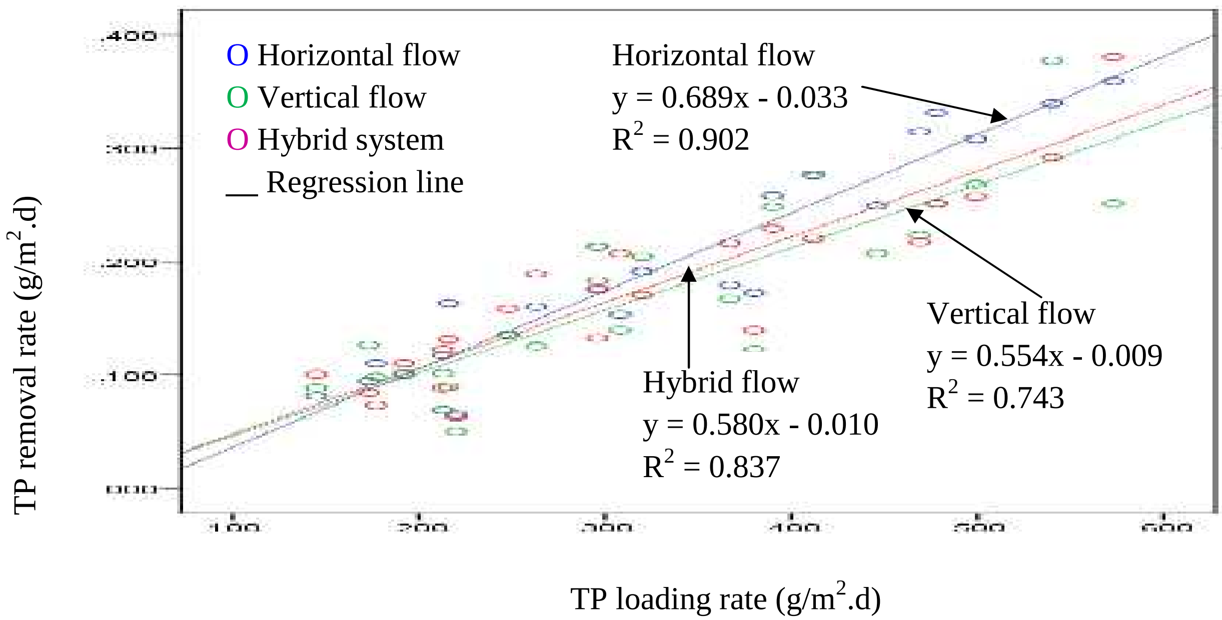

Figure 4.24: TP loading rate (g/m^2.d) against the TP removal rate (g/m^2.d) of the pilot scale constructed wetland systems.

4.11 Removal of Fecal Coliform (FC)

The removal of faecal coliforms (FC) of the constructed wetland system is presented in Figure 4.25 and Table 4.20. The average concentration of FC in the inlet of the CWs system used at Kotebe WWTP to treat domestic wastewater varied from 81,167 - 110,778 CFU/100 ml (mean 95,292 ± 13,190 CFU/100 ml). The average concentration of FCs obtained in this study was noticeably lower compared to the value, 92.6 x 10^{10} ± 49.4 x 10^4 CFU/100 ml, reported in Tanzania (Mahenge, 2014), while CFU count was similar with the one reported in Colombia (Caselles-Osorio *et al*, 2011).

Similarly, the average FC counts of the effluent from the horizontal, vertical and hybrid CWs system varied between 1,317 - 2,410 CFU/100 ml (mean 2,010 ± 477 CFU/100 ml); 2,133 - 4,183 CFU/100 ml (mean 3,250 ± 927 CFU/100 ml) and 2,611 - 4,545 CFU/100ml (3,353 ± 834 CFU/100ml), respectively.

Table 4-20: The mean FC concentration (CFU/100ml) and removal efficiencies (%) of the HF, VF and hybrid flow pilot scale CW systems.

Seasons of the year	**Horizontal flow CW**			**Vertical flow CW**			**Hybrid flow CW**		
	Influent FC	Effluent FC	RE (%)	Influent FC	Effluent FC	RE (%)	Influent FC	Effluent FC	RE (%)
Dec-Feb	101000± 26393	2410± 1672	97.6	101000± 26393	3811± 1304	96.2	101000± 26393	3217± 2087	96.8
Mar-May	110778± 21089	2167± 494	98.0	110778± 21089	4183± 1091	96.2	110778± 21089	4545± 643	95.9
Jun-Aug	81167 ± 4839	1317± 260	98.4	81167± 4839	2133± 5834	97.4	81167± 4839	2611± 481	96.8
Sep-Nov	88222± 9708	2145± 482	97.6	88222± 9708	2872± 581	96.7	88222± 9708	3039± 994	96.6
Annual	95292 ± 13190	2010 ± 477	97.9	95292 ± 13190	3250 ± 927	96.6	95292 ± 13190	3353 ± 834	96.5

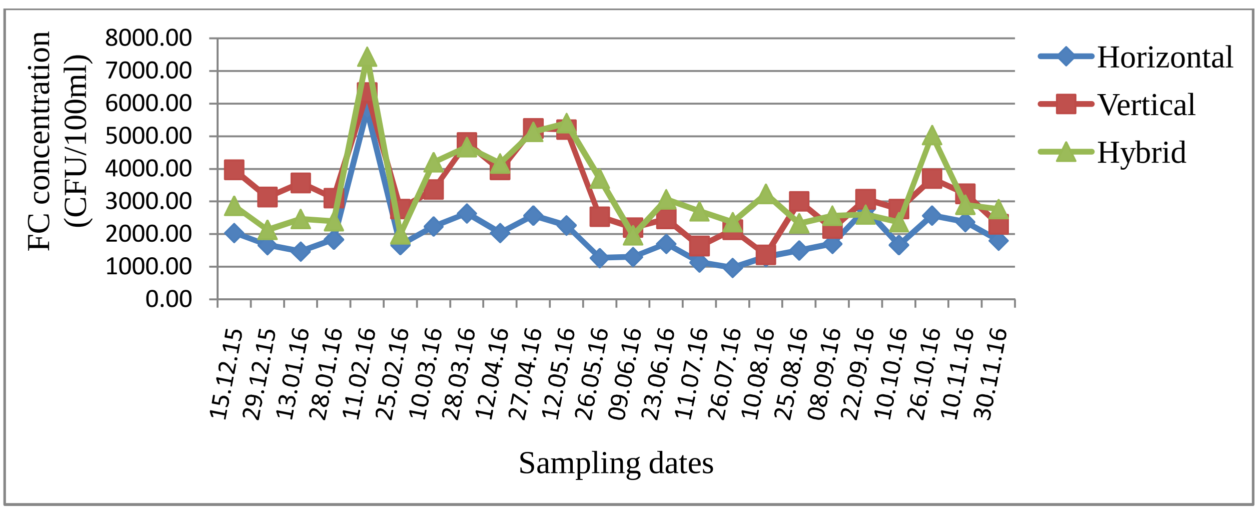

Figure 4.25: Concentration of FC of the effluents of the horizontal, vertical and hybrid pilot scale constructed wetland systems Vs the sampling dates.

Accordingly, the FC removal efficiencies of the horizontal, vertical and hybrid flow CW systems applied at Kotebe WWTP were 97.9% ± 0.38, 96.6% ± 0.57 and 96.5% ± 0.43, respectively. The performance of removal of FC of the different CW was significantly different from one another in removing FCs (one-way ANOVA; $F_{0.95}$ (2, 69) = 29.518; $P < 0.05$).

Regarding the seasonal performance, the horizontal flow CW had the minimum and maximum removal percentage of 97.6% in December - February and 98.4% in June - August, respectively; whereas the vertical flow CW system showed the minimum and maximum removal percentage of 96.2% in March - May and 97.4% in June - August, respectively. Similarly, the hybrid flow system showed the minimum removal efficiency of 95.9% in March - May and the maximum removal efficiency of 96.8% June - August. The seasonal variations in the performance of the FC removal of the the performances of both the horizontal and vertical flow systems differed significantly from season to season. While the hybrid flow system was not significantly different from season to season in removing FC; one-way ANOVA results of the horizontal, vertical and hybrid systems were $F_{0.95}$ (3, 20) = 3.125; $P < 0.05$, $F_{0.95}$ (3, 20) = 8.052; $P < 0.05$ and $F_{0.95}$ (3, 20) = 2.380; $P > 0.05$, respectively.

The data however showed that effluent FC was by far higher than even the discharge limit (400 CFU/100 ml) set by EEPA (2003). Even the concentration of FC of the CW in this study was high compared to the WHO guideline for the safe use of wastewater for unrestricted irrigation, ≤ 1,000 CFU/100ml (WHO, 2006).

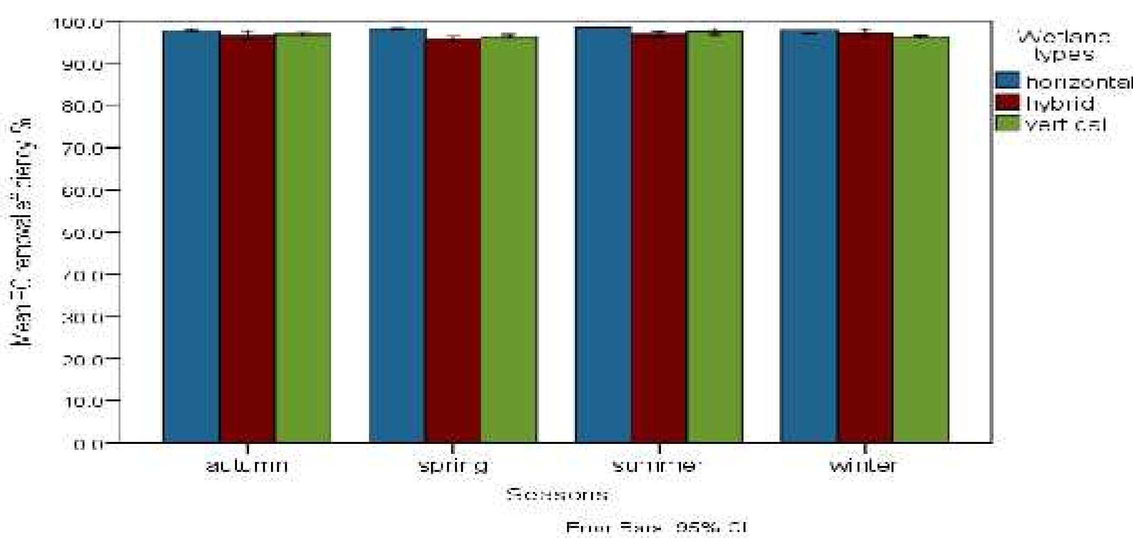

Figure 4.26: Seasonal FC removal efficiencies of the horizontal, vertical and hybrid pilot scale constructed wetland systems.

Table 4.21 shows the FC removal efficiency of subsurface flow CWs applied in other countries. The removal efficiency obtained in this study was comparable with the result reported in Colombia (Caselles-Osorio *et al*, 2011), Tanzania (Mahenge, 2014) and Pakistan (Mustafa, 2011). The result obtained in this study was lower than FC removal efficiency, 99.99%, reported from Egypt (Abou Elela *et al*, 2014).

The lower removal efficiency, in general, is attributed to low hydraulic retention time and there is a need to give adequate time (> 5-15 days) that is required to allow the system to operate more in a steady state condition for treatment of sewage to acceptable levels. In addition, it requires effective secondary and tertiary applications for the removal of pathogens such as FCs (Raymundo, 2008; Mahenge, 2014). Treatment wetlands show considerable potential for removing fecal bacteria from domestic wastewater (Sleytr *et al*, 2007; Fountoulakis *et al*, 2009; Vallejos *et al*, 2015).

In general, the FC removal efficiency of the pilot scale constructed wetland systems during the first monitoring period of this study was high. This could be due to well adaptation and growth of wetland plants and favorable environmental conditions. Macrophytes-based systems turned out to be a good alternative for wastewater treatment concerning bacterial removal and water quality. In contrast, those systems without plants show lower efficiencies than their corresponding planted wetlands. It is also found that mean removal efficiencies and surface removal rates turn out to be significantly high in wetlands, and some increases in removal efficiencies are associated with warm season (Garcia *et al*, 2008; Foladori *et al*, 2015; Wu *et al*, 2016). Zurita and Carreon-Alvarez (2015) pointed out that a wetland tends toward a better performance during the maturity period reached by the system, noticeable through the presence of well-developed macrophytes.

Table 4-21: Comparison of current FC removal efficiencies to subsurface flow constructed wetlands applied in other countries.

Country	FC Removal Efficiency, %	References
Egypt	99.9	Abou-Elela *et al*, 2014
Turkey	99.0	Tunsciper *et al*, 2012
Pakistan	98.0	Mustafa, 2013
Tanzaina	98.0	Mahenge, 2014
Colombia	96.0	Caselles-Osorio *et al*, 2011
Ethiopia	96.5 - 97.9	This study

This study shows that the FC loading rate was in the range between 3.168 x 10^7 - 6.527 x 10^7 CFU/m^2/d (mean 4.193 x 10^7 ± 8.91 x 10^6 CFU/m^2.d) (Fig 4.27). The linear correlation between FC loading rate and FC removal rate was high and almost similar among the three wetland systems. For that reason, the linear correlation was (R^2 = 0.999) for the horizontal flow wetland, (R^2 = 0.999) for the vertical flow wetland and (R^2 = 0.999) for the hybrid flow bed. Although the three systems showed almost the same linear correlation between FC loading rate and FC removal rate, the capacity to hold and remove FC was slightly better in the horizontal and vertical systems as the concentration increases.

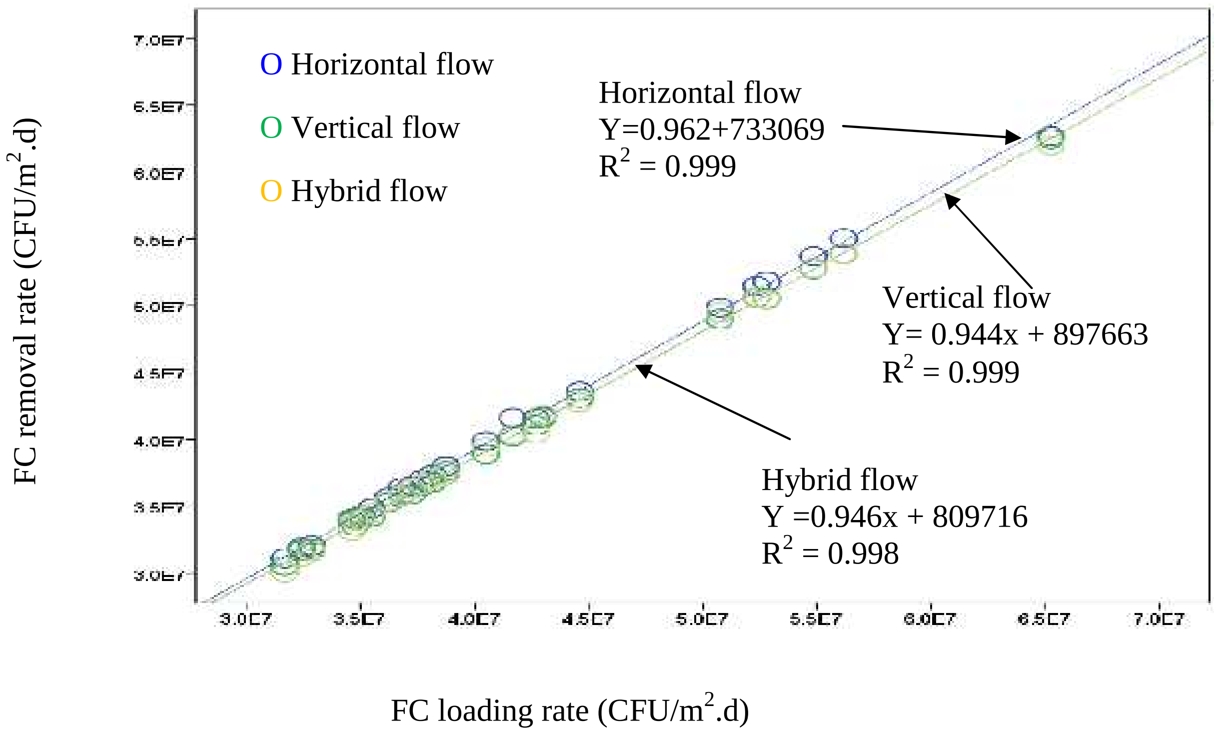

Figure 4.27: FC loading rate against FC removal rate of the horizontal, vertical and hybrid flow pilot scale constructed wetland systems.

4.12 Kinetic parameters determination

The first order plug flow equation (Equation 2.12 to 2.14) which was discussed in section 2.8 was used to calculate the areal removal rate constants. Accordingly, the rate constants for each of the parameters were calculated using the annual and seasonal mean concentrations of the influent, effluents of the horizontal, vertical and hybrid flow systems and the loading rate of the respective pollutants. The results are presented in Table 4.15 and Annex-Table 12.

The average values of the areal removal rate constants of wastewater pollutants showed that the values for BOD_5, TN, PO_4^{3-} - P and TP were higher than the literature values (Table 4.22) (Korkusuz *et al*, 2004; Kadlec *et al*, 2000; Konnerup *et al*, 2009) and the hybrid system showed increased areal removal rate constants except for TP. However, the literature values of the areal removal rate constants for COD and TSS were higher than the result obtained in this study (Zhang *et al*, 2014; Abdelhakeem *et al*, 2016). The areal removal rate constants of NH_4^+-N were almost the same among the three wetland systems and with the literature values. In the meantime, the values of areal removal rate constants for BOD, COD and TSS were higher than those of NH_4^+, TN, PO_4^{3-} and TP among all the three constructed wetland types, then confirming the low removal efficiency of CWs in removing nitrogen and phosphorous .

Moreover, the values of areal removal rate constants of each pollutant for the different seasons were calculated and compared (Annex-Table 12). Although the seasonal removal efficiencies of the pilot scale CW systems were statistically significant only for BOD_5, COD, and NH_4^+, the removal percentage values of most parameters were relatively higher in December - February and March - May, the seasons with relatively high temperature. In a similar manner, the areal removal rate constants of most pollutants were higher during those seasons (Annex - Table 12). This might indicate that temperature has an effect on the values of areal removal rate constants.

Table 4-22: Summary of areal removal rate constants, K (m/d) of the parameters considered in this study.

Parameters	**This study**			**K, literatures**	**References**
	K, HF*	K, VF**	K, HybF***		
BOD	0.098	0.113	0.121	0.077	Kadlec *et al*, 2000 as cited in Sheridan *et al*, 2014
COD	0.073	0.076	0.081	0.2	Abdelhakeem *et al*, 2016
TSS	0.097	0.080	0.084	0.82	Zhang *et al*, 2014
NH_4^+-N	0.040	0.048	0.048	0.05	Abdelhakeem *et al*, 2016
TN	0.030	0.036	0.039	0.0158	Konnerup *et al*, 2009
PO_4^{3-} - P	0.026	0.030	0.028	0.002	Korkusuz *et al*, 2004
TP	0.039	0.033	0.035	0.0149	Konnerup *et al*, 2009

* Horizontal Flow CW ** Vertical Flow CW *** Hybrid Flow CW

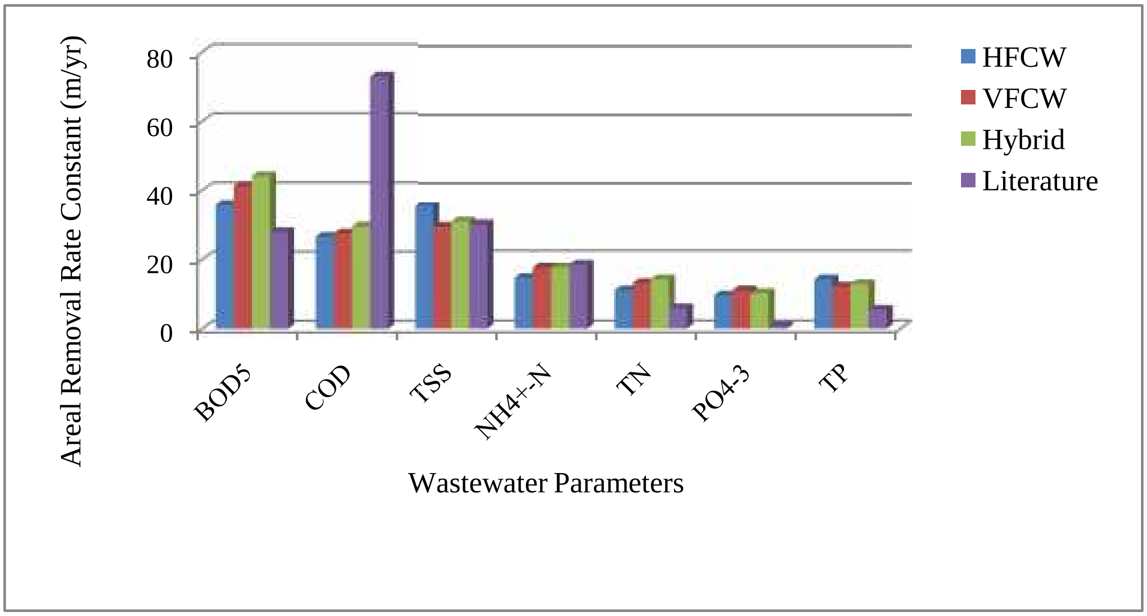

Figure 4.28: Comparison of the calculated and literature values of areal removal rate constants.

The pollutants concentration profiles decreased in an approximately exponential way through out the length of the treatment wetland system from the inlet to the outlet. The concentrations of pollutants followed this pattern over time with distance from inlet to outlet. Some pollutant concentrations decline to near-zero values while others level off to some background concentration (Kadlec et al, 2000). Theoretically, the wastewater parameters of first order model are known as 'rate constants'. So, ideally they are assumed not to be dependent on the concentrations of the inlet or loading rates (IWA, 2000). But the calculated areal removal rate constants usually increased in response to an increase of pollutants mass loading rates in this study. As is shown in Figure 4.29 - 4.35, the areal removal rate constants differed as a result of the variation of the mass loading rates. Additionally, the effect of the mass loading rate on the areal removal rate constants was not linear (Figure 4.29 - 4.35). This could have showed that the area of the wetland cell usually engaged in the process of pollutants removal was increased as the mass loading rates increased. Likewise, the different values of the areal removal rate constants of certain parameters might be linked to the effect of temperature as the reduction of some constituents was influenced by the water temperature (Kurkusuz *et al*, 2004).

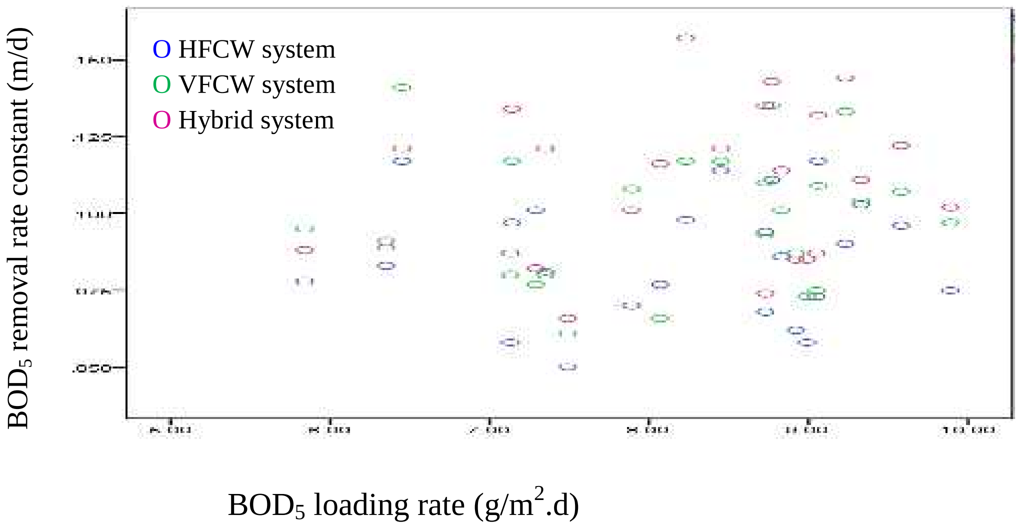

Figure 4.29: BOD_5 loading rate ($g/m^2.d$) versus areal removal rate constant (m/d) of the pilot scale constructed wetland systems.

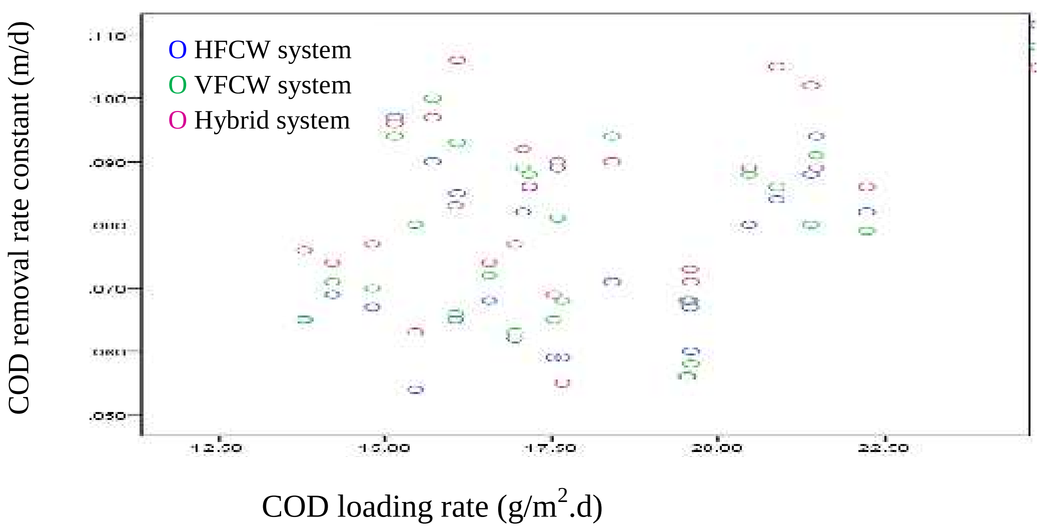

Figure 4.30: COD loading rate ($g/m^2.d$) versus areal removal rate constant (m/d) of the pilot scale constructed wetland systems.

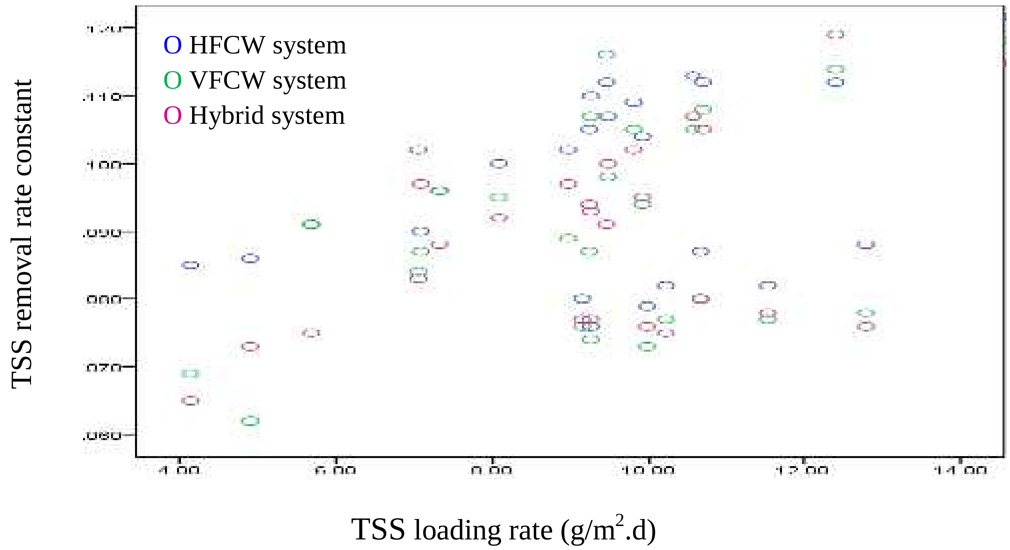

Figure 4.31: TSS loading rate (g/m^2.d) versus areal removal rate constant (m/d) of the pilot scale constructed wetland systems.

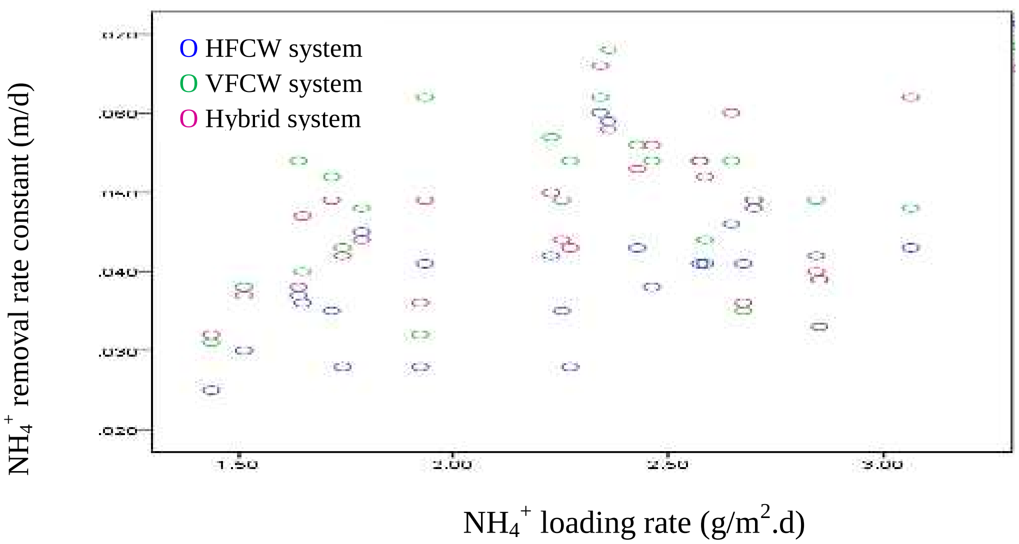

Figure 4.32: NH_4^+ loading rate (g/m^2.d) versus areal removal rate constant (m/d) of the pilot scale constructed wetland systems.

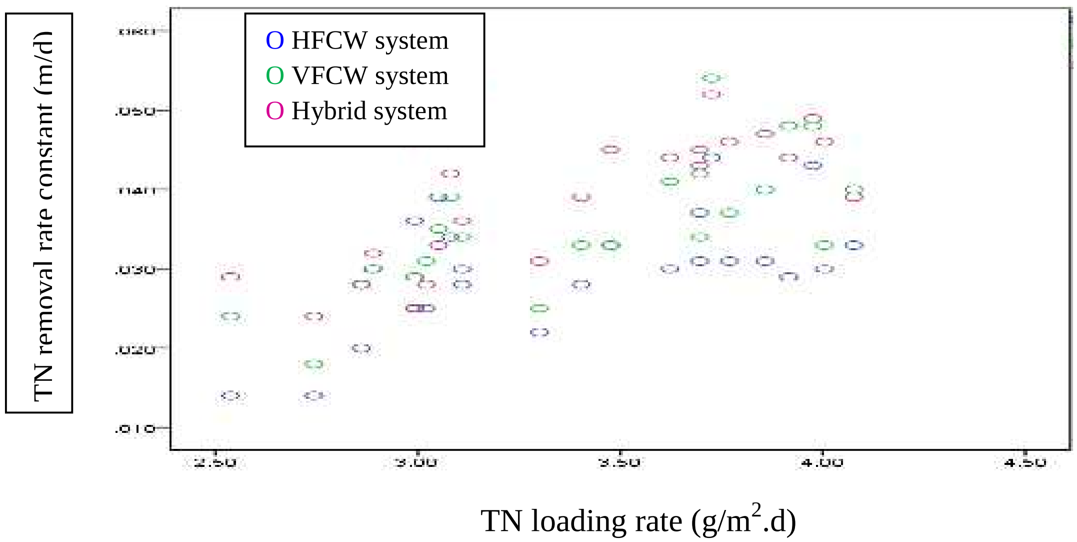

Figure 4.33: TN loading rate (g/m^2.d) versus areal removal rate constant (m/d) of the pilot scale constructed wetland systems.

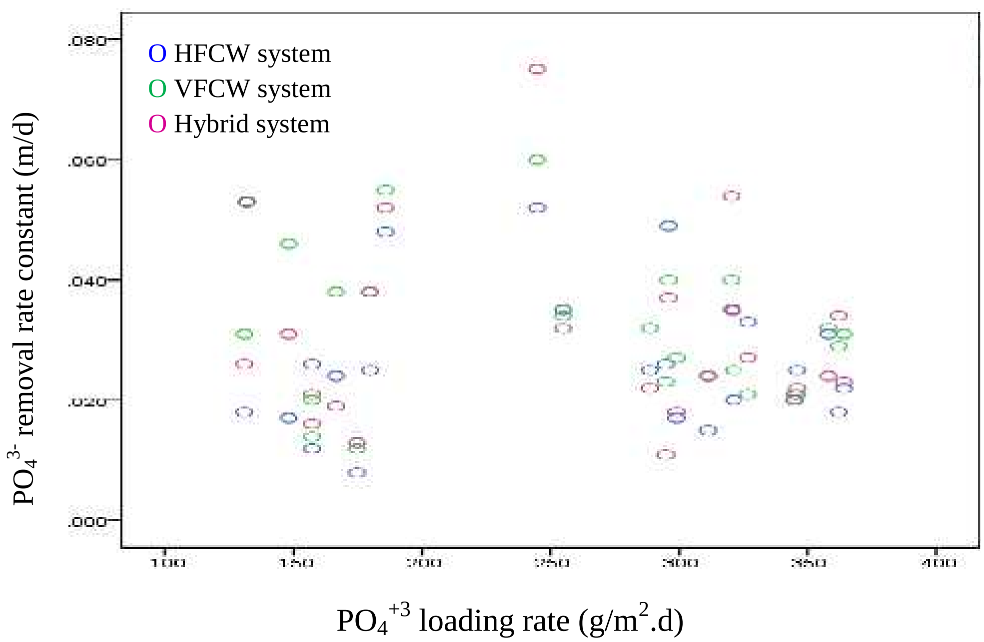

Figure 4.34: PO_4^{3-} loading rate (g/m^2.d) versus areal removal rate constant (m/d) of the pilot scale constructed wetland systems.

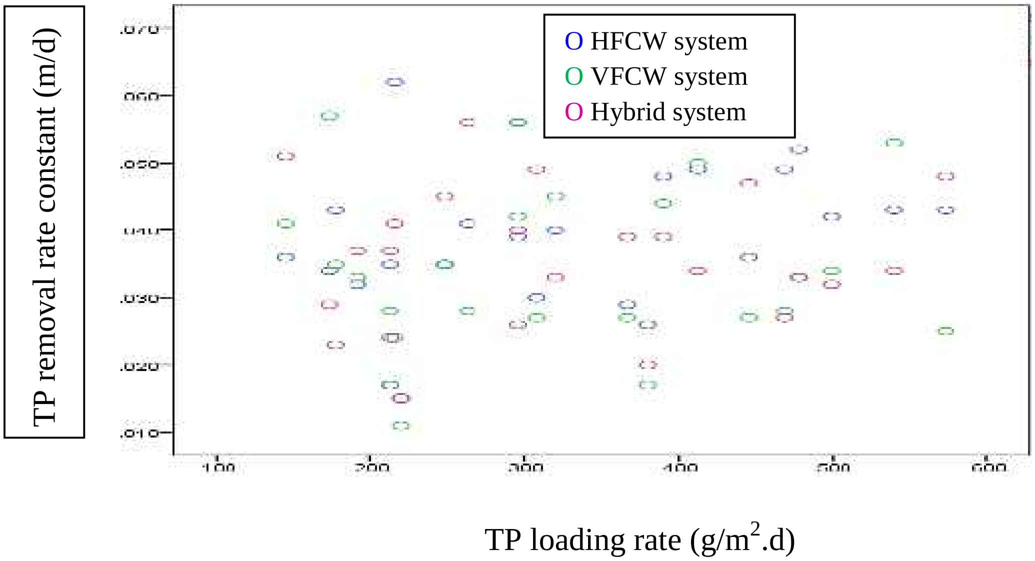

Figure 4.35: TP loading rate (g/m^2.d) versus areal removal rate constant (m/d) of the pilot scale constructed wetland systems.

4.13 Plant tissue nutrient (N and P) content

The nutrient contents of the above-ground and below-ground were determined and expressed in terms of TN and TP percentages in dry weight of *C. papyrus* biomass (Figure 4.29 and 4.30 and Table 4-23). The Below-ground and above-ground N contents of the dry weight *C. papyrus* were 1.56 ± 0.26% and 2.27 ± 0.57% for horizontal bed, 1.75 ± 0.44 and 2.74 ± 0.52% for vertical bed, and 1.80 ± 0.45% and 2.63 ±0.53% for hybrid bed, respectively.

Based on the result, the above-ground parts of *C. papyrus* had higher N content among the three CWs system.This was in agreement with the results of several studies conducted by many authors who reported that the above-ground parts of wetland plants are, in general efficient in N uptake (Korkusuz *et al*, 2004, Adhikari *et al*, 2011, Kassa and Mengistu, 2014; Chen *et al*, 2015). This could be due to the fact that nitrogen is essential element for photosynthetic plants to carryout photosynthesis.

The N contents of the below-ground and above-ground parts of *C. papyrus* in the horizontal, vertical and hybrid type CWs differed significantly from each other statistically and the one-way ANOVA results were $F_{0.95}$ (1, 10) = 7.866; $P < 0.05$, $F_{0.95}$ (1, 10) = 12.717; $P < 0.05$ and $F_{0.95}$ (1, 10) = 8.694; $P < 0.05$, respectively.

But in the case of phosphorous, below-ground parts of *C. papyrus* were found to have higher percentage than above-ground parts. The P percentage in below-ground and above-ground *C. papyrus* parts were 0.139 ±0.43% and 0.064 ± 0.033% in horizontal flow, 0.167 ± 0.063% and 0.067 ±0.029% in vertical flow, and 0.115 ± 0.026% and 0.065 ± 0.031% in hybrid flow type, respectively. The higher P content in belowground *C. papyrus* might be related to the fact that P is used by plants to develop strong root system .

The percentage of P in the below-ground and above-ground parts of *C. papyrus* in the horizontal, vertical and hybrid type CWs differed significantly from each other statistically and the one-way ANOVA results were $F_{0.95}$ (1, 10) = 11.563; $P < 0.05$, $F_{0.95}$ (1, 10) = 12.482; $P < 0.05$ and $F_{0.95}$ (1, 10) = 9.082; $P < 0.05$, respectively.

In general, it was observed in this study that the concentration of nitrogen in the wetland plant was higher than phosphorous concentration and similar result was reported in literatures (Kyambadde *et al*, 2005; Kassa and Mengistu, 2014; Costa *et al*, 2015). This was inferred to the

Redfield ratio which states that a measure of relative ecosystem requirements is the proportion among the nutrient elements in the biomass, which is often represented as a molar proportion of C: N: P =106:16:1 (Kadlec and Wallace, 2009).

The availability of nitrogen and phosphorous for the wetland plant roots and rhizomes in the filter media might improve the nutrient uptake of plants which inturn result in high concentration of N and P in the plant tissue (Vymazal *et al*, 1998). In this study, the nitrogen contents of macrophytes grown on the wetland beds of the vertical flow and hybrid flow CWs were relatively higher than those macrophytes grown on the wetland bed of the horizontal flow CW. This could be due to the production of higher nitrate within the two systems as a result of better oxygen supply compared to the horizontal flow system.

Zhang *et al* (2009) reported that the removal efficiency of planted CWs is higher than unplanted CWs for certain pollutants such as TN and NH_4-N. Moreover, the performance of planted wetlands in removing nitrogen is usually found to be efficient and steady in all months of the year (Lee and Scholz, 2007; Fonkou *et al*, 2011; Abou-Elela and Hellal, 2012; Mesquita *et al*, 2012).

In the process of assimilation, the plants reduce inorganic N to organic N compounds, plant structure. There is significantly high rate of N uptake by wetland plants from water and sediments during the growing season. Increased immobilization of nutrients by microbes and uptake by algae and epiphytes also lead to retention of inorganic N. The net annual uptake of N by macrophytes approximately ranges between 0.5 to 3.3 g N/m^2/yr (US EPA, 2000).

The uptake and release of P occur similarly as that of the microbes, but the reactions require long period of time, possibly months to years. Uptake occurs during the growth phase of the plant and release occurs during plant senescence and death, followed by decomposition (US EPA, 2000). Adhikari *et al* (2011) states that P concentration was higher in the belowground parts of wetland plants, which suggests that regular harvest of the root system would be necessary for achieving maximum P removal. Assimilation of P in vegetation is usually short-term and decomposition of detrital plant tissue is usually rapid resulting in release of P. However, the undecomposed organic P accumulates in the system and becomes an integral part of the soil/sediment P pool (Reddy *et al*, 1995).

Generally, the nutrient removal capacity of a wetland system was more dependent on individual plant biomass irrespective of plant type, i.e., on the size of individual plants or plant density. Hence, harvest of the root system might be needed for optimizing the removal of phosphorous and as well regualr removal of the above-ground plant part would be helpful for the removal of nitrogen (Adhikari *et al*, 2011).

Table 4-23: Average values of plant tissue nutrient content (%) of the pilot scale constructed wetland systems.

Nutrient	***C. papyrus* in horizontal cell**		***C. papyrus* in vertical cell**		***C. papyrus* in hybrid cell**	
	Below-Ground	Above-Ground	Below-ground	Above-ground	Below-ground	Above-ground
% of TN (May)	1.59 ± 0.17	2.18 ± 0.81	2.08 ± 0.19	2.39 ± 0.35	1.97 ± 0.619	2.61 ± 0.64
% of TN (November)	1.52 ± 0.37	2.36 ± 0.40	1.43 ± 0.35	3.08 ± 0.44	1.62 ± 0.14	2.64 ± 0.53
TN average	1.56 ± 0.26	2.27 ± 0.57	1.75 ± 0.44	2.74 ± 0.52	1.80 ± 0.45	2.63 ± 0.53
% of TP (May)	0.139 ± 0.05	0.057 ± 0.02	0.211 ± 0.04	0.081 ± 0.03	0.118 ± 0.02	0.058 ± 0.01
% of TP (November)	0.138 ± 0.05	0.071 ± 0.05	0.123 ± 0.05	0.054 ± 0.03	0.111 ± 0.04	0.071 ± 0.05
% TP average	0.139 ± 0.43	0.064 ± 0.03	0.167 ± 0.06	0.067 ± 0.03	0.115 ± 0.03	0.065 ± 0.03

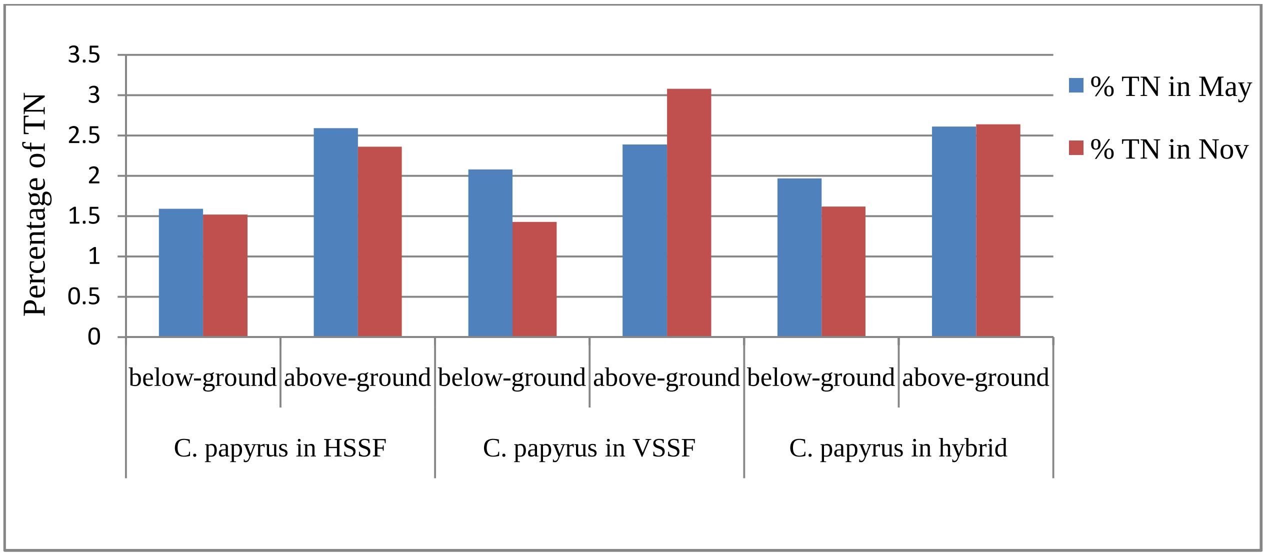

Figure 4.36: Percentage of TN in plant tissue (*C. papyrus*) of the pilot scale constructed wetland systems.

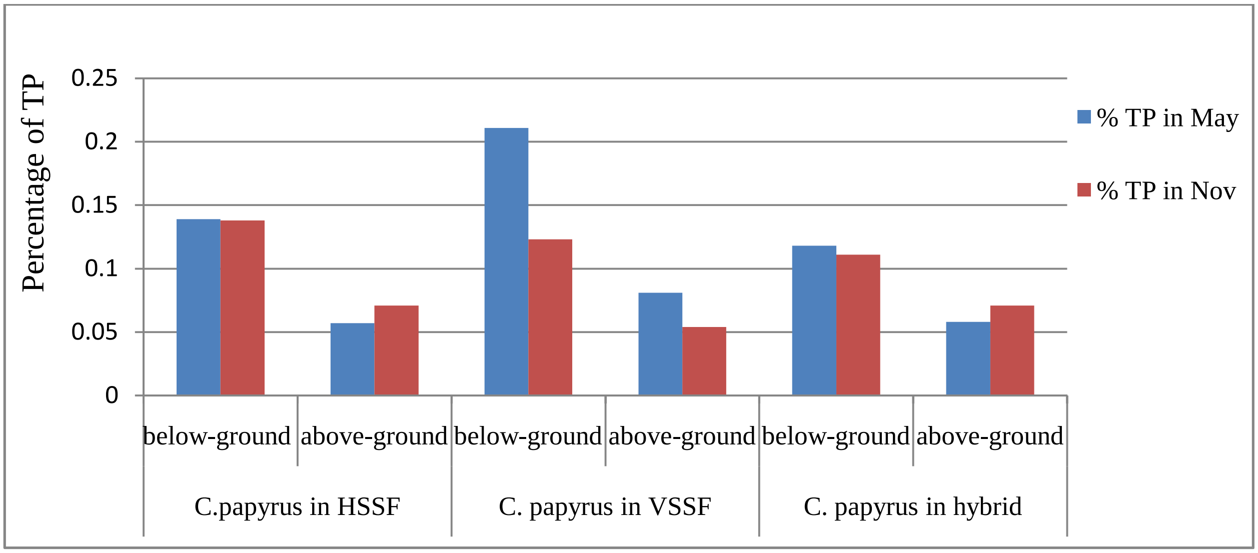

Figure 4.37: Percentage of TP in plant tissue (*C. papyrus*) of the pilot scale constructed wetland systems.

CHAPTER FIVE

CONCLUSION AND RECOMMENDATIONS

5.1 Conclusion

The application of effective, low-cost, less-energy intensive, and easily operated secondary wastewater treatment methods is an issue of great concern in Ethiopia as discharging of untreated wastewater into water bodies is the common practice in most parts of the country. It is necessary to protect the environment and public health, and beyond this, it can create further opportunity to re-use the treated wastewater for non-domestic purposes such as irrigation, construction works and ground water recharging.

The findings of this study give a snapshot of the performance of CW systems in tropical areas like Ethiopia. It is expected to contribute to the understanding of how the horizontal, vertical and hybrid subsurface flow constructed wetland systems with similar wetland plants and fill media worked in the prevailing climatic conditions in Addis Ababa. It also helps to weigh up the impact of different seasons and loading rates in the removal of pollutants in domestic wastewater. Furthermore, the result offers a clue concerning the removal rate constants of wastewater pollutants and nutrient uptake of wetland plants in the tropics.

Based on the results presented, the HFCW, VFCW and hybrid of the two systems showed good BOD, COD, TSS, NH_4^+, NO_3^-, TN, TP and FC removal efficiencies and the effluent concentrations of all wastewater pollutants except NH_4^+ and FCs comply with the discharge standard set by Ethiopian Environmental Protection Authority (EEPA, 2003). However, the annual average NH_4^+ and FC concentrations of the effluents were higher than the provisional discharge standards. The annual average percent reduction of PO_4^{3-} was low during the monitoring period although the effluent concentrations comply with the provisonal discharge standard.

Comparing the performance of the three pilot scale constructed wetland systems, the removal efficiencies of the hybrid and VFCW were higher than the HFCW system for most parameters except TSS and TP. In line with this, the results of one-way ANOVA showed that the performances of the HFCW, VFCW and the hybrid wetland systems were different from one another statistically for BOD_5, TSS, NH_4^+, TN and FC; while the efficiencies of the three systems did not differ statistically one another for the parameters: COD, NO_3^-, PO_4^{3-}, and TN. In the interim, the correlation between removal rate and loading rate were stronger for BOD_5, COD, TSS, TN and TP whereas there was relatively weak correlation for NH_4^+-N, NO_3^- -N and PO_4^{3-} -P. In general terms, the result showed that the removal rates of wastewater pollutants were increased as the loading rate increased although the association varied from parameter to parameter in this study.

On the other hand, the effect of seasons was demonstrated in the removal efficiencies of all the three CW cells and higher percent reductions were observed during the dry/hot seasons (December - May). One-way ANOVA to compare the performances of the CWs in different seasons revealed that the removal efficiencies of all the three CW systems differed significantly from season to season for BOD_5, COD, and NH_4^+. But the percent reductions of other pollutants did not differ statistically from season to season.

With regard to areal removal rate constants, the values obtained in this study for BOD_5, TN, PO_4^{3-} and TP were higher than the literature values while the value for COD was lower than the literature value. The areal removal rate constants for TSS and NH_4^+-N were similar with the values reported in literatures. Among the three pilot scale subsurface flow CW cells, the hybrid system showed relatively higher removal rate constants than the HFCW and VFCW cells for all wastewater parameters.

Concerning the nutrient content of the wetland plants /*C. papyrus*, it was observed that the TN content of the above-ground part of the wetland plant was higher than the TN content of below-ground plant part. On the contrary, the TP content of the below-ground plant part was higher than the TP content of the above-ground part. The result of one way ANOVA revealed that both the TN and TP content of the below-ground and above-ground plant parts in the HFCW and VFCW as well as the hybrid systems differed significantly.

In general, it can be concluded that the treatment performance of the pilot-scale CW systems applied at Kotebe WWTP was very promising for the promotion and application of CWs as an alternative wastewater treatment system to protect the environment and public health. Ethiopia has favorable climatic conditions for the implementation of CW systems and hence, the technology can continue as competent solution to alleviate the inherent environmental problems associated with discharging of untreated wastewaters. It is possible to use CW systems for wider application in different towns for the treatment of municipal wastewater or in small communities/institutions such as universities, colleges, military camps, farms, factories and hospitals.

5.2 Recommendations

Based on the results obtained from this study, the following recommendations are forwarded for further studies and wider application of constructed wetlands as alternative wastewater treatment methods.

1. The removal efficiency of the constructed wetland systems for the removal of most wastewater pollutants considered in this study was good and it is recommended to use the method particularly in small towns and various institutions. However, the selection of wetland plants and fill media should be based on adequate knowledge and experience to optimize the efficiency of constructed wetlands in removing pollutants particularly nitrogen and phosphorous.
2. Integrated constructed wetland systems which comprise of two or more cells connected in series should be employed to improve the quality of effluents.
3. The performance of the pilot scale constructed wetland systems under similar environmental conditions should be evaluated before their implementation at full scale level and also the pilot scale CW systems should be monitored for at least one year to properly address the effect of different seasons.
4. The characteristics of raw wastewaters in wet and dry seasons should be determined and taken in to account during designing and operation of constructed wetland systems.
5. More detailed research works related to investigating the profile of microorganisms and identifying more effective macrophytes and wetland media to optimize nutrient removal should be conducted.

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Annexes:

Table 1: Mean, standard error, minimum and maximum values of BOD_5 (mg/L) within the influent and effluents of the HFCW, VFCW and hybrid of the HFCW and VFCW systems.

Descriptive Statistics

	N	Minimum	Maximum	Mean	Std. Deviation
Dec-Feb (Influent)	6	162.33	224.67	198.3333	21.08179
Mar-May (Influent)	6	146.67	217.67	194.3900	25.55951
Jun-Aug (Influent)	6	132.67	205.67	163.4450	25.09931
Sep-Nov (Influent)	6	167.00	204.33	189.1650	14.95608
Dec-Feb (HFCW-effluent)	6	10.00	27.67	15.5567	6.53263
Mar-May (HFCW-effluent)	6	6.33	16.33	10.8867	3.94807
Jun-Aug (HFCW-effluent)	6	13.33	32.33	23.1650	8.36807
Sep-Nov (HFCW-effluent)	6	18.00	45.67	31.4450	10.52766
Dec-Feb (VFCW-effluent)	6	5.00	15.00	9.5000	3.90788
Mar-May (VFCW-effluent)	6	4.00	11.67	7.3350	2.55708
Jun-Aug (VFCW-effluent)	6	9.00	27.67	17.5550	7.62098
Sep-Nov (VFCW-effluent)	6	11.33	32.33	22.3300	8.14862
Dec-Feb (Hyb CW-effluent)	6	6.00	10.00	8.1117	1.96326
Mar-May (Hyb CW-effluent)	6	4.00	9.33	5.7767	2.16593
Jun-Aug (Hyb CW-effluent)	6	8.67	23.00	15.7800	5.72827
Sep-Nov (Hyb CW-effluent)	6	10.67	25.33	18.5000	6.55096
Valid N (listwise)	6				

Table 2: Mean, standard error, minimum and maximum values of COD (mg/L) within the influent and effluents of the HFCW, VFCW and hybrid of the HFCW and VFCW systems.

Descriptive Statistics

	N	Minimum	Maximum	Mean	Std. Deviation
Dec-Feb (Influent)	6	344.00	505.33	447.9983	65.06278
Mar-May (Influent)	6	357.00	465.00	399.0567	39.27258
Jun-Aug (Influent)	6	323.00	445.00	374.0550	44.07695
Sep-Nov (Influent)	6	313.33	445.33	390.1083	51.96378
Dec-Feb (HFCW-effluent)	6	38.00	78.67	60.9450	14.28724
Mar-May (HFCW-effluent)	6	46.00	83.00	61.6117	14.40239
Jun-Aug (HFCW-effluent)	6	67.67	103.00	84.2217	13.53198
Sep-Nov (HFCW-effluent)	6	72.33	125.00	102.0550	18.21191
Dec-Feb (VFCW-effluent)	6	41.00	83.33	63.9433	15.82704
Mar-May (VFCW-effluent)	6	36.67	64.33	51.3883	10.76329
Jun-Aug (VFCW-effluent)	6	65.00	93.67	78.6683	11.53208
Sep-Nov (VFCW-effluent)	6	56.67	123.33	91.2767	26.35577
Dec-Feb (Hyb CW-effluent)	6	38.67	72.00	53.7767	12.64386
Mar-May (Hyb CW-effluent)	6	33.00	61.00	47.7783	10.14388
Jun-Aug (Hyb CW-effluent)	6	56.00	85.33	68.9450	12.71809
Sep-Nov (Hyb CW-effluent)	6	56.00	116.00	84.3333	21.20147
Valid N (listwise)	6				

Table 3: Mean, standard error, minimum and maximum values of TSS (mg/L) within the influent and effluents of the HFCW, VFCW and hybrid of the HFCW and VFCW systems.

Descriptive Statistics

	N	Minimum	Maximum	Mean	Std. Deviation
Dec-Feb (Influent)	6	129.00	240.33	186.9433	41.52406
Mar-May (Influent)	6	94.00	290.67	189.1117	73.96847
Jun-Aug (Influent)	6	207.67	262.33	230.2783	20.52277
Sep-Nov (Influent)	6	160.33	282.00	217.8883	43.06007
Dec-Feb (HFCW-effluent)	6	10.67	21.00	15.8350	3.76360
Mar-May (HFCW-effluent)	6	12.00	32.33	18.5567	7.40859
Jun-Aug (HFCW-effluent)	6	33.67	41.00	36.5567	2.75305
Sep-Nov (HFCW-effluent)	6	13.67	25.67	19.9467	4.05212
Dec-Feb (VFCW-effluent)	6	18.33	39.00	28.4433	7.89983
Mar-May (VFCW-effluent)	6	19.67	49.00	30.8883	9.77642
Jun-Aug (VFCW-effluent)	6	36.67	51.67	46.1667	5.35218
Sep-Nov (VFCW-effluent)	6	20.33	40.00	28.7767	7.51902
Dec-Feb (Hyb CW-effluent)	6	16.67	30.67	23.4450	5.92649
Mar-May (Hyb CW-effluent)	6	16.67	44.67	25.3917	10.10311
Jun-Aug (Hyb CW-effluent)	6	36.00	44.67	39.7783	3.27756
Sep-Nov (Hyb CW-effluent)	6	21.67	44.33	34.1667	8.29155
Valid N (listwise)	6				

Table 4: Mean, standard error, minimum and maximum values of NH_4^+ (mg/L) within the influent and effluents of the HFCW, VFCW and hybrid of the HFCW and VFCW systems.

Descriptive Statistics

	N	Minimum	Maximum	Mean	Std. Deviation
Dec-Feb (Influent)	6	53.17	69.58	58.4950	6.06465
Mar-May (Influent)	6	43.92	64.68	55.7450	7.57025
Jun-Aug (Influent)	6	37.30	43.67	39.6133	2.34730
Sep-Nov (Influent)	6	32.63	64.83	49.2150	13.28853
Dec-Feb (HFCW-effluent)	6	13.73	26.33	20.2817	5.21357
Mar-May (HFCW-effluent)	6	17.10	24.70	20.9933	2.71070
Jun-Aug (HFCW-effluent)	6	14.67	22.97	18.1117	3.16745
Sep-Nov (HFCW-effluent)	6	17.42	30.53	23.4217	4.98532
Dec-Feb (VFCW-effluent)	6	11.40	23.50	16.5650	4.20873
Mar-May (VFCW-effluent)	6	10.75	21.53	17.1750	4.50627
Jun-Aug (VFCW-effluent)	6	10.90	21.10	14.6117	3.55482
Sep-Nov (VFCW-effluent)	6	14.63	27.23	19.4133	5.81890
Dec-Feb (Hyb CW-effluent)	6	11.93	17.03	15.1933	1.92236
Mar-May (Hyb CW-effluent)	6	14.43	24.90	18.3317	3.73288
Jun-Aug (Hyb CW-effluent)	6	12.83	19.27	15.1117	2.37746
Sep-Nov (Hyb CW-effluent)	6	14.87	26.70	20.3500	5.14674
Valid N (listwise)	6				

Table 5: Mean, standard error, minimum and maximum values of NO_3^- (mg/L) within the influent and effluents of the HFCW, VFCW and hybrid of the HFCW and VFCW systems.

Descriptive Statistics

	N	Minimum	Maximum	Mean	Std. Deviation
Dec-Feb (Influent)	6	5.90	14.50	10.0500	2.87836
Mar-May (Influent)	6	3.47	9.30	6.0900	2.19695
Jun-Aug (Influent)	6	4.47	11.60	8.0783	2.90027
Sep-Nov (Influent)	6	3.33	9.63	5.8267	2.59512
Dec-Feb (HFCW-effluent)	6	1.80	7.57	4.9017	2.38606
Mar-May (HFCW-effluent)	6	1.17	3.07	1.8283	.70822
Jun-Aug (HFCW-effluent)	6	.97	3.10	2.6517	.83219
Sep-Nov (HFCW-effluent)	6	.80	3.53	1.8833	1.12552
Dec-Feb (VFCW-effluent)	6	2.60	4.70	3.5700	.92566
Mar-May (VFCW-effluent)	6	.73	2.43	1.4267	.61079
Jun-Aug (VFCW-effluent)	6	.83	3.37	1.9233	.83930
Sep-Nov (VFCW-effluent)	6	.63	3.57	1.8167	1.01348
Dec-Feb (Hyb CW-effluent)	6	.93	6.07	3.2117	1.98321
Mar-May (Hyb CW-effluent)	6	1.07	2.60	1.5550	.56362
Jun-Aug (Hyb CW-effluent)	6	.83	2.63	1.6933	.72627
Sep-Nov (Hyb CW-effluent)	6	1.07	2.63	1.6067	.57712
Valid N (listwise)	6				

Table 6: Mean, standard error, minimum and maximum values of TN (mg/L) within the influent and effluents of the HFCW, VFCW and hybrid of the HFCW and VFCW systems.

Descriptive Statistics

	N	Minimum	Maximum	Mean	Std. Deviation
Dec-Feb (Influent)	6	82.33	92.67	87.2233	4.02185
Mar-May (Influent)	6	57.67	90.33	80.7783	12.01259
Jun-Aug (Influent)	6	62.33	70.67	67.3333	2.96003
Sep-Nov (Influent)	6	65.00	77.33	71.1117	4.43919
Dec-Feb (HFCW-effluent)	6	41.00	46.00	43.1667	1.62904
Mar-May (HFCW-effluent)	6	31.33	45.67	37.6117	5.27408
Jun-Aug (HFCW-effluent)	6	28.67	45.33	35.1667	6.15381
Sep-Nov (HFCW-effluent)	6	32.33	45.33	39.2767	4.32843
Dec-Feb (VFCW-effluent)	6	32.00	42.67	36.2233	3.71150
Mar-May (VFCW-effluent)	6	25.00	38.67	32.4450	5.03743
Jun-Aug (VFCW-effluent)	6	24.33	41.00	33.3867	5.51214
Sep-Nov (VFCW-effluent)	6	28.67	42.33	34.8883	4.47325
Dec-Feb (Hyb CW-effluent)	6	30.00	38.33	32.1650	3.12421
Mar-May (Hyb CW-effluent)	6	26.00	32.67	29.5000	2.21971
Jun-Aug (Hyb CW-effluent)	6	21.33	38.67	30.7783	6.78958
Sep-Nov (Hyb CW-effluent)	6	27.00	36.67	32.9433	3.63583
Valid N (listwise)	6				

Table 7: Mean, standard error, minimum and maximum values of PO_4^{3-} (mg/L) within the influent and effluents of the HFCW, VFCW and hybrid of the HFCW and VFCW systems.

Descriptive Statistics

	N	Minimum	Maximum	Mean	Std. Deviation
Dec-Feb (Influent)	6	5.80	8.23	6.9950	.89717
Mar-May (Influent)	6	7.07	8.27	7.6717	.47993
Jun-Aug (Influent)	6	2.97	7.27	4.0917	1.58178
Sep-Nov (Influent)	6	3.00	6.73	4.6000	1.32828
Dec-Feb (HFCW-effluent)	6	2.60	5.37	4.0567	.94046
Mar-May (HFCW-effluent)	6	3.53	5.13	4.5533	.64537
Jun-Aug (HFCW-effluent)	6	1.97	3.27	2.3950	.48344
Sep-Nov (HFCW-effluent)	6	.90	3.30	1.9617	.83712
Dec-Feb (VFCW-effluent)	6	2.70	4.90	3.7717	.78306
Mar-May (VFCW-effluent)	6	3.93	4.87	4.2933	.36909
Jun-Aug (VFCW-effluent)	6	1.20	2.97	2.0067	.67846
Sep-Nov (VFCW-effluent)	6	.90	3.07	1.8433	.86929
Dec-Feb (Hyb CW-effluent)	6	2.80	5.27	4.1583	.85861
Mar-May (Hyb CW-effluent)	6	3.33	4.97	4.3333	.64357
Jun-Aug (Hyb CW-effluent)	6	1.67	2.53	2.1000	.36540
Sep-Nov (Hyb CW-effluent)	6	.90	3.00	1.8050	.92115
Valid N (listwise)	6				

Table 8: Mean, standard error, minimum and maximum values of TP (mg/L) within the influent and effluents of the HFCW, VFCW and hybrid of the HFCW and VFCW systems.

Descriptive Statistics

	N	Minimum	Maximum	Mean	Std. Deviation
Dec-Feb (Influent)	6	4.37	11.33	8.2833	2.70450
Mar-May (Influent)	6	7.27	13.03	10.4067	2.22662
Jun-Aug (Influent)	6	3.30	6.73	4.7700	1.14666
Sep-Nov (Influent)	6	3.93	8.63	6.2100	1.64140
Dec-Feb (HFCW-effluent)	6	2.10	4.47	3.2617	.95120
Mar-May (HFCW-effluent)	6	2.93	4.87	3.9167	.76521
Jun-Aug (HFCW-effluent)	6	1.20	3.27	2.0683	.81835
Sep-Nov (HFCW-effluent)	6	1.80	4.73	2.9700	1.15310
Dec-Feb (VFCW-effluent)	6	2.07	5.43	3.5933	1.40493
Mar-May (VFCW-effluent)	6	2.63	7.33	4.8267	1.62267
Jun-Aug (VFCW-effluent)	6	1.30	3.27	2.3900	.72014
Sep-Nov (VFCW-effluent)	6	1.07	5.87	3.2783	1.68343
Dec-Feb (Hyb CW-effluent)	6	1.87	5.50	3.4850	1.38587
Mar-May (Hyb CW-effluent)	6	3.40	5.73	4.6267	1.04951
Jun-Aug (Hyb CW-effluent)	6	1.03	2.83	2.1600	.65663
Sep-Nov (Hyb CW-effluent)	6	1.67	5.47	3.1283	1.41843
Valid N (listwise)	6				

Table 9: Mean, standard error, minimum and maximum values of FC (CFU/100ml) within the influent and effluents of the HFCW, VFCW and hybrid of the HFCW and VFCW systems.

Descriptive Statistics

	N	Minimum	Maximum	Mean	Std. Deviation
Dec-Feb (Influent)	6	79000.00	148333.33	101000.0000	26393.59959
Mar-May (Influent)	6	72000.00	127666.67	110777.7783	21088.87845
Jun-Aug (Influent)	6	73666.67	88000.00	81166.6683	4838.50172
Sep-Nov (Influent)	6	74666.67	97666.67	88222.2250	9708.33869
Dec-Feb (HFCW-effluent)	6	1466.67	5800.00	2410.0000	1671.61237
Mar-May (HFCW-effluent)	6	1266.67	2633.33	2166.6667	493.96248
Jun-Aug (HFCW-effluent)	6	966.67	1700.00	1316.6667	259.70025
Sep-Nov (HFCW-effluent)	6	1666.67	2766.67	2144.4467	481.51106
Sep-Nov (VFCW-effluent)	6	2166.67	3700.00	2872.2233	581.34470
Dec-Feb (VFCW-effluent)	6	2766.67	6333.33	3811.1117	1303.95258
Mar-May (VFCW-effluent)	6	2533.33	5233.33	4183.3333	1091.12556
Jun-Aug (VFCW-effluent)	6	1366.67	3000.00	2133.3333	583.85698
Sep-Nov (VFCW-effluent)	6	2166.67	3700.00	2872.2233	581.34470
Dec-Feb (Hyb CW-effluent)	6	2000.00	7433.33	3216.6667	2087.39469
Mar-May (Hyb CW-effluent)	6	3700.00	5400.00	4544.4450	643.13957
Jun-Aug (Hyb CW-effluent)	6	1966.67	3233.33	2611.1117	480.58498
Sep-Nov (Hyb CW-effluent)	6	2366.67	5033.33	3038.8900	993.84918
Valid N (listwise)	6				

Table 10: Mean, standard error, minimum and maximum values of DO (mg/L) within the influent and effluents of the HFCW, VFCW and hybrid of the HFCW and VFCW systems.

Descriptive Statistics

	N	Minimum	Maximum	Mean	Std. Deviation
Dec-Feb (Influent)	6	.24	.45	.3750	.08758
Mar-May (influent)	6	.35	1.42	.8667	.50698
Jun-Aug (Influent)	6	.98	1.38	1.2033	.15148
Sep-Nov (Influent)	6	.40	.92	.5567	.20695
Dec-Feb (HFCW - Effluent)	6	.35	1.16	.8300	.31962
Mar-May (HFCW - Effluent)	6	.94	2.52	1.7333	.55164
Jun-Aug (HFCW - Effluent)	6	2.26	3.01	2.7817	.28666
Sep-Nov (HFCW - Effluent)	6	.85	1.62	1.4200	.29237
Dec-Feb (VFCW - Effluent)	6	.56	1.77	1.1150	.46976
Mar-May (VFCW - Effluent)	6	1.24	3.02	2.1017	.61506
Jun-Aug (VFCW - Effluent)	6	2.50	4.11	3.3267	.52095
Sep-Nov (VFCW - Effluent)	6	1.16	2.52	1.8817	.47893
Dec-Feb (Hyb CW - Effluent)	6	.65	1.37	1.0167	.26136
Mar-May (Hyb CW - Effluent)	6	1.37	2.90	2.2433	.62289
Jun-Aug (Hyb CW - Effluent)	6	2.66	3.50	3.2100	.30816
Sep-Nov (Hyb CW - Effluent)	6	1.23	2.28	1.6567	.37739
Valid N (listwise)	6				

Table 11: Mean, standard error, minimum and maximum values of water Temperature (^{o}C) of the influent and effluents of the HFCW, VFCW and hybrid of the HFCW and VFCW systems.

Descriptive Statistics

	N	Minimum	Maximum	Mean	Std. Deviation
Dec-Feb (Influent)	6	20.27	23.13	21.5617	1.07216
Mar-May (Influent)	6	21.13	23.10	22.4000	.78991
Jun-Aug (Influent)	6	16.60	19.67	18.3283	1.39240
Sep-Nov (Influent)	6	14.13	17.57	16.0517	1.39158
Dec-Feb (HFCW - Effluent)	6	19.93	22.03	20.9717	.72494
Mar-May (HFCW - Effluent)	6	20.00	21.40	20.8217	.51223
Jun-Aug (HFCW - Effluent)	6	14.57	18.93	16.6283	2.01454
Sep-Nov (HFCW - Effluent)	6	11.30	16.10	13.5183	1.89751
Dec-Feb (VFCW - Effluent)	6	19.33	20.80	20.0950	.61426
Mar-May (VFCW - Effluent)	6	20.83	21.33	21.0033	.17282
Jun-Aug (VFCW - Effluent)	6	14.13	18.13	15.6717	1.72509
Sep-Nov (VFCW - Effluent)	6	11.93	15.83	13.9767	1.75192
Dec-Feb (Hyb CW - Effluent)	6	19.77	20.97	20.4117	.43306
Mar-May (Hyb CW - Effluent)	6	19.63	21.97	21.1283	.80123
Jun-Aug (Hyb CW - Effluent)	6	14.80	18.93	16.3267	1.49140
Sep-Nov (Hyb CW - Effluent)	6	12.03	15.37	13.1383	1.41546
Valid N (listwise)	6				

Table 12: Areal removal rate constants, K (m/d) of the parameters considered in this study.

BOD			
Seasons	K in HSSF type	K in VF type	K in Hybrid type
Dec - Feb	0.112	0.134	0.141
Mar - May	0.127	0.144	0.155
Jun - Aug	0.086	0.098	0.103
Sep - Nov	0.079	0.094	0.102
Annual	0.098	0.113	0.121
COD			
Seasons	K in HSSF	K in VF	K in Hybrid
Dec - Feb	0.088	0.086	0.093
Mar - May	0.082	0.090	0.093
Jun - Aug	0.066	0.069	0.075
Sep - Nov	0.059	0.064	0.067
Annual	0.073	0.076	0.081
TSS			
Seasons	K in HSSF	K in VF	K in Hybrid
Dec - Feb	0.109	0.083	0.091
Mar - May	0.102	0.080	0.088
Jun - Aug	0.081	0.071	0.077
Sep - Nov	0.105	0.089	0.082
Annual	0.097	0.080	0.084
NH_4^+-N			
Seasons	K in HSSF	K in VF	K in Hybrid
Dec - Feb	0.047	0.055	0.059
Mar - May	0.043	0.052	0.049
Jun - Aug	0.034	0.044	0.042
Sep - Nov	0.033	0.041	0.039
Annual	0.040	0.048	0.048
TN			
Seasons	K in HSSF	K in VF	K in Hybrid
Dec - Feb	0.031	0.039	0.044
Mar - May	0.034	0.040	0.044
Jun - Aug	0.029	0.031	0.034
Sep - Nov	0.026	0.031	0.034
Annual	0.030	0.036	0.039
PO_4^{3-}- P			
Seasons	K in HSSF	K in VF	K in Hybrid
Dec - Feb	0.024	0.027	0.023
Mar - May	0.023	0.026	0.025
Jun - Aug	0.023	0.031	0.029
Sep - Nov	0.038	0.040	0.041
Annual	0.026	0.030	0.028

TP			
Seasons	K in HSSF	K in VF	K in Hybrid
Dec - Feb	0.042	0.038	0.039
Mar - May	0.033	0.024	0.026
Jun - Aug	0.037	0.030	0.035
Sep - Nov	0.032	0.028	0.030
Annual	0.039	0.033	0.035

Figure 1: Site clearing, excavation work and preparation for the construction of the pilot scale constructed wetlands at Kotebe WWTP, Addis Ababa-Ethiopia

Figure 2: Construction process of the pilot scale constructed wetland systems employed at Kotebe WWTP, Addis Ababa-Ethiopia.

Fig 3: The process of pipe installation, sealing of the wetland systems bed and application of the fill media at Kotebe WWTP, Addis Ababa-Ethiopia.

Fig 4: Source, transportation and plantation of the wetland plant/*C. papyrus*/ to the pilot scale constructed wetlands system applied at Kotebe WWTP, Addis Ababa-Ethiopia.

Figure 6: Adaptation and growth of the wetland plants of the pilot scale constructed wetland systems employed at Kotebe WWTP, Addis Ababa-Ethiopia.