Treatment of Domestic Wastewater by Subsurface Flow Constructed Wetland.

A Thesis
by

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The thesis titled **"TREATMENT OF DOMESTIC WASTEWATER BY SUBSURFACE FLOW CONSTRUCTED WETLAND"**, submitted by **Maharam Dakua**, Roll No - 1009042116, Session - October 2009, has been accepted as satisfactory in partial fulfillment of the requirement for the degree of M. Sc. Engineering (Civil and Environmental) on 23rd of June, 2015.

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It is hereby declared that this thesis or any part of it has not been submitted elsewhere for the award of any degree or diploma.

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Abstract

Constructed wetlands have been used for decades mostly for the treatment of domestic or municipal sewage. For a small community with limited funds for expanding or updating wastewater treatment plants, constructed wetlands are an attractive option. In addition, rural municipalities have access to adequate inexpensive land, and wetlands blend into a natural landscape setting and once the wetlands are designed and constructed, annual maintenance costs are low. Therefore, this study focuses on the design of constructed wetlands that might serve as a treatment plant for domestic wastewater, will be economic and help re-use of wastewater to reduce demand on current available sources.

In urban areas like Dhaka in Bangladesh, water is a growing crisis with the rapidly growing rate of urbanization. It is necessary now to explore ways of using resources in a sustainable way to reduce pressure on centralized systems. Through recycling and reuse of generated wastewater, this crisis could definitely be reduced which also will make the currently available fresh water sufficient for the city. This study works on sub-surface flow constructed wetland systems to test whether it can be a reliable treatment method for recycling grey water which have low organic and nutrient loading. The performance of sub-surface constructed wetlands to treat grey water at household level is tested in this experiment where the removal efficiency of BOD_5, COD, ammonia (total), orthophosphate, TSS, and Fecal Coliform (FC) were tested.

The findings from the study indicate that a basin with surface area less than 5 m^2 is good enough to treat almost 18,000 L of grey water per day. Normally, 60-70% of total wastewater from households is grey water. Assuming that per capita use per day is 225 L, then a constructed wetland with surface area of 5 m^2 would treat grey water of almost 20 families (6 person/family). But it should be noted that the flow rate should not be above 12 L/min. In that case a sedimentation tank should be designed to control the flow rate into the basin.

The results show that removal efficiency of TSS varies in the range from 70-95%. Removal efficiency of BOD is more than 90%. Removal efficiency of BOD is more than 90%. Removal efficiency of total ammonia varies in the range from 70-95%. Removal efficiency of orthophosphate varies in the range from 45-70%. Removal of FC was found high in constructed wetlands. Vertical flow constructed wetlands are more efficient in removing pollutants from water. It was also experienced in the study that removal rate is higher with vegetation, except for suspended solids. Basin 1 also shows higher efficiency than basin 2 that indicates that higher travel time and length increase removal efficiency. The study shows that subsurface constructed wetlands can reduce the concentration of pollutants that are present in domestic grey water, to improve downstream water quality. As the systems produce an effluent low in BOD, which has a much lower total-NH_3 and P, and show good color and less turbidity, the system can be used at community/household level that will help to reduce the water crisis of Dhaka city.

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Acronyms and Abbreviations

BOD - Biochemical Oxygen Demand

COD - Chemical Oxygen Demand

CW - Constructed Wetlands

EPA - (US) Environmental Protection Agency

ET - Evapotranspiration

FC - Fecal Coliform

FWS - Free Water Surface

HF - Horizontal Flow

LPCD - Liter Per Capita Per Day

OTR - Oxygen Transfer Rate

PE - Population Equivalent

RBTS - Reed Bed Treatment System

SF - Subsurface Flow

TDS - Total Dissolved Solids

TKN - Total Kjeldahl Nitrogen

TN - Total Nitrogen

TOC - Total Organic Carbon

TP - Total Phosphate

TSS - Total Suspended Solids

VF - Vertical Flow

Chapter 1: Introduction

1.1 Background

Wetlands are among the most important ecosystems on Earth because of their unique conditions and their role as ecotones between terrestrial and aquatic systems (Mitsch and Gosselink, 1993). The ability of wetlands to transform and store organic matter and nutrients has resulted in wetlands often being described as “the kidneys of the landscape” (Mitsch and Gosselink, 1993). Constructed wetlands also have great potential as a clean-up technology for varieties of wastewater (Lorion, 2001).

Constructed wetlands have been used for decades mostly for the treatment of domestic or municipal sewage. For a small community with limited funds for expanding or updating wastewater treatment plants, constructed wetlands are an attractive option. In addition, rural municipalities have access to adequate inexpensive land, and wetlands blend into a natural landscape setting and once the wetlands are designed and constructed, annual maintenance costs are low. Therefore, this study focuses on the design of constructed wetlands that might serve as a treatment plant for domestic wastewater, will be economic and help re-use of wastewater to reduce demand on current available sources.

In order to assist treating wastewaters, Constructed Wetlands (CWs) are engineered systems that have been designed and constructed to utilize the natural processes involving wetland vegetation, soils, and the associated microbial assemblages. CWs for wastewater treatment may be classified according to the life form of the dominating macrophyte into systems with free-floating, floating leaved, rooted emergent and submerged macrophytes (Brix and Schierup, 1989). Further division could be made according to the wetland hydrology (free water surface and subsurface systems). Again, subsurface flow CWs could be classified according to the direction of the flow; horizontal and vertical (Vymazal and Kröpfelová, 2008). Constructed wetlands with free water surface (FWS CWs) are not used as much as the HF or VF systems despite being one of the oldest designs in Europe (Brix, 1994; Vymazal et al., 1994; Vymazal, 2001). Constructed wetlands could also be combined in order to achieve a higher treatment effect by using advantages of individual systems. Most hybrid constructed wetlands combine VF and HF stages (Vymazal, 2005).

Removal of organics is high in all types of constructed wetlands. VF constructed wetlands are nearly always used for primary or secondary treatment while FWS are often used for tertiary treatment (Kadlec and Knight, 1996) and HF CWs are often used for treatment of wastewater diluted with storm water runoff (Vymazal and Kröpfelová, 2008). In combined systems, the advantages of the HF and VF systems can be combined to complement each other. It is possible to produce an effluent low in BOD, which is fully nitrified and partly denitrified and hence has a much lower total-N concentrations (Cooper, 1999; Cooper, 2001).

Due to the scarcity of fresh water all over the world, balancing the supply and demand of fresh water has always been a great challenge. The recycle and reuse of wastewater is considered as a strategy of water demand management (WDM) system. Reuse of wastewater minimizes the demand for the freshwater (Redwood, 2007). Grey water reuse in many parts of the world, including both industrial and developing countries, has gained significance for last few decades. Many investigations have been conducted on domestic grey water quality analysis, treatment and reuse in the EU, USA, Middle-east countries, Japan and Australia. Grey water treatment systems have been successfully implemented in the US, Japan and Australia to reclaim grey water for non-potable uses. With the technological advancement and public acceptance, grey water seems to be a potential source of water saving (Al-Jayyousi, 2003).

In urban areas like Dhaka in Bangladesh, water is a growing crisis with the rapidly growing rate of urbanization. It is necessary now to explore ways of using resources in a sustainable way to reduce pressure on centralized systems. Through recycling and reuse of generated wastewater, this crisis could definitely be reduced which also will make the currently available fresh water sufficient for the city. This study works on sub-surface flow constructed wetland systems to test whether it can be a reliable treatment method for recycling grey water (typically 60-70 % of total wastewater) which have low organic and nutrient loading. The performance of sub-surface constructed wetlands to treat grey water at household level is tested in this experiment where the removal efficiency of BOD_5, COD, ammonia (total), orthophosphate, TSS, and fecal coliform were tested.

Reuse of this treated water for toilet, car washing, horticulture and cleaning purposes, where the quality of water need not to be of drinking standard, will reduce the demand of fresh water. As the systems produce an effluent low in BOD, which has a much lower total-NH_3 and P, and show good color and less turbidity, the system can be used at community/household level that will help to reduce the water crisis of Dhaka city. In this research, it was intended to evaluate the removal efficiency of subsurface constructed wetlands to reduce the concentration of pollutants that are present in domestic grey water, to improve downstream water quality. Analysis on removal of pollutants by wetland plants and filter materials was also tested.

1.2 Objectives Of The Study

The overall objective of this study was to assess performance of subsurface flow constructed wetland to treat domestic wastewater. The specific objectives were:

- Evaluate the removal performance of pollutants, such as TSS, organic matter, nutrients and fecal coliform, thus improving downstream water quality.
- Analyze removal of pollutants by wetland plants and filter materials.

- Cost analysis of constructed wetland comparing with other options as a treatment system.
- Determine optimum loading and area parameters for hybrid and single-staged wetlands.

1.3 Organization Of The Thesis

The study has been presented in five chapters in this report. Chapter 1 presents a general introduction with importance and objective of the study. Chapter 2 describes literature review on subsurface flow constructed wetlands, its mechanism to remove pollutants and performance in different conditions. Chapter 3 presents the methodology of the study. Chapter 4 shows the results and also discusses the removal efficiency along with some design parameters. Chapter 5 draws the conclusion of the study as well as provides some recommendation for future works.

Chapter 2: Literature Review

2.1 Wetlands

The term wetland describes a diverse Spectrum of ecological systems. Scientific consensus of what constitutes a wetland has been subjectively influenced by definitions that attempt to encompass regulatory and environmental concerns. These concerns have been heightened by historic conversions of wetlands to dry lands and the resulting losses of a variety of natural functions originally provided by the former wetlands. Scientific definitions of wetland types have also been refined as the various structural and functional aspects of these ecosystems have been better described through accelerated research efforts.

A basic understanding of wetland landform will increase an engineer's ability to design constructed wetlands successfully as part of water pollution control systems. This chapter provides a general description of what wetlands are, where they occur, and how they can be constructed for water quality treatment.

2.1.1 Wetland in general

The technical meaning of the term wetland includes a wide range of ecosystems. Areas that are not flooded can still be classified as wetlands because of saturated soil conditions, where water is at or below the ground surface during part of it typical growing season. Wetland areas that are deeply flooded grade imperceptibly into aquatic ecosystems as water depth exceeds the growth limits of emergent or submergent vegetation. Figure 1 shows how wetlands lie on a continuum between dry lands (uplands) and deeply flooded lands (aquatic systems).Because this is a true continuum, with temporal and biological variability, there is no absolute hydrological demarcation between these ecosystems, and all definitions are somewhat arbitrary (IWA, 2000).

Figure 2 shows structural components typical of wetland ecosystems. Starting with the unaltered sediments or bedrock with the wetlands, these typical components are:

- Underlying strata :unaltered organic, mineral or lithic strata, which are typically saturated with or impervious to water and are below the active: rooting zone of the wetlands vegetation
- Hydric soils: the mineral to organic soil layer of the wetland, which is frequently saturated with Water and contains roots, rhizomes, tubers, tunnels. burrows and other active connections to the surface environment
- Detritus: the accumulation of live and dead organic material in a wetland, which consists of dead emergent plant material, dead algae, living and dead animal (primarily invertebrates) and microbes (fungi and bacteria)

- Water: standing water, which provides a habitat for aquatic organisms including fish and other vertebrate animals, submerged and floating plant species that depend on water for buoyancy and support, living algae and populations of microbes
- Emergent vegetation: vascular rooted, hydrophytic plant species, which contain structural components that emerge above the water surface including both herbaceous and woody plant species. (IWA, 2000)

Natural wetlands usually have all of these attributes. Constructed wetlands can have less mature components, especially soil organic matter, which forms over an extended period of time. The structural components of natural wetlands are highly variable and depend on hydrology, underlying sediment types, water quality, and climate and succession maturity (IWA, 2000).

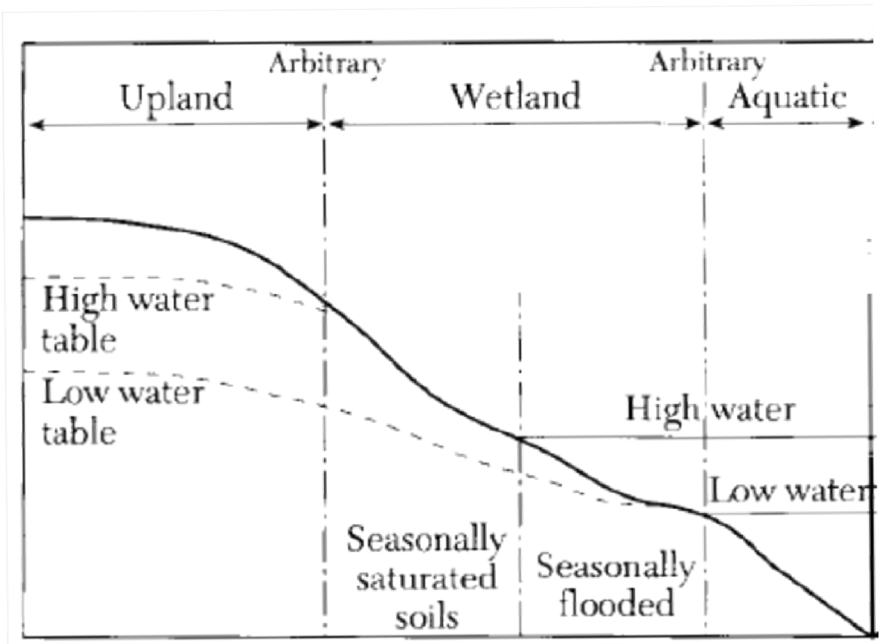

Figure 1: Wetlands are transitional areas between uplands, where excessive water is not a factor for plant growth, and aquatic ecosystems, where flooding excludes rooted, emergent vegetation (kadlec & knight 1996).

2.1.1.1 Hydrology

The water status of a wetland defines its extent and is the determinant of species composition in natural wetlands (Mitsch & Gosselink 1993). Hydrologic conditions also influence the soils and nutrients, which in turn influence the character of the biota. The flows and storage volume determine the length of time that water spends in the wetland, and thus the opportunity for interactions between water-borne substances and the wetland ecosystem.

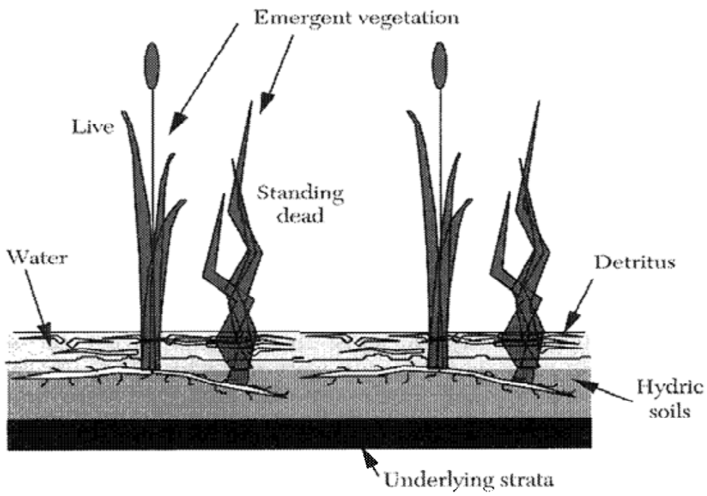

Figure 2: Structural components of wetland.

2.1.1.2 Soils

Many wetland soils are characterized by a lack of oxygen, induced by flooding. Oxygen diffusion in flooded soils is nearly 10,000 times slower than in dry soils (Armstrong 1978). Well-aerated, upland soils rapidly experience a decline in soil oxygen and redox potential when they are flooded. Continuous or seasonal inundation combined with the production of large amounts of dead organic matter (litter fall) results in nearly perpetual soil anaerobic in many wetlands. The resulting lower dissolved oxygen level results in the accumulation of organic matter in wetland soils because of a decreased level of microbial activity and organic decomposition (IWA, 2000).

Wetland soils have a high trapping efficiency for a variety of chemical constituents; they are retained within the hydrated soil matrix by forces ranging from chemical bonding to physical dissolution within the water of hydration. The combined phenomena are referred to as sorption. A significant portion of the chemical binding is cation exchange, which is the replacement of one positively charged ion, attached to the soil or sediment, with another positively charged ion. The humic substances found in wetlands contain large numbers of hydroxyl and carboxylic functional groups, which are hydrophilic and serve as cation binding sites (IWA, 2000).

Soil microbial populations have significant influence on the chemistry of most wetland soils. Important transformations of nitrogen, iron, sulphur and carbon result from microbial processes. These microbial processes are typically affected by the concentrations of reactants as well as the redox potential and pH of the soil. Several nitrogen transformations occur in Wetlands. Organic nitrogen is biologically transformed to ammonia nitrogen through the process of mineralization (=ammonification). Mineralization results as a consequence of the decomposition of organic matter, resulting from the actions of both aerobic and anaerobic microbes. Ammonia is in turn converted to nitrite and nitrate nitrogen through aerobic microbial processes called nitrification. Nitrate nitrogen can be further transformed to nitrous oxide or nitrogen gas in anoxic or anaerobic wetland soils by the action of another group of microbes (denitrifies). Nitrogen gas can also be transformed to organic nitrogen by bacterial nitrogen fixation in some aerobic and some anaerobic wetland soils (IWA, 2000).

When the reduction of nitrate stops by depletion of this electron acceptor, the reduction of ferric oxide starts in waterlogged soils. Ferric oxides are assumed to be one of the most abundant electron acceptors in soils as well as in sediments. The direct enzymic oxidation of Fe^{2+} (and also Mn^{2+}) is continued to a restricted range of organisms; most bacteria cause precipitation of Fe and/or Mn by indirect means by altering E_h or pH, which in turn leads to chemical oxidation and precipitation (Grant & Long 1985).

Sulphate can be reduced to sulphide by obligate anaerobic bacteria in wetlands. The sulphate serves as an electron acceptor in the absence of free oxygen at low redox potentials. Sulphides can provide a source of energy for chemoautotrophic and photosynthetic bacteria in aerobic wetlands, resulting in the formation of elemental sulphur and sulphate. The sulphide is in turn capable of precipitating metal sulphides.

Organic soil carbon is degraded microbially to carbon dioxide by aerobic respiration when oxygen is available, and by fermentation under anaerobic conditions. In fermentation, organic matter serves as the terminal electron acceptor, forming acids and alcohols. Methane can he formed in wetland soils by the action of bacteria at very low redox potentials.

The sediments that form in surface flow treatment wetlands are often different from those that form in natural wetlands, for a number of reasons. First, the enhanced activity of various microbes, fungi, algae and soft bodied invertebrates leads to a greater proportion of fine detritus than leaf, root and stem fragments. There is a significant formation of low-density biosolids (sludge). Secondly, there can be a precipitation of metal hydroxides or sulphides, which add mineral flocks to the sediments. Finally, there is often a high ionic strength associated with effluents being treated, reflected in a high content of dissolved salt. The effect of high ionic strength is to alter the structure of the highly hydrated organic materials that comprise wetland sediments and soils. Some of the same types of material accrete in the pore spaces of subsurface flow (SSF) wetlands (IWA, 2000).

Some measure of performance control can be exerted by the use of specially tailored bed media for constructed treatment wetlands. If sands, soils or gravels are borrowed from natural sources, there will be a period of adaptation as hydric soil properties develop. However; a bed material can be chosen that is manufactured to have a very large phosphorus sorption capacity, such as expanded clay (jenssen et al. 1994). This design philosophy is now quite different from that for most treatment wetlands: the intent is to exhaust a short-term capacity regenerate the wetland and repeat the cycle. This can be a feasible strategy in some cases, provided that the expense of regeneration coupled with its frequency are within acceptable economic bounds.

2.1.1.3 Vegetation

Macrophytic plants provide much of the visible structure of wetland treatment systems. There is no doubt that they are essential for the high levels of water quality improvement typical of most wetland treatment systems. The numerous studies measuring treatment with and without plants have concluded almost invariably that performance is higher when plants are present.

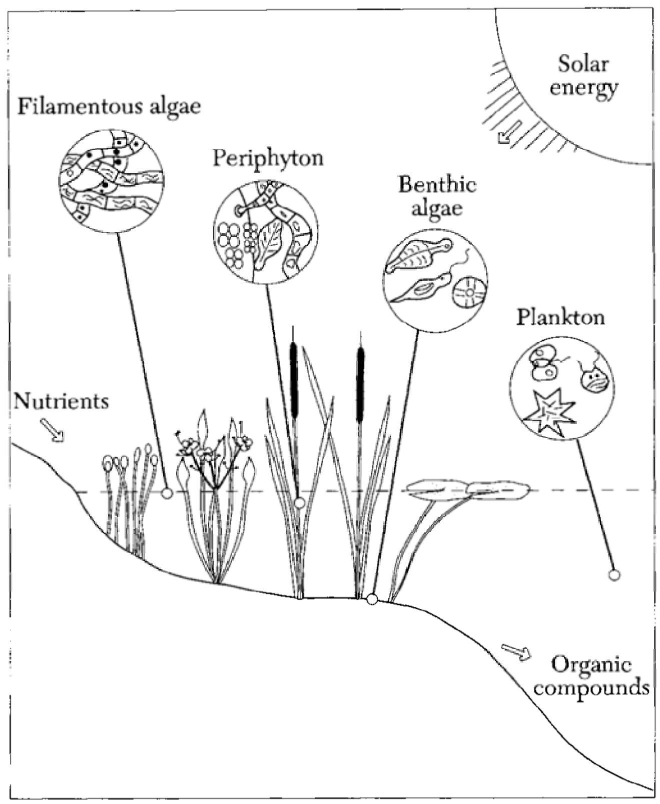

Figure 3: Algae and macrophytes in treatment wetlands (Kadlec &Knight 1996).

This finding led some researchers to conclude that wetland plants were the dominant source of treatment because of their direct uptake and sequestering of pollutants. It is now known that plant uptake is the principal removal mechanism only for some pollutants and some types of treatment wetland (for example with free-floating plants with a regular harvest) and only in lightly loaded systems. During an initial successional period of rapid plant growth, direct pollutant immobilization in wetland plants can be important. For many other pollutants, plant uptake is generally of minor importance compared with microbial and physical transformations that occur within most wetlands. Macrophytic plants are essential in wetland treatment systems because they provide structure and a source of reduced carbon for the microbes that mediate most of the pollutant transformations that occur in wetlands. The diversity of wetland plant adaptations provides the wetland treatment system designer with numerous options and potential problems. Some plant species produce large amounts of carbon that are able to support heterotrophic microbes important in nutrient transformations. Other plant species provide shading of the water surface, in turn controlling algal growth and concentrations of suspended Solids in the discharge from the wetland treatment system. An understanding of the ecological properties of these wetland plant species is essential for the successful design, construction and operation of wetland treatment systems (IWA, 2000).

2.2 Constructed Wetlands

2.2.1 Technology description

A wetland is a complex assemblage of water, substrate, plants (vascular and algae), litter (primarily fallen plant material), invertebrates (mostly insect larvae and worms) and an array of microorganisms (most importantly bacteria). The mechanisms that are available to improve water quality are therefore numerous and often interrelated. These mechanisms include:

- Settling of suspended particulate matter
- Filtration and chemical precipitation through contact of the water with the substrate and litter
- Chemical transformation
- Adsorption and ion exchange on the surfaces of plants, substrate, sediment and litter
- Breakdown, and transformation and up- take, of pollutants and nutrients by micro-organisms and plants
- Predation and natural die-off of pathogens. The most effective treatment wetlands are those that foster these mechanisms.

Constructed wetlands are a cost-effective and technically feasible approach to the treatment of wastewater and runoff for several reasons:

- Wetlands can be less expensive to build than other treatment options
- Operation and maintenance expenses(energy and supplies) are low U operation and maintenance require only periodic, rather than continuous, on-site labor
- Wetlands are able to tolerate fluctuations in flow
- Wetlands are able to treat wastewaters with low organic load (too low for activated sludge)
- They facilitate water reuse and recycling. In addition: 0 they provide habitat for many wetland organisms
- They can be built to fit harmoniously into the landscape
- They provide numerous benefits in addition to water quality improvement, such as wildlife habitat and the aesthetic enhancement of open spaces
- They are an environmentally sensitive app- roach that is viewed with favor by the general public.

Wetland treatment systems use water tolerant plant species and shallow, flooded or saturated soil conditions to provide various types of wastewater treatment. The two basic types of wetland treatment systems include constructed free water surface (FWS) or surface flow (SF) wetlands, and constructed SSF wetlands.

Constructed wetlands mimic the optimal treatment conditions found in natural wetlands but provide the flexibility of being constructible at almost any location and can be used for treatment of primary and secondary waste waters as well as waters from a variety of other sources including storm waters, landfill leach ate, industrial and agricultural wastewaters, and acid-mine drainage.

Surface flow wetlands are densely vegetated by a variety of plant species and typically have water depths less than 0.4 m. Open water areas can be incorporated into a design to provide for the optimization of hydraulics and for wildlife habitat enhancement. According to WPCF (1990), typical hydraulic loading rates are between 0.7 and 5.0 cm d^{-1}(between 2 and 14 ha per 1000 m^3 d^{-1}) in constructed surface flow treatment wetlands.

SSF wetlands use a bed of soil or gravel as a substrate for the growth of rooted emergent wetland plants. Pretreated wastewater flows by gravity, horizontally or vertically, through the bed substrate, where it contacts a mixture of facultative microbes living in association with the substrate and plant roots. The bed depth in SSF flow wetlands is typically between 0.6 and 1.0 m, and the bottom of the bed is sloped to minimize water flow overland.

Most frequently used species in SSF constructed wetlands are common reed (phragmites australis), cattail (Typha spp.), bulrush (Scirpus spp.), reed canary grass (Phalaris arundinacea) and sweet manna grass (Glyceria fnaxifrul) Some oxygen enters the bed substrate by direct atmospheric diffusion and some through the plant, resulting in a mixture of aerobic and anaerobic zones. Most of the saturated bed is anoxic or anaerobic under most wastewater design loadings. According to WPCF (1990), typical hydraulic loading rates in SSF wetlands range from 2 to 20 cm d^{-1} (from 0.5 to 5 ha Dm^{-3} d^{-1}).

Wetlands have been found to be effective in treating biochemical oxygen demand, suspended solids, nitrogen and phosphorus, as well as for decreasing the concentrations of metals, organic chemicals and pathogens. Effective wetland performance depends on adequate pretreatment, conservative constituent and hydraulic loading rates, the collection of monitoring information to assess system performance, and knowledge of successful operation strategies.

A common difficulty experienced by wetland treatment systems has been inadequate oxygen supply. When wetland systems are overloaded by oxygen-demanding constituents, or are operated with excessive water depth, highly reduced conditions occur in the sediments, resulting in plant stress and decreased removal efficiencies for biochemical oxygen demand and ammonia nitrogen. A common problem encountered in SSF constructed wetlands is an inadequate hydraulic gradient and resulting surface flows (IWA, 2000).

2.3 Types Of Constructed Wetland

2.3.1 Free water surface treatment wetlands

The FWS wetland technology started with the ecological engineering of natural wetlands for wastewater treatment (Ewel & Odum 1984; Kadlec & Tilton 1979). Constructed FWS treatment wetlands mimic the hydrological regime of natural wetlands. In surface flow (SF) wetlands, water flows over the soil surface from an inlet point to an outlet point or in a few cases, is totally lost to evapotranspiration and infiltration within the wetland (IWA, 2000).

FWS treatment wetlands have some properties in common with facultative lagoons and also have some important structural and functional differences. Water column processes in deeper zones within treatment wetlands are nearly identical to ponds with surface autotrophic zones dominated by plank tonic or filamentous algae, or by floating or submerged aquatic macrophytes. Deeper zones tend to the dominated by anaerobic microbial processes in the absence of light. However, shallow emergent macrophyte zones in treatment wetlands and aerobic lagoons can be quite dissimilar. Emergent wetland plants tend to cool and shade the water. Net carbon production in vegetated wetlands tends to be higher than that in facultative ponds because of high gross primary production in the form of structural carbon, accompanied by resistance to degradation and low rates of' decomposition of organic carbon in the oxygen-deficient Water column. This high availability of' carbon and the short diffusional gradients in shallow vegetated wetlands result in differences in biogeochemical cycling compared with ponds and lagoons (IWA, 2000).

During the process of elemental cycling within the wetland, chemical free energy is extracted by the hoterotrophic biota, and fixed carbon and nitrogen are lost to the atmosphere. A smaller portion of the phosphorus and other non-volatile elements can be lost from the mineral cycle and buried in accreting sediments within the wetland. Wetlands are autotrophic ecosystems, and the additional fixed carbon and nitrogen from the atmosphere is processed simultaneously with the pollutants introduced from the wastewater source. The net effect of' these complex processes is a general decrease in pollutant concentrations between the inlet and outlet of treatment wetlands. However, because of the internal autotrophic processes of the wetland, outflow pollutant concentrations are seldom zero, and in some cases for some parameters they can exceed inflow concentrations (IWA, 2000).

2.3.2 Subsurface flow treatment wetlands

Many of the earliest treatment wetlands in Europe were SSF systems constructed to treat mechanically pretreated municipal waste waters. Soil- and gravel-based SSF wetlands are still the most prevalent application of this technology in Europe (Cooper et al. 1996; Brix 1994;

Vymazal et al. 1998). SSF wetlands that use gravel substrates have also been used extensively in the United States (Reed 1992). This technology is generally limited to systems with low flow rates and can be used with less than secondary pretreatment.

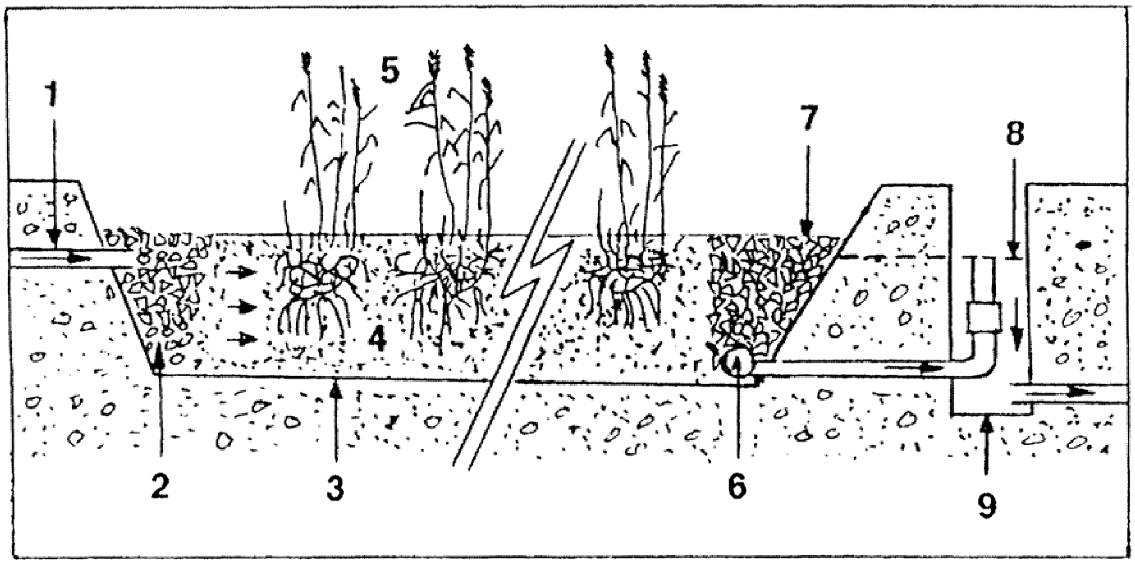

Figure 4: Longitudinal section of a constructed wetland with horizontal SSF.

Key: 1, inflow of mechanically pretreated waste water;2, distribution zone filled with large stones; 3, impermeable liner; 4, medium(e.g. gravel, sand, crushed stones); 8 water level in the bed maintained with outlet structure, 9, out flow (Vymazal 1997).

2.3.2.1 Horizontal-flow systems

Figure 4 shows a typical arrangement for the constructed wetland with a horizontal flow (HF). It is called ‘horizontal flow’ because the wastewater is fed in at the inlet and flows slowly through the porous medium under the surface of the bed in a more or less horizontal path until it reaches the outlet zone, where it is collected and discharged at the outlet (see Figure 2.6). During this passage, the waste water will come into contact with a network of aerobic, anoxic and anaerobic zones. The aerobic zones occur around roots and rhizomes that leak oxygen into the substrate. During the passage of the wastewater through the rhizosphere, the wastewater is cleaned by microbiological degradation and by physical and chemical processes (Brix 1987; Cooper et al. 1996). In Europe, the most common term for HF constructed wetlands is the RBTS, because a frequently used plant is common reed (Phragmites australis). However, reed canary grass (Plalaris arundinacea), sweet manna grass (Glyceria maxima) and cattails (Typha spp.) are also used in Europe. In the USA, bulrushes (Scripus spp.) are also used. In North America the term vegetated submerged bed (VSB) is also used (IWA, 2000).

The concept of treating wastewater in constructed wetlands with horizontal subsurface flow was developed in Germany in the 19705. The first operational constructed wetland was started in 1974 in Othfresen in Germany and the treatment process was called the RZM (Kickuth 1977). The RZM system consists of a plastic-lined bed containing emergent macrophytes growing in soil. However, these soil-based systems, as a result of low hydraulic conductivity of the soil media, suffered from surface runoff, preventing the wastewater from coming into contact with the rhizosphere. The problem of surface runoff was overcome by the use of more porous media such as gravel (Cooper 1990).

Organic compounds are degraded aerobically as well as an aerobically by bacteria attached to plant underground organs (that is, roots and rhizomes) and media surfaces. The oxygen required for aerobic degradation is supplied directly from the atmosphere by diffusion or oxygen leakage from the macrophyte roots and rhizomes in the rhizosphere. Numerous investigations have shown that the oxygen transport capacity of the reeds is insufficient to ensure aerobic decomposition in the rhizosphere and that anoxic and anaerobic decomposition are important in HF constructed wetlands (for example, Brix 1990).

Settleable and SS that are not removed in pro-treatment systems are effectively removed by filtration and settlement, Settlement will take place in quiescent areas of any HF constructed wetland (Cooper et al. 1996).

Nitrogen is removed in HF constructed wetlands by nitrification and denitrification, volatilization, adsorption and plant uptake. The major removal mechanism of nitrogen in HF constructed wetlands is nitrification and denitrification. Ammonia is oxidized to nitrate by nitrifying bacteria in aerobic zones, and nitrates are converted to gaseous nitrogen by denitrifying, bacteria in anoxic zones (Cooper el al. 1996). Field measurements have shown that the oxygenation of the rhizosphere of HF constructed wetlands is insufficient and that incomplete nitrification is therefore the major cause of limited nitrogen removal. Volatilization, plant uptake and adsorption are much less important in nitrogen removal.

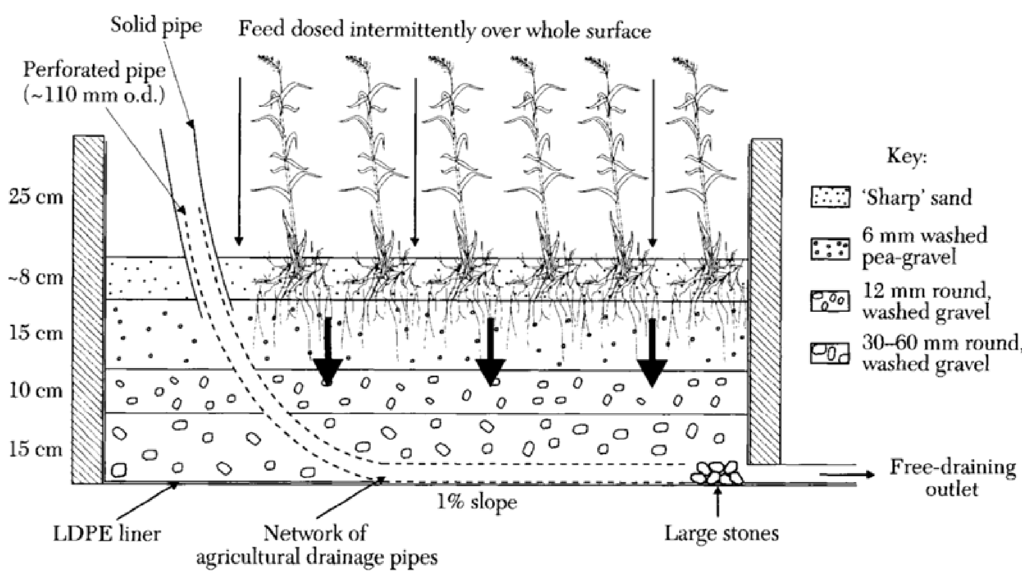

Figure 5: Typical arrangement of a VF reed bed system (Cooper 1996).

Phosphorus is removed from Wastewater in HF wetlands primarily by ligand exchange reactions, in which phosphate displaces water or hydroxyl ions from the surface of Fe and Al hydrous oxides. However, media used for HF wetlands (such as pea gravel or crushed stones) usually do not contain great quantities of Fe, Al or Ca, and therefore the removal of phosphorus generally low (IWA, 2000).

2.3.2.2 Vertical-flow systems

Vertical-flow (VF) treatment wetlands are frequently planted with common reed. Other emergent wetland plants such as cattails or bulrush can also be used. VF reed beds typically look like the system shown in Figure 5. They are composed of a flat bed of gravel upped with sand, with reeds growing at the same sort of densities as HF systems. They are fed intermittently. The liquid is dosed on the bed in a large batch, flooding the surface. The quid then gradually drains vertically down through the bed and is collected by a drainage network at the base. The bed drains completely free, allowing air to refill the bed. The next dose of liquid traps this air and this together with the aeration caused by the rapid dosing on the bed leads to good oxygen transfer and hence the ability to decompose BOD and to nitrify ammonia nitrogen (Cooper et al. 1996).

As with the HF systems, the reeds in VF systems will transfer some oxygen down into the rhizosphere, but it will be small in comparison with the oxygen transfer created by the dosing system. VF treatment wetlands are very similar in principle to a rustic biological filter (Cooper et al. 1996). They are less good at the removal of SS and in most cases will be followed by a HF bed as part of a multistage treatment wetland system.

The earliest form of VF system is that of Seidel in Germany in the 1970s, sometimes called the Max Planck Institute Process (MPIP) or the Krefeld Process. Interest in the particular process seemed to wane, but it has been revived in the past six years because of the need to produce beds that nitrify. Operators and designers were disappointed in the ability of the early HF systems to oxidize ammonia to nitrate. In retrospect, this was clearly related to the fact that the ability of the reeds to transfer oxygen was greatly overestimated. Most HF systems have very low levels of dissolved oxygen in the effluent. Under these circumstances there will be no oxygen remaining to oxidize the ammonia nitrogen to nitrate. Because of this poor performance, designers and researchers started looking for alternative designs of reed bed that could oxidize the ammonia nitrogen (IWA, 2000).

2.4 Applications Of The Technology

There are an expanding number of application areas for constructed wetlands technology. During the early years (pre-1985) of the development of the technology, virtually all emphasis was on the treatment of domestic and municipal wastewater. In recent years there has been a branching to include a very broad spectrum of wastewaters, including industrial and storm waters.

2.4.1 Domestic and municipal wastewaters

There are several roles for constructed wetlands in the treatment of domestic and municipal wastewaters. They can be positioned at any of several locations along the water quality improvement path. The commonly accepted terminology for describing that path is as follows (Metcalf & Eddy 1991)

- **Preliminary treatment** of wastewater is defined as the removal of wastewater constituents that might cause maintenance or operational problems with the treatment operations, processes and ancillary systems. Examples of preliminary operations are screening and comminution for the removal of debris and rags, grit removal for the elimination of coarse suspended matter that might cause wear or clogging of equipment, and flotation for the removal of large quantities of oil and grease.
- **In primary treatment**, a portion of the SS and organic matter is removed from the wastewater. This removal is usually accomplished with physical operations such as screening and sedimentation. More advanced methods of primary treatment include those that also provide a partial biodegradation of organic compounds. Frequently used units are primary clarifiers for larger flows and septic and Imhoff tanks for smaller applications. The effluent from primary treatment will ordinarily contain considerable organic matter and will have a relatively high BOD.
- **Secondary treatment** is directed principally towards the removal of biodegradable organics and SS. The most common secondary treatment technologies include activated sludge process, rotating biological contactors (so-called biodiscs), oxidation ditches and trickling filters (bacterial beds). Disinfection is frequently included in the definition of conventional secondary treatment in the USA but not in Europe, where it is less frequently applied.
- **Advanced treatment** of wastewater is defined as the level of tertiary treatment required beyond conventional secondary treatment to remove constituents of concern including nutrients, decreased levels of nitrogen (ammonia), toxic compounds and increased amounts of organic material and SS. Disinfection is typically regarded as tertiary treatment in Europe.

Constructed wetland technology is generally applied in two general themes for domestic and municipal wastewaters: for accomplishing secondary treatment and for accomplishing advanced treatment.

2.4.2 Wetlands for secondary treatment

2.4.2.1 Subsurface flow

Constructed SSF wetland treatment systems can provide secondary treatment of municipal or domestic wastewater after mechanical pretreatment consisting of a combination of screens, grit and grease chambers, sedimentation, septic and Imhoff tanks. The number of SSF constructed wetlands in operation in Europe is at present ca. 5000. In Germany alone, nearly 3500 systems are in operation (Borner et al. 1998). Many systems are also in operation in Denmark (200-400), the UK (400-600), Austria (ca. 160), Czech Republic (ca. 80), Poland (ca. 50), Slovenia (ca, 20) and Norway (ca. 10). In general, most European SSF treatment wetlands are designed to treat domestic or municipal wastewaters from sources of less than 500 population equivalent (PE). However most systems are designed for small sources of pollution (less than 50 PE) and many systems are designed for single households. Only a small number of systems were designed for larger sources of pollution (more than 1000 PE) (Vymazal et al. 1998a).

A common local problem faced by home owners and others in rural and non-sewered areas is poor site conditions that do not permit the installation and satisfactory performance of conventional on-site systems such as septic tank rain-fields. Practical solutions are needed, and there is great interest and desire in abating water pollution with effective, simple, reliable and affordable wastewater treatment processes. In recognition of this need, the Tennessee Valley Authority (TVA) began a demonstration of the constructed wetlands technology in 1986 as an alternative to conventional, mechanical processes, especially for small communities. Constructed wetlands can be scaled down from municipal systems to small systems, such as those for schools, camps and even individual homes. The systems are effective, simple, affordable, aesthetically pleasing, and educational. Guidelines have been developed by TVA provide state-of-the-art and simple instructions for designing, constructing and operating instructed Wetlands for small wastewater flows (TVA 1991).

An on-site SSF wetland can be a discharge system (i.e. discharges to a surface water) or a non-discharge system (discharges to surface waters are eliminated by percolation, aided by evaporation and transpiration). A non-discharge system is used where conventional on-site methods are ineffective owing to poor site conditions (e.g. low soil percolation, shallow soils, high groundwater table or Karst topography). A non-discharge system is classified as 'on-site' if it is located within the property boundaries of the owners producing wastewater. The smallest systems are for single houses with limited wastewater. Different rules often apply to systems treating lower flows. A frequent use of on-site SSF wetlands is to replace failed adsorption fields or as an alternative to conventional systems where percolation rates are low. The technology might also be an alternative to low pressure mound systems by constructing the on-site SSF system on top of bedrock, impermeable clay or high groundwater. Systems can e reliably designed to meet 'secondary' level permit limits (IWA, 2000).

In the early 1980s, when SSF constructed Wetlands were introduced, the system usually consisted of only one bed, regardless of size, Hydraulic problems led to a changed approach. At present, for larger systems (larger than ca. 50 PE or 5 m^3 d^{-1}), a multi-cell configuration is used. The most frequently used configurations are presented in Figure 6. The cells are usually rectangular with aspect ratios (length: width ratios) of between 0.3 and 3 (IWA, 2000).

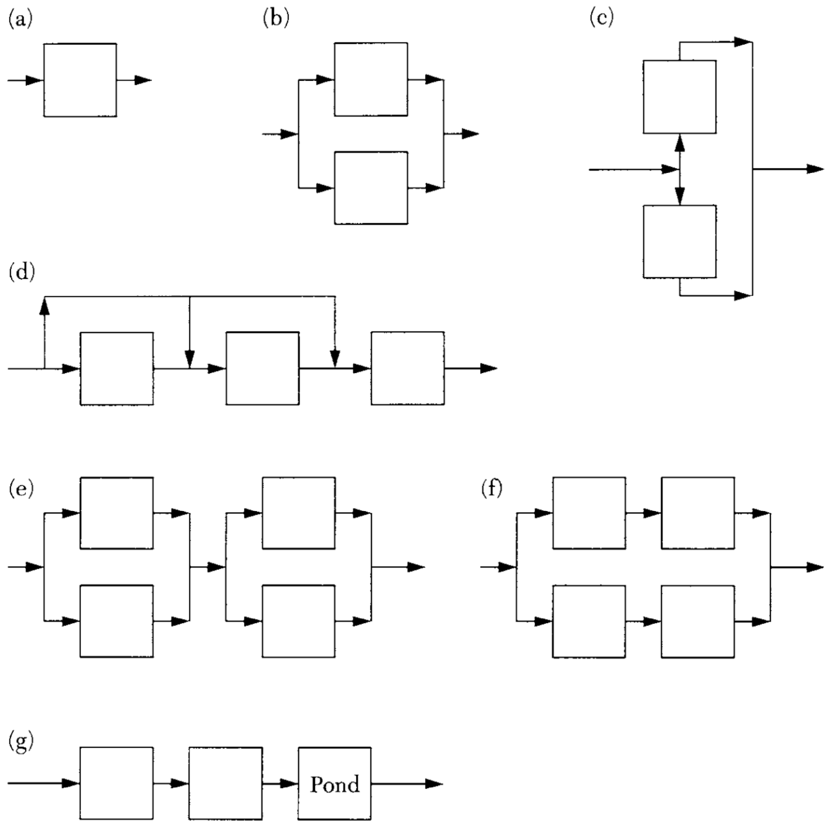

Figure 6: Alternative multi-cell SSF configurations: (a)single bed; (b, c) two parallel cells; (d) beds in series with a bypass; (e) two parallel cells in a series; (f) two series cells in parallel; (g) pond as a final step (Vymazal 1998a).

At present, most systems use coarse media (pea gravel, crushed stones) with a size fraction between 5 and 32 mm. Common reed (Phragmites australis) is the most frequently used plant in Europe, but reed canary grass (Phalaris aroundinacea), cattails (Typha spp.) and sweet manna grass (Glyceria maxima) are also used either singly or in combination with common reed.

The treatment performance obtained in constructed wetlands with subsurface horizontal water flow is good in terms of removal of SS and BOD but lower in terms of nutrient removal (Tables 1 and 2). However, the treatment capacity of the SSF systems in terms of nutrient removal is low but comparable to the treatment efficiency of conventional treatment systems without a special regime for nutrient removal (i.e. nitrification and denitrification, phosphorus precipitation). Selection of system design should carefully consider the desired final effluent quality. Where only the removal of SS and BOD is required and where land is readily available and inexpensive, SF systems and one-unit SSF systems can be used. In sites with more stringent effluent quality demands, including demands for the removal of nitrogen and phosphorus, combined systems consisting of VF beds with intermittent loading followed by horizontal SSF beds should be selected. The medium in the beds should be selected on the basis of hydraulic conductivity and phosphorus-binding capacity. Such multi- stage systems are more expensive in terms of construction, operation and maintenance than one-unit SF and SSF systems, but they might still be significantly cheaper than 'high- technology' alternatives (IWA, 2000).

2.4.2.2 Surface flow

Constructed FWS wetlands are typically not used at this time for complete secondary treatment of municipal wastewater. However, there are applications for secondary treatment in the USA and elsewhere that might serve as models by which to judge the success of this application. It is sometimes advantageous to supplement an undersized conventional secondary treatment plant with wetlands to bring the combination back to compliance with secondary standards. This was the design goal for Columbia, Missouri (Brunner et *al.*1993), and for Wetwang, UK (Hiley 1990).

Facultative lagoons can provide secondary effluents, but they suffer from operational problems that might sometimes be best solved by adding a constructed wetland. For instance, warm summer temperatures can create algal populations that create TSS in excess of secondary standards. Wetlands can provide the TSS removal to bring the system into compliance. An example of this use of FWS wetlands is the Ouray, Colorado, USA, system (Andrews & Cockle 1996). The lagoon treatment at that site is over-taxed during the summer months and cannot meet a 30 mgl^{-1}standard on a seasonal basis, although it is achieved on an annual basis. The addition of a treatment wetland lowers the annual BOD and decreases the seasonal values below 30 mg l^{-1} (IWA, 2000).

2.4.3 Tertiary and higher treatment

Data from North American treatment wetlands receiving secondary or better influents were summarized by the NADB (1993) and Kadlec & Knight (1996).

2.4.3.1 Subsurface flow

A vast quantity of data on tertiary treatment systems is now available via the database gained from the Severn Trent Water and UK water companies' experiences. Green & Upton (1995) described the effluent quality in BOD, TSS, ammonia and total organic nitrogen performance for 29 sites for the calendar year 1993. On the basis of these data, it is clear that designing a tertiary treatment SSF wetland at 1m^2 per PE will achieve an effluent of less than 5 mg BOD$_5$l^{-1} and 10 mg TSSl^{-1}, and in many cases will achieve very substantial nitrification. In Severn Trent it has become standard practice to use 0.7 m^2 per PE tor tertiary treatment (Green & Upton 1995); smaller areas per head are used for some short-term or remedial applications.

Table 1: Average influent and effluent concentrations and mass loading rates of various pollutants in soil-based subsurface HF constructed reed beds in Europe.

Parameter	n	Influent Mean	Influent SD	Effluent Mean	Effluent SD
Concentrations (mg l^{-1})					
SS	77	98.6	81.6	13.6	11.1
BOD_5	80	97.0	81.0	13.1	12.6
TN	73	28.5	14.7	18.0	10.7
TP	67	8.6	4.5	6.3	3.5
Mass loading rates (g m^{-2} d^{-1})					
SS	51	5.22	6.37	1.06	1.50
BOD_5	66	4.80	5.97	0.89	1.34
TN	57	1.15	0.79	0.78	0.77
TP	50	0.33	0.27	0.26	0.26

Summarized by Brix (1994b) with data coombs (1990) and schierup et al.(1990a).

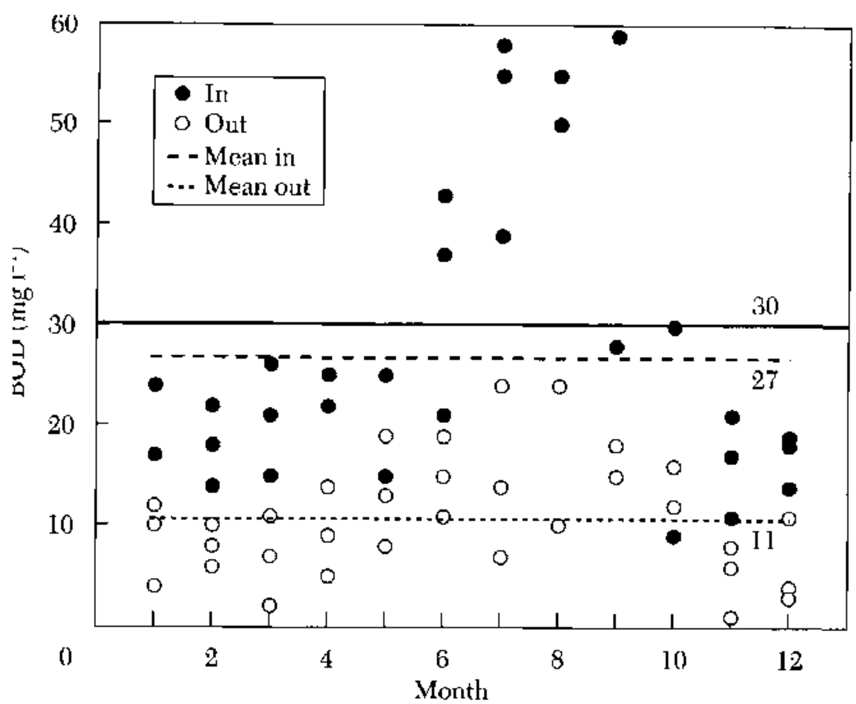

Figure 7: The use of FWS wetlands in Ouray, Colorado, USA, in 1993-95 to treat lagoon effluent(based on data from Andrews & Cockle (1996)).

Table 2: Average influent and effluent concentrations and mass loading rates of various pollutants in gravel-based subsurface HF constructed reed beds in the Czech Republic (Vymazal 1998b).

Parameter	n	Influent		Effluent	
		Mean	SD	Mean	SD
Concentrations (mg l^{-1})					
SS	37	71.9	47.2	10.8	7.1
BOD_5	39	87.4	65.7	11.9	11.4
TN	26	46.1	18.5	27.6	9.7
TP	27	6.4	3.8	3.1	2.1
Mass loading rates (g m^{-2} d^{-1})					
SS	31	3.34	3.11	0.44	0.42
BOD_5	35	3.36	2.86	0.53	0.67
TN	26	1.39	0.91	0.80	0.16
TP	24	0.30	0.18	0.18	0.16

2.4.3.2 Surface flow

SF wetlands in North America normally receive municipal water of approximately secondary quality or better, this is in contrast with the subsurface technology of northern Europe, which typically treats settled or primary influents. There are several hundred FWS treatment wetlands in the USA that are polishing secondary or tertiary wastewaters (NADB 1993).

In the USA, from a regulatory standpoint, there is fairly strong emphasis on creating FWS treatment wetlands of a moderately high quality. The unstated principle is one of minimizing the exposure of wildlife to poor quality waters and habitat. As a result, most of the available FWS design data from the USA are in the lower ranges of concentration. It implies that incoming water quality is near enough to wetland background for those baseline numbers to influence design.

Reducing phosphorus levels is one of the least efficient processes in wetland treatment. Low TP concentrations can be decreased still further, but a large P load removal requires a large wetland area. Consequently, some pretreatment for decreasing high TP concentrations is normally cost effective. The addition of iron and alum are the most frequent choices. For FWS wetlands, the point of addition needs to be upstream of the wetland because there is no effective way of providing chemical contacting in the wetland itself. The SSF wetland has an advantage in this regard because the media can be amended with the P-removing chemicals. The oxidation of ammonium nitrogen is more efficient when there is a small diffusional resistance to providing the oxygen to the dissolved or sorbed nitrogen. Neither horizontal SSF nor FWS wetlands are particularly good in this regard because the reaeration potential of the water sheet is relatively low. However, this tendency towards anaerobiosis is quite beneficial for the reduction of oxidized nitrogen to nitrous oxide and nitrogen gas. The greatest efficiency for N reduction is therefore achieved when the wetland is assisted by some form of nitrification pretreatment. Mechanical nitrification devices, planted or unplanted sand or gravel filters, or VF wetlands are candidates for the provision of supplemental nitrification (IWA, 2000).

2.5 Plants and Planting

2.5.1 Varieties of vegetation

Constructed wetlands can be planted with a number of adapted, emergent wetland plant species. Wetlands created as part of compensatory mitigation or for wildlife habitat typically include a large number of planted species. However in constructed wetland treatment systems, diversity is typically quite low.

2.5.1.1 Subsurface-flow wetland vegetation

The three genera of wetland plants that are most frequently used in SF wetland treatment systems are also used in SSF wetlands. Commonly used plants are Phalaris arundinacea (reed canary grass), Typha spp. (cattails), Scirpus spp, (bulrushes) and Glyceria maxima (sweet mannagrass). However, the most frequently used plant species worldwide is Phragmites australis (common reed). This species has remarkable growth rates, root development and tolerance to saturated soil conditions. They are known to provide some ancillary benefits in terms of wildlife habitat in the UK (Merritt 1994).

Phragmites is planted by using rhizomes, seedlings or field-harvested reeds. All of these techniques are effective if the plants are healthy and if adequate (but not excessive) soil moisture is maintained during plant establishment.

Planting densities between 2 and 6 m^{-2} (20,000-60,000 ha^{-1}) are normally recommended for Phragmites (Cooper 1990; ATV 1989).

A gravel bed will require planting because seed banks are typically lacking and the medium is not optimal for germination. If a portion of the bed remains flooded, a litter layer can develop that is conducive to the germination of wetland plant seeds, thus permitting invasion. More frequently, a portion of the bed can remain too dry, permitting invasion by terrestrial species (weeds).

The presence of macrophytes is important for many, if not all, pollutant-removal functions in SSF wetlands also. However, the question of which plant might be best has not yet been resolved. The results of various side-by-side investigations are inconclusive, as will be discussed for ammonium removal. The project at Santee, California, USA, ranked Scirpus best, Phragmites second and Typha a distant third, close to no plants (fourth) (Gersberg *et al.* 1984). The project at Lake Buena Vista, Florida, USA, ranked Sagittaria better than Scirpus (DeBusk *et al.* 1989). The project at Hamilton, New Zealand, ranked Glyceria better than Schoenoplectus (Scirpus) better than no plants (van Oostrom & Cooper 1990). The project at Pretoria, South Africa, ranked Phragmites better than Scirpus better than Typha for lagoon effluent, but Scirpus better than Typha better than Phragmites for settled sewage (Batchelor *et al.* 1990). Bavor *et al.* (1988) found very little difference between Schoenoplectus (Scirpus) and Typha and no plants at

Richmond, NSW, Australia. At Hardin, Kentucky, USA, Phragmites was better than Scirpus (NADB 1993).

All of these results read like a set of football game results: the reader can use a sequence of his or her choice to prove that a particular plant is better than another, just as game scores can be used to establish one team's superiority.

Wetland plants require nutrients for growth and reproduction, and the rooted macrophytes take up nutrients primarily through their root systems. Some uptake also occurs through immersed stems and leaves from the surrounding water. Because wetland plants are very productive, considerable quantities of nutrients can be bound in the biomass. The uptake capacity of emergent macrophytes, and thus the amount that can be removed if the biomass is harvested, is roughly in the range 30-150 kg P $ha^{-1}yr^{-1}$ and 200-250O kg N $ha^{-1}yr^{-1}$ (Brix & Schierup 1989; Gumbricht 1993*a, b*; Brix 1994). The highly productive Eichhornia crassipes (water hyacinth) has a higher uptake capacity (approx. 350 kg p and 2000 kg N $ha^{-1}yr^{-1}$), whereas the capacity of submerged macrophytes is lower (less than 100 kgP and 700 kg N $ha^{-1}yr^{-1}$). However, the quantities of nutrients that can be removed by harvesting are generally insignificant in comparison with the loading into the constructed wetlands with the wastewater (Brix 1994; Geller 1996). If the wetlands are not harvested, the vast majority of the nutrients that have been incorporated into the plant tissue will be returned to the water by decomposition processes. Long-term storage of nutrients in the wetland systems results from the undecomposed fraction of the litter produced by the various elements of the biogeochemical cycles as well as the deposition of refractory nutrient-containing compounds (Kadlec & Knight 1996).

It is well documented that aquatic macrophytes release oxygen from roots into the rhizosphere and that this release influences the biogeochemical cycles in the sediments through the effects on the redox status of the sediments (Barko *et al.* 1991; Sorrell and Boon 1992). Qualitatively this is easily detected by the reddish color associated with oxidized forms of iron on the surface of the roots. However, the quantitative magnitude of the oxygen release under conditions in .situ remains a matter of controversy (Bedford *et al.* 1991; Sorrell and Armstrong 1994).

Oxygen release rates from the roots depend on the internal oxygen concentration, the oxygen demand of the surrounding medium and the permeability of the root walls (Sorrell & Armstrong 1994). Wetland plants conserve internal oxygen because of suberized and lignified layers in the hypodermis and outer cortex (Armstrong & Armstrong 1988). These stop radial leakage outwards, allowing more oxygen to reach the apical meristem. Thus, wetland plants attempt to minimize their oxygen losses to the rhizosphere. Wetland plants do, however, leak oxygen from their roots. Rates of oxygen leakage are generally highest in the sub-apical region of roots and decrease with distance from the root apex (Armstrong 1979). The oxygen leakage at the root tips serves to oxidize and detoxify potentially harmful reducing substances in the rhizosphere. Species possessing an internal convective through-flow ventilation system have higher internal oxygen concentrations in the rhizomes and roots than species relying exclusively on the diffusive transfer of oxygen (Armstrong & Armstrong 1990), and the convective through-flow of gas significantly increases the root length that can be aerated, in comparison with the length by diffusion alone (Brix 1994). Wetland plants with a convective through-flow mechanism therefore have the potential to release more oxygen from their roots than species without convective through-flow.

Root systems also release other substances besides oxygen. In some early studies Dr K. Seidel from the Max Planck Institute in Germany showed that the bulrush Schoenoplectus released antibiotics from its roots (Seidel 1964, 1966). A range of bacteria (coli-forms, Salmonella and enterococci) obviously disappeared from polluted water by passing through a vegetation of bulrushes. It is also well known that a range of submerged macrophytes releases compounds that affect the growth of other species. However, the role of this attribute in treatment wetlands has not yet been experimentally verified. Plants also release a wide range of organic compounds by the roots (Rovira 1965, 1969; Barber & Martin 1976). The magnitude of this release is still unclear, but reported values are generally 5-25% of the photo synthetically fixed carbon. This organic carbon exuded by the roots might act as a carbon source for denitrifiers and thus increase nitrate removal in some types of treatment wetland (Platzer 1996).

2.6 Mechanisms And Results For Water Quality Improvement

2.6.1 Suspended solids

The existence of a subsurface air/water interface causes sediment processing in the SSF wetland to differ considerably from that in SF wetlands. Macrophyte leaf and seed litter are mostly contained on the surface of the bed and do not interact with the water flowing in the interstices below. Most vertebrates and invertebrates do not interact with the water. Resuspension is not caused by wind or vertebrate activities.

However, many particulate processes do operate in the water-filled voids. Particles settle into stagnant micropockets or are strained by flow constrict-ions. They can also impinge on substrate granules and stick as a result of several possible inter-particle adhesion forces. These physical processes are termed granular medium filtration (Metcalf & Eddy 1991). Higher velocities can dislodge adhering or deposited material, which forms the basis for the back-washing method of filter regeneration. Generation of particulate material can occur via all the mechanisms shown for FWS wetlands. Below-ground macrophyte parts - roots and rhizomes - die, decay and produce fine detrital fragments. Many other organisms are present in the bed that can contribute to TSS via the same route: algae, fungi and bacteria all die and contribute particulate matter to the water flowing in the pore space. These micro-organisms arc unevenly distributed spatially within the gravel bed, with more organisms located near the inlet and near the bottom (Bavor *et al.* 1988).

2.6.1.2 Clogging

In SSF bed, particulate matter accumulates in voids, blocking them. This clogging process is counteracted by the decomposition of organic particulates. At a minimum, the mineral content of the trapped solids contributes to pore blockage.

Root growth decreases the available pore space in SSF wetlands. Studies on beds with bulrushes have shown that roots and rhizomes are typically located in the upper 30 cm of the bed (US Environmental Protection Agency 1993). Phragmites roots and rhizomes have been reported to penetrate further in some instances (Gersberg *et al.* 1986), but other investigations show only 20-40 cm penetration (Schierup 1990; Saurer 1992). The below-ground biomass of Phragmites is on the order of 2000 g dry matter m^{-2}, which approximates a quarter of the void volume in a 30 cm root zone.

The end result of subsurface biological and vegetative activity is the build-up of solids within the pore spaces of the medium. That build-up is larger near the inlet and larger near the top of the bed (Tanner & Sukias 1994; Kadlec & Watson 1993). A significant portion of the pore volume can be blocked by accumulated organic matter, leading to increased hydraulic gradients and decreased retention times (Tanner & Sukias 1994). The deposits consist of low-density biosolids together with fine mineral particulates, which can have a very low bulk density.

2.6.2 Biochemical oxygen demand

Settleable organics are rapidly removed in wetland systems under quiescent conditions by deposition and filtration. Attached and suspended microbial growth is responsible for the removal of soluble organic compounds, which are degraded aerobically as well as anaerobically. The oxygen required for aerobic degradation is supplied directly from the atmosphere by diffusion or oxygen leakage from the macrophyte roots into the rhizosphere. The uptake of organic matter by the macrophytes is negligible compared with biological degradation (Watson *et al.* 1989; Cooper *et al.* 1996).

Basic to the understanding of any biological treatment mechanism is an understanding of the microorganisms undertaking the treatment. To continue to reproduce and function properly an organism must have a source of energy, carbon for the synthesis of new cellular material, and inorganic elements (nutrients) such as nitrogen, phosphorus, sulphur, potassium, calcium and magnesium. Some organic nutrient an also be required. Often industrial effluents require the addition of nutrients such as phosphorus or nitrogen for effective biological treatment.

The two main sources of cell carbon are organic chemicals and carbon dioxide. Organisms that use organic carbon for the formation of cell tissue arc called heterotrophs. Organisms that derive cell carbon from carbon dioxide are called autotrophs. Both groups use light or a chemical oxidation reduction reaction as an energy source for cell synthesis. If the major objective of treatment is a decrease in organic content (carbonaceous BOD), the heterotrophic organisms are of primary importance because of their requirement for organic material as a carbon source and their higher metabolic rate (IWA, 2000).

2.6.2.1 Aerobic degradation

The aerobic degradation of soluble organic matter is governed by aerobic heterotrophic bacteria in accordance with the following reaction:

$(CH_2O) + O_2 \rightarrow C0_2 + H_2O$. (5.8)

The autotrophic group of bacteria that degrade organic compounds containing nitrogen under aerobic conditions are called the nitrifying bacteria; the process is called ammonification and will be discussed below, Cooper *et al.* (1996) pointed out that both groups consume organics, but the greater metabolic rate of the heterotrophs means that mainly they are responsible for the decrease in the BOD of the system. An insufficient supply of oxygen to this group greatly decreases the performance of aerobic biological oxidation; however if the oxygen supply is not limited, aerobic degradation is governed by the amount of active organic matter available to the organisms.

Biological degradation can take place within the bulk wastewater; although rates are usually low owing to the small numbers of bacteria present (Polprasert 1998). Nearly all degradation takes place within bacterial Elms present on solid surfaces, including sediments, soils, medium, litter and live submerged plant parts.

2.6.2.2 Anaerobic degradation

Anaerobic degradation is a multi-step process that occurs within constructed wetlands in the absence of dissolved oxygen (Cooper *et al.* 1996). The process can be performed by either facultative or obligate anaerobic heterotrophic bacteria. In the first step the primary end products of fermentation are fatty acids such as acetic acid (Equation 5.9), butyric acid and lactic acid (Equation 5.10), alcohols (5.11) and the gases CO_2 and H_2 (Vymazal 1995):

$C_6H_{12}O_6 \rightarrow 3CH_3COOH + H_2$, (5.9)

$C_6H_{12}O_6 \rightarrow 2CH_3CHOHCOOH$ (lacticacid), (5.10)

$C_6H_{12}O_6 \rightarrow 2CO_2 + 2CH_3CH_2OH$ (ethanol), (5.11)

Acetic acid is the primary acidformed in most flooded soils and sediments. Strictly anaerobic sulphate-reducing (Equation 5.12) and methane-forming (Equations 5.13 and 5.14) bacteria then utilize the end-products of fermentation and, in fact, depend on the complex community of fermentative bacteria to supply substrate for their metabolic activities. Both groups are important in the decomposition of organic matter and carbon cycling in wetlands (Grant or Long 1985; Valiela 1984; Vymazal 1995):

$CH_3COOH + H_2S0_4 \rightarrow 2CO_2 + 2H_2O + H_2S$, (5.12)
$CH_3COOH + 4H_2 \rightarrow 2CH_4 + 2H_2O$, (5.13)
$4H_2 + CH_2 \rightarrow 2CH_4 + 2H_2O$. (5.14)

The acid-forming bacteria are fairly adaptable, but the methane-formers are more sensitives and will operate only in the pHrange 6.5-7.5.

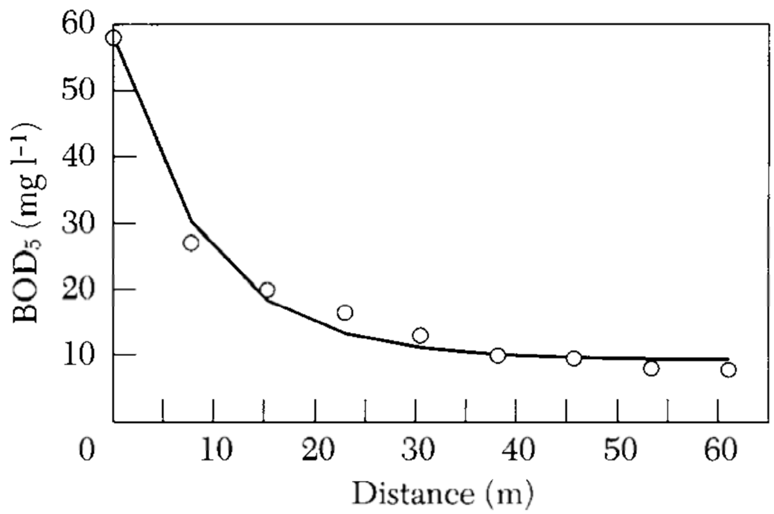

Figure 8: Longitudinal profile BOD in FWS wetland in Arcata, California, USA. Each paint is an 8-month average. The line is a plot of the areal model, with values k = 0.57md-1, C0 =9.5 mgl-1 and R2 = 0.984.

2.6.3 Nitrogen

Nitrogen is a key element in wetland biogeochemical cycles. Nitrogen occurs in a number of different oxidation states in wastewaters and in treatment wetlands, and numerous biological and physicochemical processes can transform nitrogen between these different forms.

The major removal mechanism of organic nitrogen in treatment wetlands is the sequential processes of ammonification, nitrihcatiou and denitritication. Ammonia is oxidized to nitrate by nitrifying bacteria in aerobic zones. Organic N is mineralizcd to ammonia by hydrolysis and bacterial degradation. Nitrates are converted to dinitrogen gas (N_2) and nitrous oxide (N_2O) by denitrifying bacteria in anoxic and anaerobic zones. The oxygen required for nitrification is supplied by diffusion from the atmosphere and leakage from macrophyte roots. Nitrogen is also taken up by plants, incorporated into the biomass and released back as organic nitrogen after decomposition. Other removal mechanisms include volatilization and adsorption. On average, these mechanisms are generally of less importance than nitrification-denitrification, but they can be seasonally important (IWA, 2000).

2.6.3.1 Ammonia volatilization

Ammonia volatilization is a physicochemical process in which NH_4-N is known to be in equilibrium between gaseous and hydroxy forms as indicated below:

$$NH_3(aq.) + H_20 \text{ -> } NH_4^+ + OH^- \qquad (5.20)$$

Reddy & Patrick (1984) pointed out that losses of NH_3 through volatilization from flooded soils and sediments are insignificant if the pH is below 7.5 and very often losses are not serious if the pH is below 8.0. At pH of 9.3 the ratio of ammonia to ammonium ion is 1:1, and the losses via volatilization are significant. Algal photosynthesis in constructed wetlands as well as photosynthesis by free-floating and submerged macrophytcs often creates high pH values.
In a broad literature review, Vymazal (1995) summarized that volatilization rate is controlled by the NH_4^+ concentration in water, temperature, wind velocity, solar radiation, the nature and number of aquatic plants, and the capacity of the system to change the pH in diurnal cycles (the absence of CO_2 increases volatilization).

2.6.3.2 Ammonification (mineralization)

Ammonification (mineralization) is the process in which Org-N is converted into inorganic N, especially NH_4-N. Mineralization rates are fastest in the oxygenated zone and decrease as mineralization switches from aerobic to facultative anaerobic and obligate anaerobic micro-flora. The rate of ammonification in Wetlands is dependent on temperature, pH, the C:N ratio of the residue, available nutrients in the system, and soil conditions such as texture and structure (Reddy & Patrick 1984). The optimum pH range for the ammonification process is between 6.5 and 8.5. In saturated soils, pH is buffered around neutrality, whereas under well-drained conditions the pH value of the soil decreases as a result of nitrate accumulation and the production of H* ions during mineralization (Patrick & Wyatt 1964). Reddy et al. (1979) concluded from published data that the rate of aerobic ammonification doubles with a temperature increase of 10 °C.

2.6.3.3 Nitrification/denitrification

Nitrification

Nitrification is usually defined as the biological oxidation of ammonium to nitrate with nitrite as an intermediate in the reaction sequence. Nitrification is a chemoautotrophic process. The nitrifying bacteria derive energy from the oxidation of ammonia and/or nitrite, and carbon dioxide is used as a carbon source for the synthesis of new cells. These organisms require O_2 during NH_4-N oxidation to nitrite-N and nitrite-N oxidation to nitrate-N (Equations 5.21, 5.22, and 5.23). The oxidation of ammonium to nitrate is a two-step process (Wallace & Nicholas 1969; Hauck 1984):

$NH_4^+ + 1.5O_2 \rightarrow NO_2^- + 2H^+ + H_2O$ (5.21)
$NO_2^- + 0.5O_2 \rightarrow N0_3$ (5.22)
$NH_4^+ + 2O^2 \rightarrow NO_3^- + 2H+ + H_2O$ (5.23)

The first step, the oxidation of ammonium to nitrite, is executed by strictly chemolithotrophic (strictly aerobic) bacteria, which are entirely dependent on the oxidation of ammonia for the generation of energy for growth. In soil, species belonging to the genera Nitrosospira (Nitrosospira briensis), Nitrosovibrio (Nitrosovibrio tenuis), Nitrosolobus (Nitrosolobus multiformis), Nitrosococcus (Nitrosococcus nitrosus) and Nitrosomonas (Nitrosomonas europaea) have been identified. Nitrosomonas europaea is also found in fresh waters (Grant & Long 1981, 1985; Schmidt 1982). The probable reaction sequence for the oxidation of ammonia to nitrite by Nitroso group bacteria is (Hauck 1984);

$NH_3/NH_4^+ \rightarrow NH_2OH \rightarrow NOH \rightarrow$
Ammonia hydroxylamine nitroxyl
$NO_2.NH_2OH \rightarrow NO_2^-$ (5.24)
nitrohydroxylamine nitrite

The postulated intermediate compounds NOH and $NO_2.NH_2OH$ have never been isolated, but their participation in the reaction sequence is consistent with the assumption that two electrons are transferred for each oxidation step between $NH4^-$and NO_2^- (Hauck 1984, and references cited therein).

The second step in the process of nitrification, the oxidation of nitrite to nitrate, is performed by facultative chemolitrotrophic bacteria, which can also use organic compounds in addition to nitrite for the generation of energy for growth. In contrast with the ammonia-oxidizing bacteria, only one species of nitrite-oxidizing bacteria is found in the soil and fresh water, i.e. Nitrobacteria winogradskyi (Grant & Long 1981). Schmidt (1982), how- ever, reported that a genus Nitrospira was found in addition to Nitrabacter in soil and fresh water as well as in marine environments. In addition, in contrast to ammonia-oxidizing bacteria, at least some species of nitrite- oxidizing bacteria can grow mixotrophically on nitrite and a carbon source, or are even able to grow in the absence of oxygen (Bock et al, 1986).

Vymazal (1995) summarizes that nitrification is influenced by temperature, pH, alkalinity inorganic C source, the microbial population and concentrations of NH4-N and dissolved oxygen. The optimum temperature for nitrification in pure cultures ranges from 25 to 35 ^{0}C and in soils from 30 to 40 °C. Lower temperatures (below 15 °C) have a greater effect on nitrification rate than temperatures between 15 and 35 °C. Cooper et al. (1996) pointed out that the minimum temperatures tor growth of Nitrosomonas and Nitrobacter are 5 and 4 °C, respectively.

Nitrifying bacteria are sensitive organisms and are extremely susceptible to a wide range of inhibitors, including high concentrations of arnmoniacal nitrogen. A narrow pH optimum range (7.5£.6) also exists; however acclimatized systems can be operated to nitrify at a much lower pH value. Approximately 4.3 mg of O_2 per mg of ammoniacal nitrogen oxidized to nitrate nitrogen is needed. In the conversion process, a large amount of alkalinity is consumed, ca, 8.64 mg of HCO_3^- per mg of ammoniacal nitrogen oxidized (Cooper et al. 1996).

Denitrification

The first anoxic oxidation process to occur after oxygen depletion is the reduction of nitrate to molecular nitrogen or nitrogen gases. This process is called denitrification. From a biochemical viewpoint, denitrification is a bacterial process in which nitrogen oxides (in ionic and gaseous forms) serve as terminal electron acceptors for respiratory electron transport. Electrons are carried from an electron- donating substrate (usually, but not exclusively, organic compounds) through several carrier systems to a more oxidized N form. The resultant free energy is conserved in ATR after phosphorylation, and is used by the denitrifying organisms to support respiration. Denitrification is illustrated by the following equation (Hauck 1984):

$$6(CH_2O) + 4NO_3^- \rightarrow 6CO_2+2N_2+6H_2O \qquad (5.25)$$

This reaction is irreversible and occurs in the presence of available organic substrate only under anaerobic or anoxic conditions (E_h = +350 to +100 mV), in which nitrogen is used as an electron acceptor in place of oxygen. More and more evidence is being provided from pureculture studies that nitrate reduction can occur in the presence of oxygen. Hence, in waterlogged soils nitrate reduction might also start before the oxygen is depleted (Laanbroek 1990).

Gaseous N production during denitrification can also be depicted as follows (Hauck 1984);

$$4(CH_2O) + 4NO_3 \rightarrow 4HCO_3^-+2N_2O+2H_2O \qquad (5.26)$$

$$5(CH_2O) + 4NO_3 \rightarrow H_2CO_3^-+4HCO^-+ 2N_2+ 2N_2 \qquad (5.27)$$

Denitrifying ability has been demonstrated in 17 genera of bacteria. Most denitrifying bacteria are chemoheterotrophs, obtaining energy solely through chemical reactions and use organic compounds as electron donors and as a source of cellular carbon (Hauck 1984). The genera Bacillus, Miorococcus and Pseudomonas are probably the most important in soils; in the aquatic environment the most important are Pseudomonas, Aeramonas and Vibrio (Grant & Long 1981). Other denitrifiers include members of the genera Achromobacter, Aerobacter, Alcaligenes, Azospirillum, Breoibacterium, Flavnlvacterium, Spirillum and Thiobacillus. A list of genera involved in the denitrification process has been given by Focht & Verstraete (1977). When oxygen is available, these organisms oxidize a carbohydrate substrate to carbon dioxide and water (Reddy & Patrick 1984). Aerobic respiration with oxygen as an electron acceptor or anaerobic respiration using nitrogen for this purpose is accomplished by denitrifier with the same series of electron transport system. This facility to function both as an aerobe and as an anaerobe is of great practical importance because it enables denitrification to proceed at a significant rate soon after the onset of anoxic conditions (a redox potential of ca, 300 mV) Without change in microbial population (Hauck 1984). Because denitrification is performed almost exclusively by facultative anaerobic heterotrophs that substitute oxidized N forms for O_2 as electron acceptors in respiratory processes, and because these processes follow aerobic biochemical routes it can be misleading to refer to denitrification as an anaerobic process; rather, it is one that takes place under anoxic conditions (Hauck 1984).

It is generally agreed that the actual sequence of biochemical changes from nitrate to elemental gaseous nitrogen is (Vymazal 1995)

$$2NO_3 \rightarrow 2\ NO_2 \rightarrow 2NO \rightarrow N_2O \rightarrow N_2. \qquad (5.28)$$

Vymazal (1995) summarizes the environmental factors known to influence denitrification rates, including the absence of O_2, redox potential, soil moisture, temperature, pH, the presence of denitrifies, soil type, organic matter and the presence of overlying water. The quantity of N_2O evolved during denitrification depends on the amount of nitrogen denitrified and the ratio of N_2 to N_2O produced. The ratio is also affected by aeration, pH, temperature and the ratio of nitrate to ammonia in the denitrifying system.

Cooper et al. (1996) pointed out that the presence of dissolved oxygen suppresses the enzyme needed for denitrification and is a critical parameter. The optimum pH range lies between 7 and 8; however, alkalinity produced during denitrification can result in an increase in pH. Denitrification is also strongly temperature-dependent and proceeds only very slowly at temperatures below 5°C. The process of denitrification and its consequences have been reviewed extensively by Payne (1981).

Nitrification and denitrification are known to occur simultaneously in flooded soils in which both aerobic and anaerobic zones exist, such as would occur in a flooded soil or water bottom containing an aerobic surface layer over an anaerobic layer, or in the aerobic rhizosphere microsites in otherwise anaerobic soil. In combination these two reactions, a balanced equation occurring in aerobic and anaerobic layers, can be written as (Reddy or Patrick 1984)

$$24NH_4^+ + 48O_2 \rightarrow 24NO_3^- + 24H_2O + 48H^+ \qquad (5.29)$$

$$24NO_3^- + 5C_6H_{12}O_6 + 24H^+ + 42H_2O \rightarrow 12N_2 + 30CO_2 \qquad (5.30)$$

$$24NH_4^+ + 5C_6H_{12}O_6 + 48O_2 \rightarrow 12N_2 + 30CO_2 + 66H_2O + 24H^+ \qquad (5.31)$$

2.6.3.4 Plant uptake

The potential rate of nutrient uptake by a plant is limited by its net productivity (growth rate) and the concentration of nutrients in the plant tissue. Nutrient storage is similarly dependent on plant tissue nutrient concentrations and also on the ultimate potential for biomass accumulation, that is, the maximum standing crop. Desirable traits of a plant used for nutrient assimilation and storage would therefore include rapid growth, high tissue nutrient content and the capability of attain a high standing crop (biomass per unit area) (Reddy or DeBusk 1987). In the literature there are many reviews on nitrogen concentrations in plant tissue as well as nitrogen standing stocks for plants found in natural stands (Reddy or DeBusk 1987; Vymazal 1995). The uptake capacity of emergent macrophytes, and thus the amount that can be removed if the biomass is harvested, is roughly in the range 1000-2500 kg N ha^{-1} yr^{-1} (Table 5.3). The highly productive water hyacinth (Eichhirnia crassipes) has a higher uptake capacity (up to nearly 6000 kg N ha^{-1} yr^{-1}), whereas the capacity of submerged macrophytes is lower (ca. 700 kg N ha^{-1} yr^{-1}) (Brix 1994; Vymazal 1995). However, only a few data have been reported for plants from constructed wetlands treating wastewaters. In addition, it is important to note that the amounts of nutrients that can be removed by harvesting in secondary treatment systems are generally insignificant in comparison with the loadings into the constructed wetlands with the wastewater (Brix 1994). This is especially true of constructed wetlands with emergent macrophytes. It has been reported that under optimum conditions the amount of nitrogen removed with the biomass does not exceed 10% of the total removed nitrogen (Gersberg et al. 1985; Herskowitz 1986; Vymazal et al. 1999). The removal of nutrients through harvesting might be more important in treatment systems designed for polishing.

If the Wetland is not harvested, the vast majority of the nutrients that have been incorporated into the plant tissue will be returned to the water by decomposition processes. Long- term storage of nutrients in the wetland system results from the undecomposed fraction of the litter produced by the various elements of the biogeochemical cycles as well as the deposition of refractory nutrient-containing compounds (Brix 1996). Seasonality of plant harvesting can also be important in the amount of nutrient mass that can be harvested. For example, it is not possible to harvest common reed during the period of peak standing stock because the plant is easily killed by such activity Phragmites does not translocate storage products to its rhizomes during the growing season; it moves them to the rhizomes just before the end of the growing season. The best time to harvest common reed without damaging plant growth is in the early spring; however, above-ground plant tissue nutrient concentrations are about 30-50% of those during the peak growing season (IWA, 2000).

2.6.3.5 Matrix adsorption

In a reduced state, NH_4-N is stable and can be adsorbed on active sites of an SSF bed matrix or on the sediments of a FWS wetland. However, the ion exchange of NH_4-N on cation-exchange sites of the matrix is not considered to be a long-term sink for NH_4-N removal. Rather sorption of NH_4-N is presumed to be rapidly reversible. As the NH_4-N is lost from the system via nitrification, the exchange equilibrium is expected to redistribute itself. The sorbed NH_4-N in a continuous-flow system will therefore be in equilibrium with the NH_4-N in solution. In the course of seasonal variations in ammonium content in the water, there can be alternate loading and unloading of sorption Sites. Intermittent loading of a system will show rapid removals of NH_4-N by adsorption mechanisms owing to the depletion of NH_4-N on the sorption sites during rest periods (IWA, 2000).

2.6.4 Phosphorus

Constructed and natural Wetlands are capable of absorbing new phosphorus (P) loadings and in appropriate circumstances can provide a low-cost alternative to chemical and biological treatment. Phosphorus interacts strongly with wetland soils and biota, which provide both short-term and sustainable long-term storage of this nutrient.

In SSF wetlands, the sorption capacity of the media can be designed to provide significant P removal (Maehlum et al, 1995). This storage eventually becomes saturated, necessitating the replacement of the medium and the re- estahlishment of the wetland.

In FWS Wetlands, soil sorption can provide initial removal, but this partly reversible storage eventually becomes saturated. For some antecedent soil conditions, there can even be an initial release of P. A new source of P acts to fertilize the wetland, and some P is used in the establishment of a new or larger standing crop of vegetation.

The sustainable removal processes involve the accretion of new wetland sediments, uptake by small organisms, including bacteria, algae and duckweed, forms a rapid-action, partly reversible removal mechanism. Cycling through growth, death and decomposition returns most of the microbiotic uptake via leaching, but an important residual contributes to long-term accretion in newly formed sediments and soils. Macrophytes, such as cattails and bulrushes, follow a similar cycle but on a slower time scale of months or years. The cletrital residual from the macrophyte cycle also contributes to the long-term storage in accreted solids. Direct settling and trapping of particulate P can contribute to the accretion process. There can also be biological enhancement of mineralogical processes, such as iron and aluminium uptake and subsequent P binding in detritus and the algae-driven precipitation of P with calcium (IWA, 2000).

2.6.5 Pathogens

Domestic Wastewaters contain human pathogens that can survive pretreatment and enter treatment wetlands. These include bacteria, viruses, protozoans and helminths. The commonly used regulatory measure is for faecal coliforms, but faecal streptococci, Salmonella, Yersinia, Pseudomonas and Clostridium have all been studied in treatment wetlands (Herskowitz 1986). These human enteric organisms are typically decreased in numbers in passage through SF wetlands. Viruses are also attenuated in wetlands (Gersberg et al. 1989). Less is known about protozoans and their cysts, such is Giardia and Cryptosporidium, but these also are decreased in wetlands (Rivera et al. 1994). Helminths, including eggs of the nematode Ascaris, and various species of amoebae, were also decreased in soil-based systems (Rivera et el. 1994).

The ecology of microorganisms in a constructed wetland, as in any biological wastewater treatment system, is extremely complex. The important organisms from a public health point of view are the pathogenic bacteria and viruses. Protozoan pathogens and helminth worms are also of particular importance in tropical and subtropical countries (Rivera et al 1995). In the aerobic environment of a VF wetland and the colder partly aerobic environment of an HF SSF system, they have minimal growth. Pathogens are removed during the passage of Wastewater through the system mainly by sedimentation, filtration and adsorption on sediments. Once these organisms are entrapped within the system their numbers decrease rapidly mainly by the processes of natural die- off and predation.

Wetlands are known to offer a suitable combination of physical, chemical and biological factors for the removal of pathogenic organisms. Physical factors include mechanical filtration, exposure to ultraviolet, and sedimentation. Chemical factors include oxidation, exposure to biocides excreted by some plants, and absorption by organic matter. Biological removal mechanisms include antibiosis (Seidel et al l978), predation by nematodes and protists, attack by lytic bacteria and viruses, and natural die-off (Gersberg et al, 1989).

Lower temperatures are known to affect the survival of sewage bacteria adversely. However, higher temperatures favor not only the pathgens hut also their predators. Physical processes, such as sorption or settling, are not particularly temperature-sensitive. Annual irradiation patterns mimic the annual temperature cycle, and hence ultraviolet-induced mortality should he higher at higher temperatures.

Some pathogens are associated with warm- blooded animals other than humans, most especially faecal coliforms, streptococci and Salmonella. It is therefore possible for birds and mammals to contribute to the occurrence of these organisms in the wetland environment. A negative effect of treatment has been observed for treatment wetlands with high bird populations (PBSI 1989). Non-zero background concentrations of faecal coliforms are typically present in natural, unimpacted Wetlands.

Removal of coliform bacteria in SSF wetlands has been described by several authors, including Gersberg et al (1989a, b), Bavor et al. (1989) and Williams et al. (1985), Gersberg et al. (1989a) demonstrated 97% (1.52 log) removal in a gravel-filled artificial wetland, planted withScirpus in Santee, California, USA, with a theoretical retention time of 1.5 d. Bavor et al. (1989) looked at, among other things, the removal of coliforms in long, gravel-filled, trenches in Richmond, Australia, comparing vegetated (Typha) and unvegetated systems. By sampling along the length of the channels, they produced data that fitted a first-order reaction equation and calculated removal rate constants for the different systems. With their model it is possible to predict the requirement of 2 d retention to achieve 90% removal in a gravel trench at 20 °C, whereas the

Typha-planted trench required more than 3 d to achieve 90% removal at 20 °C. Williams et al. (1985) also sampled for coliform bacteria along the length of gravel filled wetland systems. Their tertiary treatment beds achieved 99% removal of faecal coliforms with a retention time of 1 d. The secondary bed required approx. 2.5 d retention time to achieve 90% removal of faecal coliforms.

2.7 Hydraulic Sizing

It is necessary to select the length (L) and width (W) of constructed wetland basin, as given by the aspect ratio, L/W. The presumption that high aspect ratios would favor a more efficient (close to plug flow) mode has proved to be untrue in many tracer tests of constructed wetlands. Consequently, any aspect ratio with good inlet distribution can be considered (IWA, 2000). Aspect ratio is also a principal determinant of the hydraulic profile in the wetland. Flow through vegetation, or bed medium, creates a decreasing elevation of water surface from inlet to outlet. The decrease in water-surface elevation from inlet to outlet is the head loss for the system. The hydraulic profile must be contained properly in the wetland (IWA, 2000).

For subsurface horizontal flow wetlands, the requirements for stable and controllable water flow and for proper vegetation conditions serve to restrict the geometry of the bed and the size of the medium. The requirements are:

- Expected flows must pass through the bed without overland flow or flooding
- Expected flows must pass through the bed without stranding the plants above water; i.e. there must not be protracted excessive headspace
- Operation should remain acceptable in the likely event of changing hydraulic conductivity. As the bed clogs with roots and sediments, it should not flood
- The bed should be drainable
- The bed should be floodable
- Water levels within the system should be fully controllable through the use of inlet and outlet structures
- The configuration must fit the site, in terms of project boundaries and in terms of hydraulic profiles

Such constraints must be met for all expected operation conditions, including initial and clogged conductivity, and the range of expected operating flows, including daily maximum and minimum values (IWA, 2000).

Bed depth is usually a narrow range and is set by conditions other than hydraulics. There is no theoretical need for a slope to the top of the bed. If control of water level is to include the ability to inundate the bed totally for vegetation management, then a top slope is detrimental. The bed depth is usually selected to be in the range of 30-60 cm, based on assumptions on plant rooting depth and its effect on treatment potential. Such depth 'criteria' remain speculative. However, ice

formation can use some of the water depth, and there needs to be some room for sediment accretion in the bottom of hte bed. The upper half of the range, 45-60 cm, therefore seems to be the best choice (IWA, 2000).
The determinants of the hydraulic profile of a horizontal SSF wetland are (IWA, 2000):

- flow rate
- outlet weir or standpipe setting
- aspect ratio (more generally, system planar geometry)
- bottom slope (more generally, vertical morphology)
- media resistance (hydraulic conductivity)

The bottom slope should be set to provide for complete bed drainage. Normally a few centimeters of elevation differential allows for this requirement. Bottom slope should not be considered as the design driving force for water movement. The reason is that designs based on bed slope are excessively sensitive to changing conditions of flow and hydraulic conductivity; dryout or flooding is virtually certain to occur with such designs (IWA, 2000). For subsurface vertical flow wetlands, because the flow is vertical, the aspect ratio is no longer a determinant of hydraulics, but the conductivity of the medium becomes more important, especially during the drainage portion of the cycle. The consideration of flow and saturation are complicated, and they have not been reduced to design guidelines yet (IWA, 2000).

2.8 Cost

Constructed wetlands have been developed for the treatment of wastewater and have been widely reported as being low in construction and operating costs (US Environmental Protection Agency 1988; Reed et al. 1995). Estimating the initial capital cost of the project is a routine exercise in most respects.

When reviewing costs, the engineer must take into account other factors such as the number of cells. It is obvious that large systems will cost less per unit flow than small systems. Perhaps the most difficult cost element to access is the form of pretreatment. If the collection system is a small-diameter system with interceptor tanks, then pretreatment will produce an influent BOD in the range 120-140 mgl^{-1} with anaerobic properties. Influent TSS us averaging 30 mgl^{-1} in some of these small diameter systems. However, overall system costs must be considered as well as the treatment cost.

Wetland costs can be broken down into the following components (IWA, 2000):

- excavation
- liner
- plants
- filter media (gravel, sand)

- distribution and control structures
- fencing
- other

Chapter 3: Methodology

For the study, the methodologies that were used is described in sections below:

3.1 Experimental Setup

The experimental set up for the study are described in following sub-sections under this section.

3.1.1 Basins/reservoirs and pipe network

For the study, two small size constructed wetlands were prepared using local materials and plants. To prepare the constructed wetlands, two basins were built at the university campus, on the open spaces in the northern side of the Civil Engineering building of the campus. Among the two basins, one basin had dimensions of 9'x 5' x 4.5' (L X W X D) and another one of 6' X 5' X 4.5'. Both the basins were prepared with brick and mortar. The inner surface of the basins was made impervious by providing lining materials (cement mortar) to prevent any water loss from the tank through seepage at the bottom or from the sides. The bottom of the basin has a small slope (1" throughout the distance from inlet to outlet) towards the outlet to ensure the hydraulic gradient.

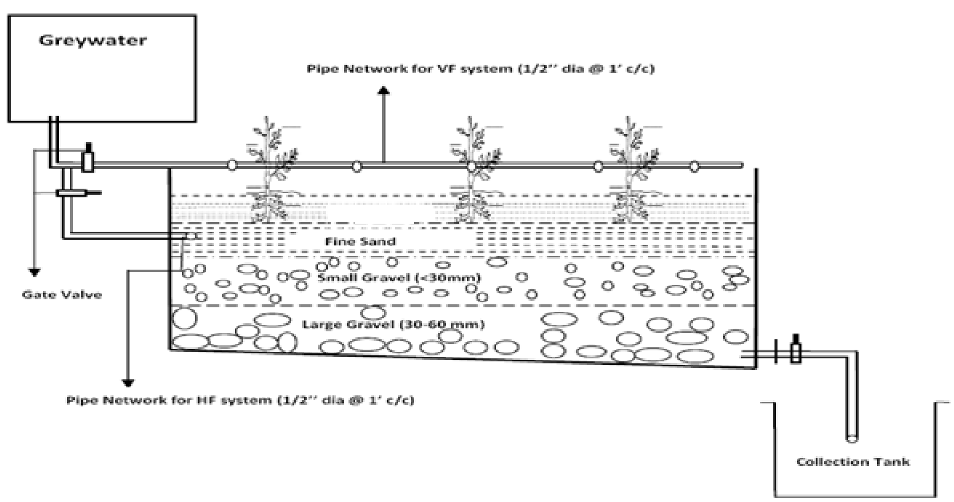

Figure 9: Schematic diagram of the experimental set up for the study.

To store grey water before applying into the constructed wetlands, a 1,000 L capacity reservoir was placed near the inlet point of the basins. This plastic reservoir was elevated above the level of inlet point so that water from the reservoir can flow into the basins through gravity flow. The outlet pipe from the grey water reservoir was connected to both the basins. The pipe networks were designed in a way so that both the basins can be used for both horizontal flow and vertical flow. A "collection tank" followed the wetland basins to collect the treated water. The collection tanks was placed underground so that the treated water from the constructed wetlands can be stored in the tank using gravity flow. The typical cross section is shown in Figure 9.

3.1.2 Filter media

As the study was designed to find the treatment efficiency of filter materials, with and without vegetation, the filter media for the basins were selected and placed carefully. The total substrata depth in this experiment was designed to be 90 cm for both the basins. At the bottom of the basin, a 30 cm layer of "large gravel" was placed. The size for the large gravels varied from 30 to 60 mm in diameter. On top of the "large gravel" layer, a 30 cm layer of "small gravel" was placed, where the size of gravels were less than 30 mm. On top of these two layers, a 30 cm layer of fine sand at the top was placed. Since it is important to consider that the layers of the filter media is selected in a way that water does not get stuck and prevents oxygen to enter through the top layer of the media and reach the root zone of the plants, the find sand was free from any loamy texture to increase the permeability through the sand layer. Before placing the filter materials into the basins, the gravels were washed properly and dried in the sun. The find sand used in the basins was also cleaned and dried in the sun before putting on top of gravel layers.

3.1.3 Vegetation

Subsurface flow constructed wetlands are planted with emergent vegetation. Selection of species is important since the plants in wetland systems provide the basis for water purification functions, as well as conduct hydrological buffering. The plants that are most often used in constructed wetlands are persistent emergent plants, such as bulrushes, spikerush, other sedges, rushes, common reed, and cattails. Not all wetland species are suitable for waste water treatment since plants for treatment wetlands must be able to tolerate the combination of continuous flooding and exposure to wastewater containing relatively high and often variable concentrations of pollutants.

As this experiment was designed to experiment small-scale treatment system for treating grey water at household level, the vegetation plants were chosen considering the landscape and maintenance requirements. Therefore, the locally available species like Solenostemon scutellarioides (family: Lamiaceae), Dracaena sanderiana (family: Ruscaceae), Homalomena rubescens (Roxb.) Kunth (family: Araceae), Calendula officinlis (family: Asteraceae) were used considering its popularity in gardening and beautification. All these plants are locally available in Bangladesh and widely used in homestead gardening. Therefore, though these are not often used in large scale constructed wetlands for wastewater treatment.

Figure 10: Two subsurface flow constructed wetland basins at BUET.

Figure 11: Collection system to collect treated grey water from wetland basins.

Figure 12: Constructed wetlands with pipe networks from the grey water reservoir to collection tank.

Figure 13: Vegetation in subsurface flow constructed wetland.

3.2 Operational Procedures

3.2.1 Collection of grey water

Grey water used in this study was transported from a nearby student's hostel. It was the water that comes out of the hostel's kitchen after washing dishes, vegetables, meat etc. To remove the garbage and large materials, the raw water was screened before feeding into the wetland basins. Approximately 1,000 L of grey water was carried from the hostel by water tanks on the day of testing and was stored in reservoir of the set up prior to feeding the wetlands. Since the amount of water fed into the wetlands was kept fixed at 1,000 L, additional amount of water to make up for any shortage of grey water was added using fresh water.

3.2.2 Application of grey water into the wetlands

After the storage tank was filled, the valves for either horizontal flow (HF) or vertical flow (VF) constructed wetland was opened so that water could flow into the basin. It is to be noted that both the basins had the option of either horizontal flow or vertical flow by making changes in the pipe network. When the pipe network for HF system was open, the VF pipe network was kept closed and vice versa. Initially clear water from overhead tank was flown into the basin for two weeks prior to start testing for horizontal or vertical system with grey water to check the functionality of the whole system and also to wash out the organic contents/debris in the filter media.

First the testing was done in the wetlands for "with vegetation" condition, where the plants were planted one month prior to the first test day to allow the plants to grow in the basins. In Horizontal Subsurface Flow System, the wastewater was fed at the inlet through pipes and flown through the porous medium under the surface of the bed in a more or less horizontal path until it reaches the outlet. While for the vertical flow sub-surface system, water was fed in batches using a network of pipes. The new batch was fed only after all the water had percolated and the bed was free of water to enable diffusion of oxygen from the air into the bed. The detailed schedule for the testing is presented in Table 3. The test was first done for basin 1 for both horizontal and vertical systems "with vegetation." Each system was tested for six days in two consecutive weeks, hence basin was first tested for horizontal flow in week one and two, and then for vertical flow in week three and four. After that the same process was followed for basin 2 for the next four weeks.

Once testing for "with vegetation" condition was completed for both the basins for both horizontal and vertical flow systems, the plants were uprooted from the basins and the top surface of the fine sand layer was removed. For next four weeks, no testing was done and a new layer of fine sand was placed on top of gravel layers for testing the performance under "without vegetation" condition. The whole field work for the study was carried out in 27 weeks.

Table 3: Operational procedure for both horizontal and vertical subsurface constructed wetlands under different scenario for both basins.

Timeline	Activity	Type of Flow	Basin Type	Vegetation	Flow Rate (L/min)
Week 1-2	Feeding clean water to check functionality of the wetlands	Vertical	Both 1 and 2	No vegetation	18-24
Week 3-6	Plantation of selected vegetation	N/A	Both 1 and 2	Allowing plants to grow before operation	N/A
Week 7-8	Feeding grey water (1,000 L)	Horizontal	Basin 1	With vegetation	10-12
Week 9-10	Feeding grey water (1,000 L)	Vertical	Basin 1	With vegetation	18-24
Week 11-12	Feeding grey water (1,000 L)	Horizontal	Basin 2	With vegetation	10-12
Week 13-14	Feeding grey water (1,000 L)	Vertical	Basin 2	With vegetation	18-24
Week 15	Uprooting all plants, removal of sand	N/A	Both 1 and 2	N/A	N/A
Week 16-18	Placing new sand layer on top of gravel layers and allowing it to settle	N/A	Both 1 and 2	N/A	N/A
Week 19	Feeding clean water to check functionality of the wetlands	Vertical Flow	Both 1 and 2	No vegetation	18-24
Week 20-21	Feeding grey water (1,000 L)	Horizontal	Basin 1	Without vegetation	10-12
Week 22-23	Feeding grey water (1,000 L)	Vertical	Basin 1	Without vegetation	18-24
Week 24-25	Feeding grey water (1,000 L)	Horizontal	Basin 2	Without vegetation	10-12
Week 26-27	Feeding grey water (1,000 L)	Vertical	Basin 2	Without vegetation	18-24

3.3 Water Sampling

Samples were collected from two locations; one from the inlet point of the basin where water was coming from the grey water reservoir, and another sample was collected from the "collection tank" where treated water passed through the wetland was stored. At the collection tank, the treated water was collected from the pipe, before falling into the tank. The samples were collected during the test days to test performance of wetlands to treat grey water. All samples were collected in the sampling bottles and were taken to the laboratory for testing immediately.

3.4 Water Quality Testing

Samples collected from the constructed wetlands, both before and after treatment were analyzed for the concentrations of Total Suspended Solids (SS), Biochemical Oxygen Demand (BOD_5), Chemical Oxygen Demand (COD), Total Ammonia, Orthophosphate and Fecal Coliform (FC).

There were total 96 samples tested for the above six parameters in the laboratory. Among these 96 samples, 48 were grey water samples and 48 were treated water samples. Table 4 illustrates the sampling frequency during the course of test.

Table 4: Sampling frequency for different types of flows in two constructed wetland basins

Timeline (week and no. of days)	Basin No.	Type of Flow	No. of grey water samples	No. of treated water samples
Week 7-8 (6 days)	1	Horizontal (with vegetation)	4	4
Week 9-10 (6 days)	1	Vertical (with vegetation)	4	4
Week 11-12 (6 days)	2	Horizontal (with vegetation)	4	4
Week 13-14 (6 days)	2	Vertical (with vegetation)	4	4
Week 20-21 (6 days)	1	Horizontal (without vegetation)	4	4
Week 22-23 (4 days)	1	Vertical (without vegetation)	4	4
Week 24-25 (4 days)	2	Horizontal (without vegetation)	4	4
Week 26-27 (4 days)	2	Vertical (without vegetation)	4	4

Chapter 4: Results and Discussion

The tests performed during the study shows efficiency of subsurface flow constructed wetlands under different conditions. As the wetlands were tested under both horizontal and vertical flow conditions, the results would help to find efficiency level of both systems to remove pollutants from grey water. Moreover, the tests were performed for different vegetation conditions; one "with vegetation" and another "without vegetation." Hence, the results would also help to identify impact of vegetation in removal of pollutants in wetlands.

In the study, removal of Total Suspended Solids (SS), Biochemical Oxygen Demand (BOD_5), Chemical Oxygen Demand (COD), Total Ammonia, Orthophosphate and Fecal Coliform (FC) by subsurface flow constructed wetlands, both horizontal flow and vertical flow systems were tested. The removal efficiency has been presented in percentage in the following sections for different tested parameters. Two basins were used having different dimensions to test the effect of size on removal performance. A range of concentrations of pollutants and solids in the grey water used in this study is shown in Table 5. The hydraulic retention times for the two basins were found different, and also varied for different types of flows. In basin 1 (length 9'), the retention time during horizontal flow was 3.5 hours, while for vertical flow it was found 2.5 hours. In basin 2 (length 6'), retention time during horizontal flow was 3 hours, while for vertical flow it was found 2.5 hours.

Table 5: Quality of grey water used in the study as influent for the constructed wetlands.

Sl. No	Parameter	Unit	Concentration in untreated grey water (lowest value - highest value)
1	Total Suspended Solids (TSS)	mg/l	57-183
2	BOD (5)	mg/l	2 - 190
3	COD	mg/l	4-637
4	Total Ammonia	mg/l	0.09-0.776
5	Orthophosphate	mg/l	0.386-1.03
6	Fecal Coliform	CFU/100 ml	120-TNTC

The following sections discuss the removal efficiency of subsurface flow constructed wetlands under different condition. The acronyms used for different operating conditions in this chapter are:

HFB1:	Horizontal subsurface flow in basin 1 **with vegetation**
HFB2:	Horizontal subsurface flow in basin 2 **with vegetation**
VFB1:	Vertical subsurface flow in basin 1 **with vegetation**
VFB2:	Vertical subsurface flow in basin 2 **with vegetation**
HFB1W:	Horizontal subsurface flow in basin 1 **without vegetation**
HFB2W:	Horizontal subsurface flow in basin 2 **without vegetation**
VFB1W:	Vertical subsurface flow in basin 1 **without vegetation**
VfB2W:	Vertical subsurface flow in basin 2 **without vegetation**

4.1 Removal of Total Suspended Solids (TSS)

Removal of Total Suspended Solids (TSS) was found very high in all types of constructed wetlands in the study. Concentrations of suspended solids in influent range from 57 mg/l to 183 mg/l. The removal efficiency of suspended solids in basin-1 is shown in Figure 14 as percentage value for all four types of scenario in the basin. Table 6 shows the values of concentrations in grey water before (inflow) and after (outflow) treatment in all four combinations in basin 1.

Table 6: Concentration of Total Suspended Solids (TSS) in inflow and outflow samples for different conditions in basin 1.

Sample No	Concentration in HF with vegetation (mg/L)		Concentration in VF with vegetation (mg/L)		Concentration in HF without vegetation (mg/L)		Concentration in VF without vegetation (mg/L)	
	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
1	57	11	81	12	153	29	161	14
2	160	26	72	9	68	7	179	20
3	83	12	93	6	92	11	156	11
4	105	31	105	17	178	30	142	11
5	154	35	149	28	183	29	91	7
6	92	16	101	5	83	12	110	10

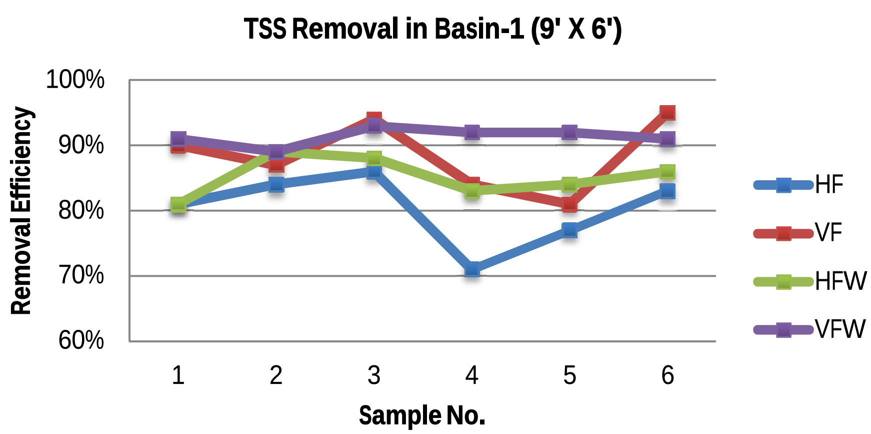

Figure 14: Removal of Total Suspended Solids (TSS) in constructed wetland basin-1 under four scenarios.

Table 7 shows the values of concentrations in grey water before (inflow) and after (outflow) treatment in all four combinations in basin 2. The removal efficiency for suspended solids in basin-2 is shown in Figure 15 as percentage value for all four scenarios in the basin.

Table 7: Concentration of Total Suspended Solids (TSS) in inflow and outflow samples for different conditions in basin 2.

Sample No	Concentration in HF with vegetation (mg/L)		Concentration in VF with vegetation (mg/L)		Concentration in HF without vegetation (mg/L)		Concentration in VF without vegetation (mg/L)	
	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
1	152	50	89	17	80	22	156	25
2	133	25	145	25	93	30	152	21
3	174	49	76	9	69	21	106	20
4	105	28	103	21	101	22	93	11
5	98	31	107	17	87	22	171	26
6	173	42	183	28	62	17	151	27

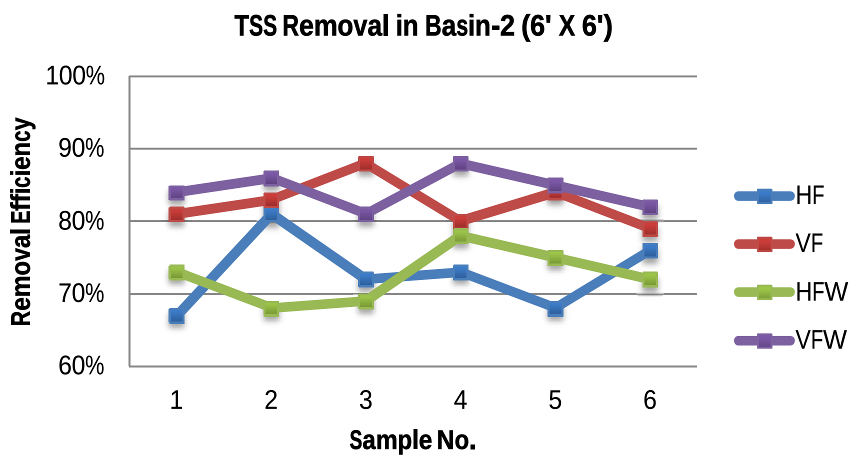

Figure 15: Removal of Total Suspended Solids (TSS) in constructed wetland basin-2 under four scenarios.

Comparison of all eight scenarios tested in the study is shown in Figure 16 and Figure 17 that indicate that vertical flow constructed wetlands are more efficient in removing suspended solids from water. It was also experienced in the study that removal rate is higher without vegetation, which is likely because of the absence of root zones that help water to travel easily into the deeper layers of the filter media. Basin 1 also shows higher efficiency than basin 2 that indicates that higher travel time and length increase removal efficiency for suspended solids.

The high TSS removal rate in sub-surface flow constructed wetlands was also found in many previous research works which is again proved under this study. Jan Vymazal (2010), in his study, found removal efficiency in horizontal and vertical flow wetlands as for TSS as high as 75% and 89% respectively, which is also close to the range of removal efficiency found in this study.

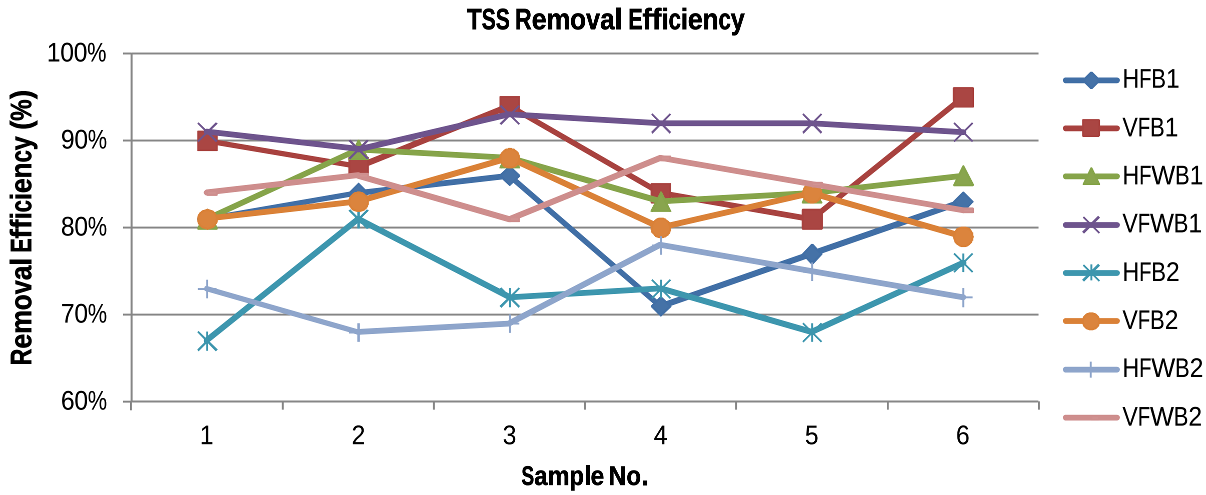

Figure 16: Suspended solid removal efficiency of constructed wetlands in two different basins with and without vegetation under four scenarios.

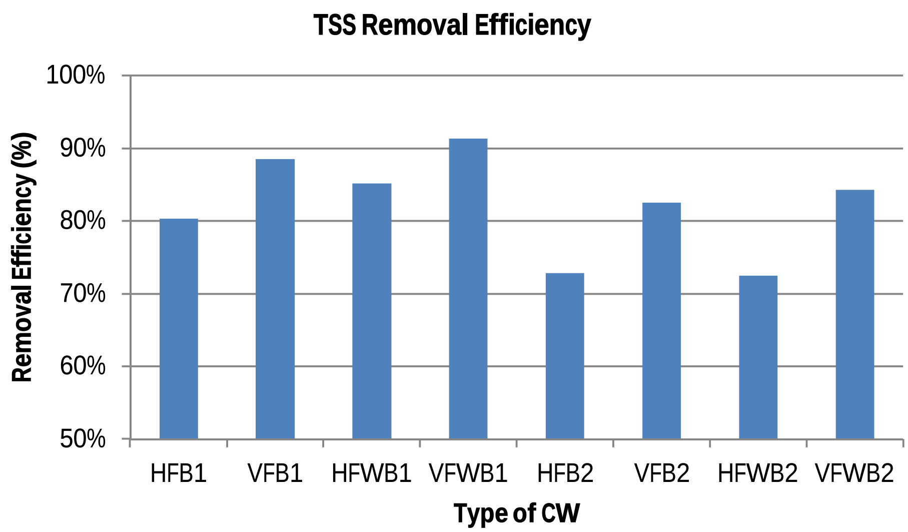

Figure 17: Comparison of suspended solid removal efficiency of different types of constructed wetlands in two different basins with and without vegetation.

4.2 Biochemical Oxygen Demand (BOD_5) Removal

Removal of Biochemical Oxygen Demand (BOD_5) from grey water was found very high in all types of constructed wetlands in the study. Concentrations of BOD_5 found in influent range from 2 mg/l to 190 mg/l. The removal efficiency of BOD_5 in basin-1 is shown in Figure 18 as percentage value for all four types of scenario in the basin. Table 8 shows the values of concentrations in grey water before (inflow) and after (outflow) treatment in all four combinations in basin 1.

Table 8: Concentration of BOD5 in inflow and outflow samples for different conditions in basin 1.

Sample No	Concentration in HF with vegetation (mg/L)		Concentration in VF with vegetation (mg/L)		Concentration in HF without vegetation (mg/L)		Concentration in VF without vegetation (mg/L)	
	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
1	63	2	89	0	156	6	19	1
2	53	1	145	1	152	3	65	1
3	15	0	76	1	106	4	98	4
4	93	1	4	0	93	2	43	1
5	168	7	61	1	171	5	104	3
6	59	1	183	4	151	11	23	2

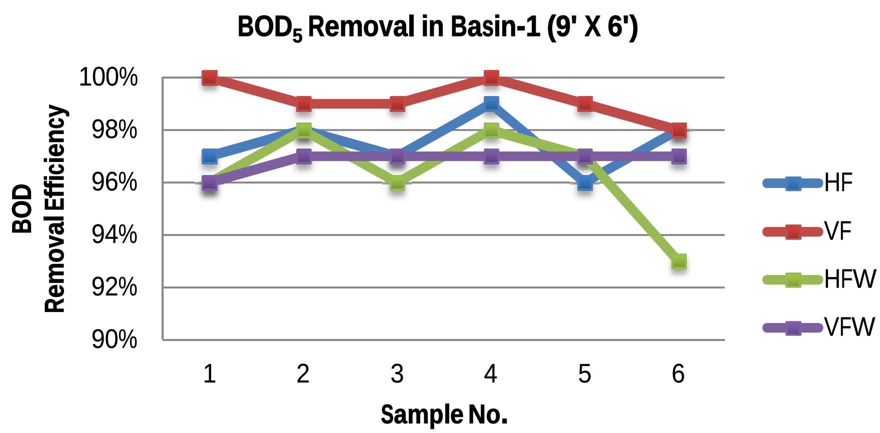

Figure 18: Removal of BOD5 in constructed wetland basin-1 under four scenarios.

Table 9 shows the values of concentrations in grey water before (inflow) and after (outflow) treatment in all four combinations in basin 2. The removal efficiency for BOD_5 in basin-2 is shown in Figure 19 as percentage value for all four scenarios in the basin.

Table 9: Concentration of BOD5 in inflow and outflow samples for different conditions in basin 2.

Sample No	Concentration in HF with vegetation (mg/L)		Concentration in VF with vegetation (mg/L)		Concentration in HF without vegetation (mg/L)		Concentration in VF without vegetation (mg/L)	
	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
1	31	2	151	6	125	9	96	6
2	96	6	63	1	93	6	190	10
3	104	7	9	0	72	4	24	1
4	12	1	172	5	26	2	164	11
5	2	0	190	10	27	2	187	7
6	19	1	69	1	39	3	143	7

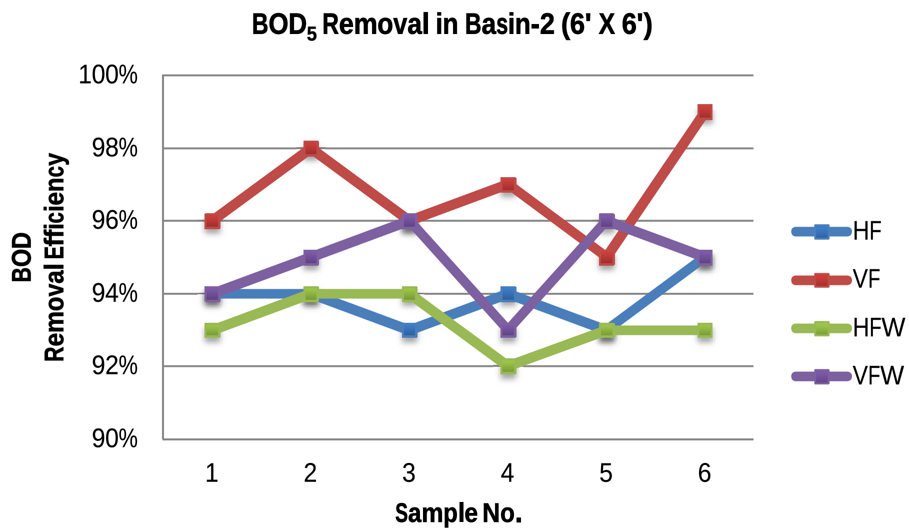

Figure 19: Removal of BOD5 in constructed wetland basin-2 under four scenarios.

Comparison of all eight scenarios tested in the study is shown in Figure 20 and Figure 21 that indicate that vertical flow constructed wetlands are more efficient in removing BOD$_5$ from water. It was also experienced in the study that removal rate is higher with vegetation, which is because of the activities in the root zones that help removing organic contents as availability of oxygen is higher due to open zones created around the roots of the plants. Basin 1 also shows higher efficiency than basin 2. This might be due to greater surface area in basin.

The high BOD$_5$ removal rate in sub-surface flow constructed wetlands was also found in many previous research works, which is also proved under this study. Jan Vymazal (2010), in his study, found removal efficiency in horizontal and vertical flow wetlands as for BOD$_5$ as high as 75% and 90% respectively, which is also close to the range of removal efficiency found in this study.

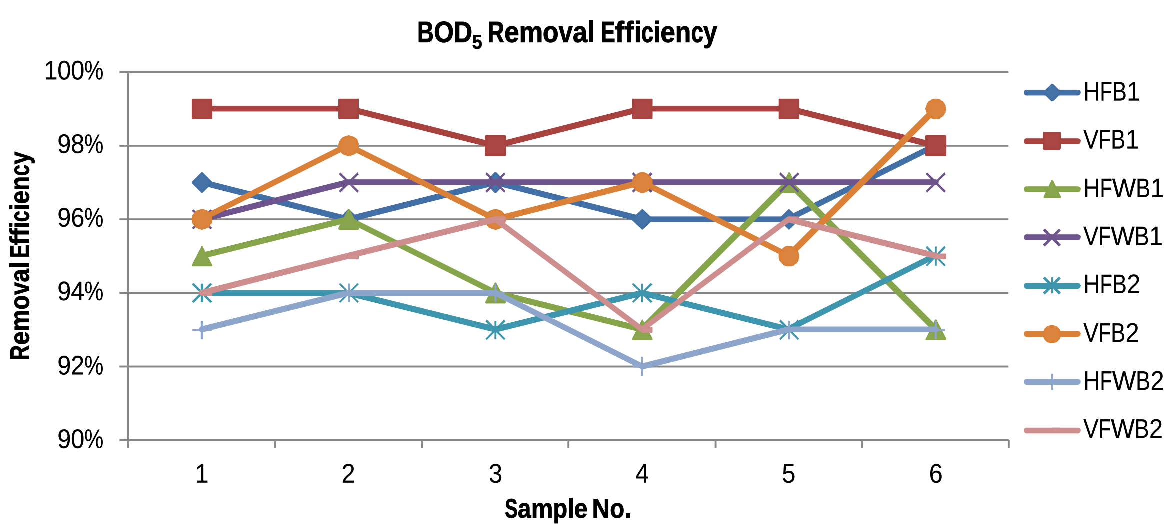

Figure 20: BOD5 removal efficiency of constructed wetlands in two different basins with and without vegetation.

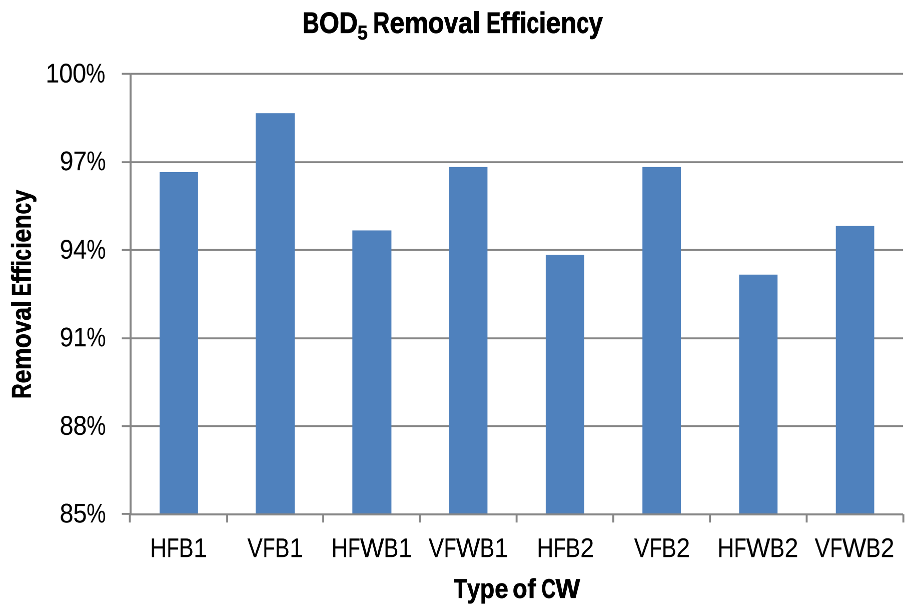

Figure 21: Comparison of BOD5 removal efficiency of different types of constructed wetlands in two different basins with and without vegetation.

4.3 Chemical Oxygen Demand (COD) Removal

Removal of Chemical Oxygen Demand (COD) from grey water was found very high in all types of constructed wetlands in the study. Concentrations of COD found in influent range from 4 mg/l to 637 mg/l. The removal efficiency of COD in basin-1 is shown in Figure 22 as percentage value for all four types of scenario in the basin. Table 10 shows the values of concentrations in grey water before (inflow) and after (outflow) treatment in all four combinations in basin 1.

Table 10: Concentration of COD in inflow and outflow samples for different conditions in basin 1.

Sample No	Concentration in HF with vegetation (mg/L)		Concentration in VF with vegetation (mg/L)		Concentration in HF without vegetation (mg/L)		Concentration in VF without vegetation (mg/L)	
	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
1	156	9	362	14	311	22	162	10
2	201	12	486	10	326	20	306	15
3	97	7	199	8	299	18	265	11
4	365	22	68	2	208	17	211	15
5	461	32	146	7	461	32	384	15
6	135	7	567	6	418	29	246	12

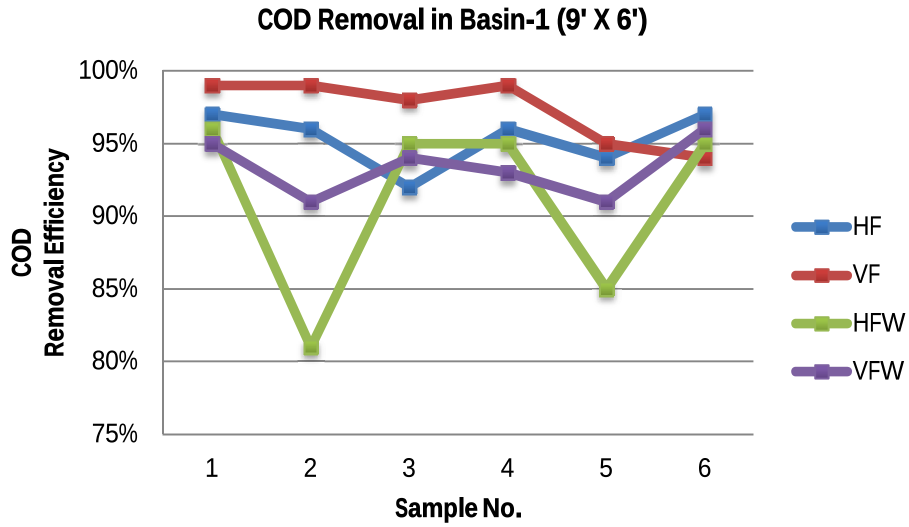

Figure 22: Removal of COD in constructed wetland basin-1 under four scenarios.

Table 11 shows the values of concentrations in grey water before (inflow) and after (outflow) treatment in all four combinations in basin 2. The removal efficiency for COD in basin-2 is shown in Figure 23 as percentage value for all four scenarios in the basin.

Table 11: Concentration of COD in inflow and outflow samples for different conditions in basin 2.

Sample No	Concentration in HF with vegetation (mg/L)		Concentration in VF with vegetation (mg/L)		Concentration in HF without vegetation (mg/L)		Concentration in VF without vegetation (mg/L)	
	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
1	131	4	396	4	410	16	317	16
2	293	12	168	2	360	68	562	51
3	306	24	82	2	416	21	127	8
4	157	6	450	5	214	11	367	26
5	69	4	349	17	193	29	637	57
6	201	6	246	15	206	10	397	16

Comparison of all eight scenarios tested in the study is shown in Figure 24 and Figure 25 that indicate that vertical flow constructed wetlands are more efficient in removing COD from water. It was also experienced in the study that removal rate is higher with vegetation, which is because of the activities in the root zones that help removing organic contents as availability of oxygen is higher due to open zones created around the roots of the plants. Basin 1 also shows higher efficiency than basin 2. This might be due to greater surface area in basin.

The high COD removal rate in sub-surface flow constructed wetlands was also found in many previous research works, which is also proved under this study. A similar study conducted in Nepal by Bista and Khatiwada revealed that removal efficiency in subsurface flow constructed wetlands for COD was above 85%, which is also close to the range of removal efficiency found in this study.

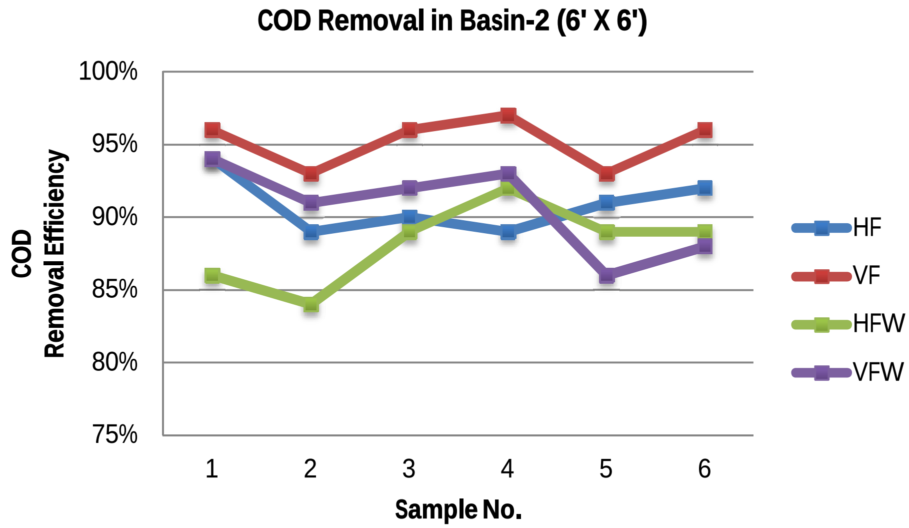

Figure 23: Removal of COD in constructed wetland basin-2 under four scenarios.

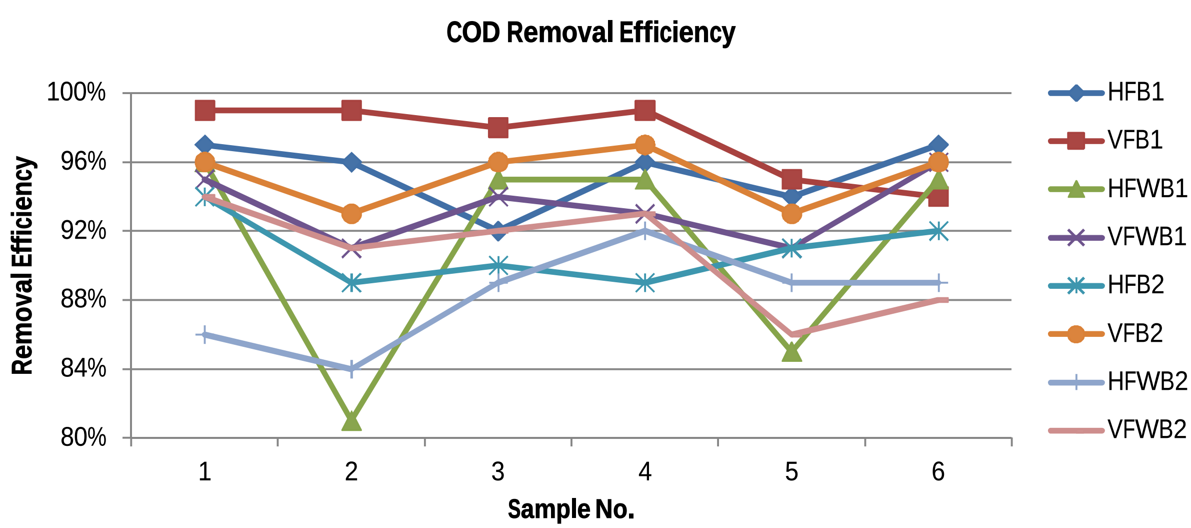

Figure 24: COD removal efficiency of constructed wetlands in two different basins with and without vegetation.

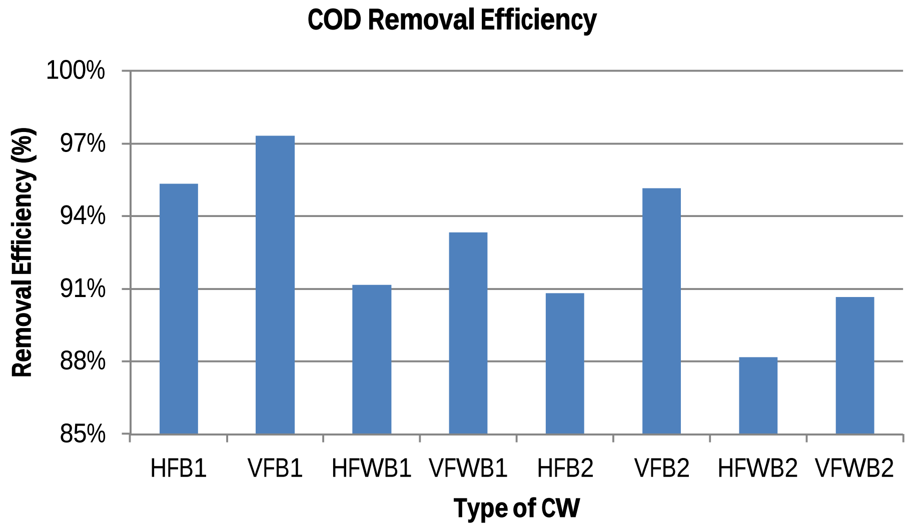

Figure 25: Comparison of COD removal efficiency of different types of constructed wetlands in two different basins with and without vegetation.

4.4 Removal of Total Ammonia

Removal of Ammonia (Total) from grey water was also found high in all types of constructed wetlands in the study. Concentrations of Ammonia (Total) found in influent range from 0.09 mg/l to 0.776 mg/l. The removal efficiency of Ammonia (Total) in basin-1 is shown in Figure 26 as percentage value for all four scenarios in the basin. Table 12 shows the values of concentrations in grey water before (inflow) and after (outflow) treatment in all four combinations in basin 1.

Table 12: Concentration of Ammonia (Total) in inflow and outflow samples for different conditions in basin 1.

Sample No	Concentration in HF with vegetation (mg/L)		Concentration in VF with vegetation (mg/L)		Concentration in HF without vegetation (mg/L)		Concentration in VF without vegetation (mg/L)	
	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
1	0.15	0.01	0.58	0.03	0.78	0.19	0.61	0.09
2	0.63	0.03	0.19	0.01	0.65	0.14	0.59	0.10
3	0.42	0.03	0.17	0.01	0.09	0.02	0.54	0.09
4	0.26	0.02	0.68	0.04	0.43	0.08	0.26	0.05
5	0.31	0.02	0.44	0.02	0.30	0.06	0.42	0.05
6	0.09	0.01	0.10	0.01	0.72	0.17	0.55	0.11

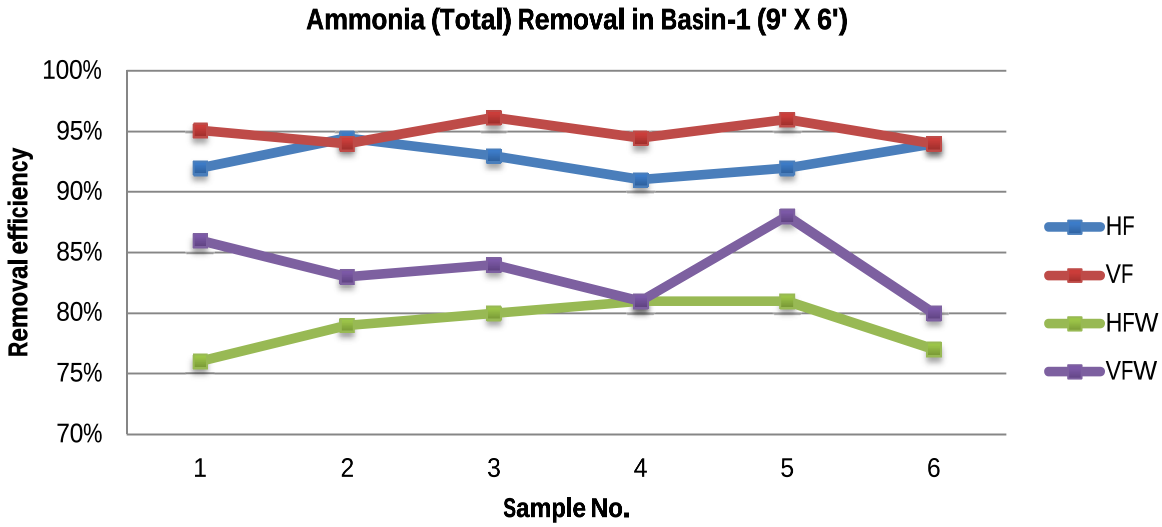

Figure 26: Removal of Total Ammonia in constructed wetland basin-1 under four scenarios.

Table 13 shows the values of concentrations in grey water before (inflow) and after (outflow) treatment in all four combinations in basin 2. The removal efficiency for Ammonia (Total) in basin-2 is shown in Figure 27 as percentage value for all four scenarios in the basin.

Table 13: Concentration of Ammonia (Total) in inflow and outflow samples for different conditions in basin 2.

Sample No	Concentration in HF with vegetation (mg/L)		Concentration in VF with vegetation (mg/L)		Concentration in HF without vegetation (mg/L)		Concentration in VF without vegetation (mg/L)	
	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
1	0.50	0.06	0.39	0.03	0.53	0.15	0.31	0.08
2	0.24	0.03	0.16	0.01	0.17	0.05	0.35	0.08
3	0.36	0.07	0.54	0.05	0.09	0.02	0.59	0.15
4	0.47	0.05	0.68	0.06	0.74	0.21	0.47	0.11
5	0.23	0.02	0.67	0.03	0.71	0.16	0.24	0.06
6	0.18	0.03	0.28	0.03	0.43	0.14	0.33	0.08

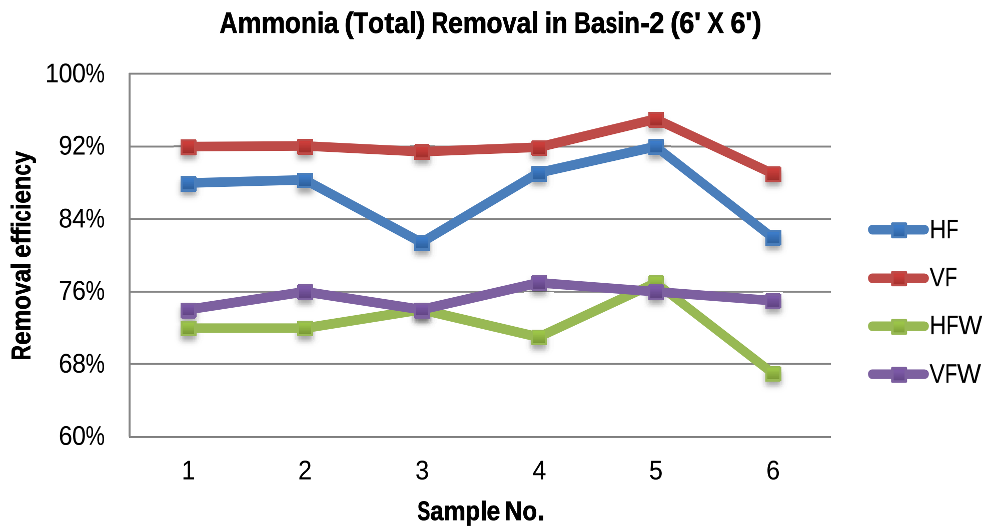

Figure 27: Removal of Total Ammonia in constructed wetland basin-2 under four scenarios.

Comparison of all eight scenarios tested in the study is shown in Figure 28 and Figure 29 that indicate that vertical flow constructed wetlands are more efficient in removing Ammonia (Total) from water. It was also experienced in the study that removal rate is higher with vegetation, which is because of the activities in the root zones that help removing organic contents as availability of oxygen is higher due to open zones created around the roots of the plants. Basin 1 also shows higher efficiency than basin 2. This might be due to greater surface area in basin 1. The high Ammonia (Total) removal rate in sub-surface flow constructed wetlands was also found in many previous research works, which is also proved under this study.

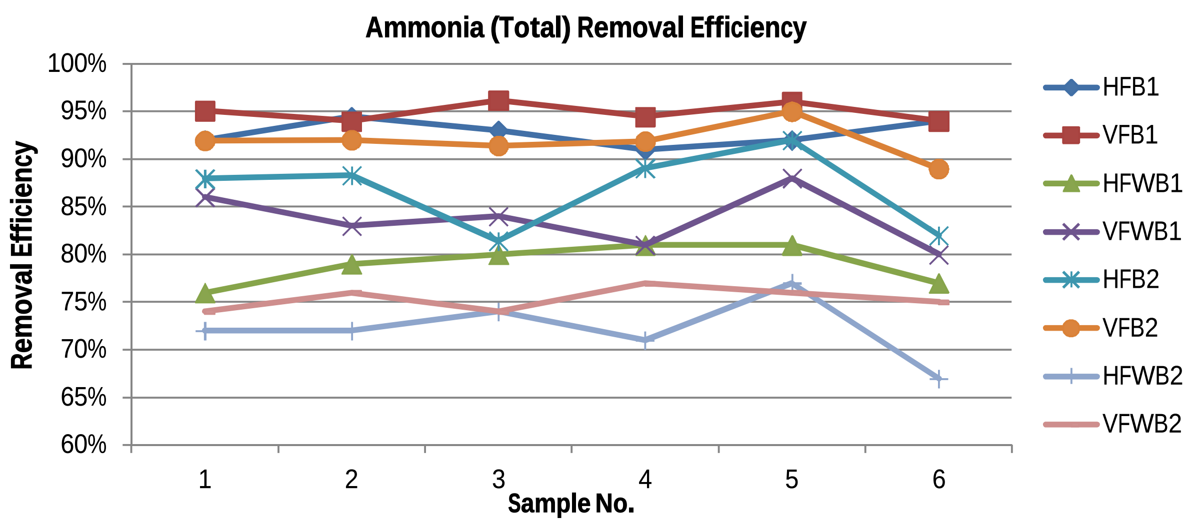

Figure 28: Total Ammonia removal efficiency of constructed wetlands in two different basins with and without vegetation.

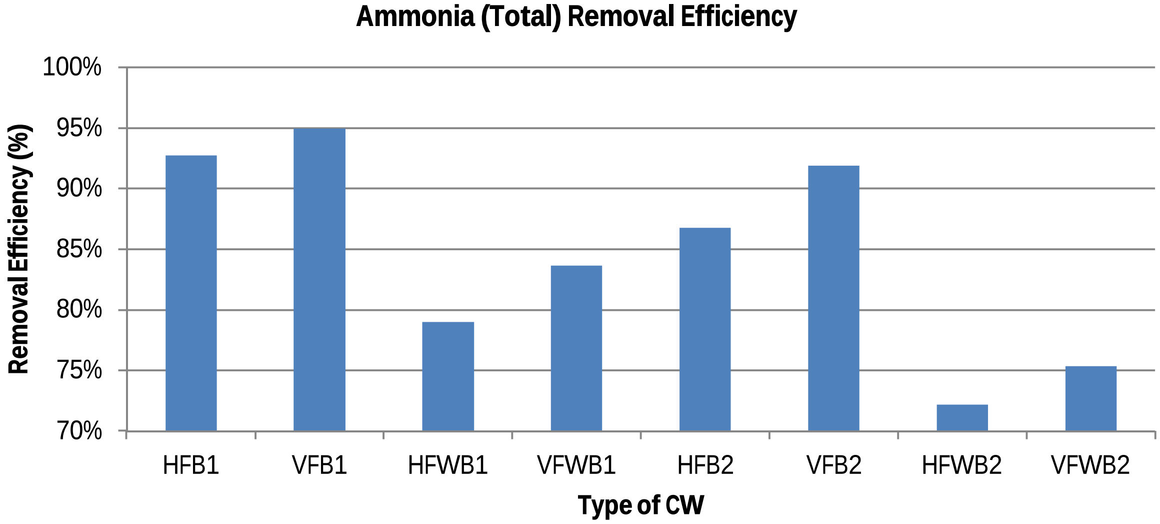

Figure 29: Comparison of Total Ammonia removal efficiency of different types of constructed wetlands in two different basins with and without vegetation.

4.5 Removal of Ortho-Phosphate

Removal of Ortho-phosphate from grey water was found relatively high in all types of constructed wetlands in the study. Concentrations of Ortho-phosphate found in influent range from 0.386 mg/l to 1.03 mg/l. The removal efficiency of Ortho-phosphate in basin-1 is shown in Figure 30 as percentage value for all four scenarios in the basin. Table 14 shows the values of concentrations in grey water before (inflow) and after (outflow) treatment in all four combinations in basin 1.

Table 14: Concentration of Ortho-phosphate in inflow and outflow samples for different conditions in basin 1.

Sample No	Concentration in HF with vegetation (mg/L)		Concentration in VF with vegetation (mg/L)		Concentration in HF without vegetation (mg/L)		Concentration in VF without vegetation (mg/L)	
	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
1	0.41	0.18	0.51	0.17	0.81	0.41	1.03	0.39
2	0.39	0.18	0.68	0.20	0.86	0.41	0.91	0.36
3	0.59	0.24	0.93	0.30	0.74	0.35	0.53	0.23
4	0.57	0.29	0.75	0.20	0.57	0.30	0.72	0.27
5	0.42	0.22	0.84	0.24	0.49	0.22	0.40	0.19
6	0.43	0.17	0.69	0.21	0.96	0.51	0.99	0.33

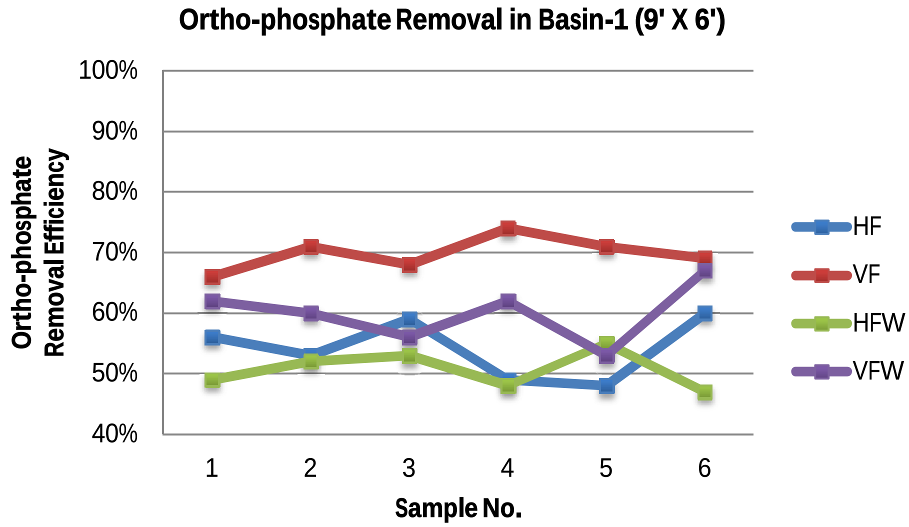

Figure 30: Removal of Ortho-Phosphate in constructed wetland basin-1 under four scenarios.

Table 15 shows the values of concentrations in grey water before (inflow) and after (outflow) treatment in all four combinations in basin 2. The removal efficiency for Ortho-phosphate in basin-2 is shown in Figure 31 as percentage value for all four scenarios in the basin 2.

Table 15: Concentration of Ortho-phosphate in inflow and outflow samples for different conditions in basin 2.

Sample No	Concentration in HF with vegetation (mg/L)		Concentration in VF with vegetation (mg/L)		Concentration in HF without vegetation (mg/L)		Concentration in VF without vegetation (mg/L)	
	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
1	1.00	0.49	0.98	0.43	0.71	0.41	0.46	0.19
2	0.81	0.43	0.52	0.22	0.56	0.31	0.86	0.34
3	0.39	0.20	0.67	0.33	0.95	0.51	0.59	0.24
4	0.75	0.41	0.54	0.21	0.64	0.35	0.47	0.20
5	0.46	0.23	0.43	0.18	0.47	0.27	0.39	0.17
6	0.67	0.37	1.02	0.46	0.82	0.43	0.85	0.32

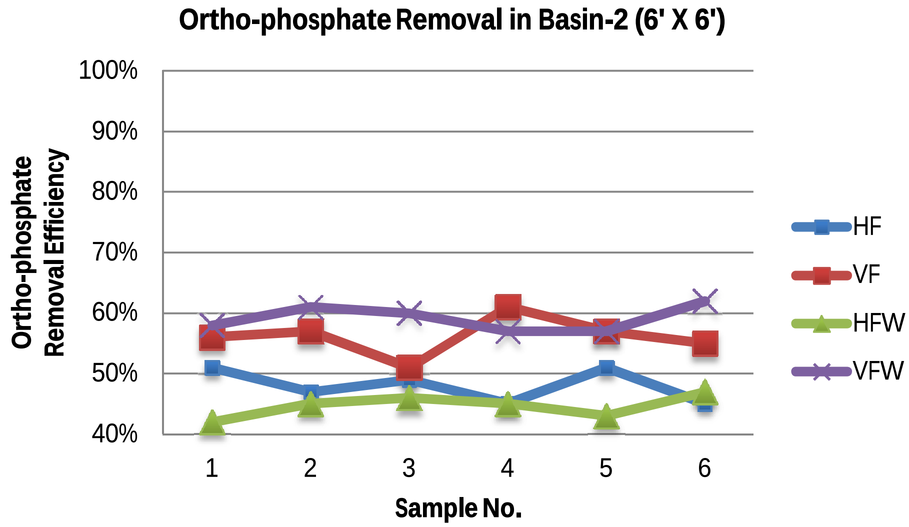

Figure 31: Removal of Ortho-Phosphate in constructed wetland basin-2 under four scenarios.

Comparison of all eight scenarios tested in the study is shown in Figure 32 and Figure 33 that indicate that vertical flow constructed wetlands are more efficient in removing Ammonia (Total) from water. It was also experienced in the study that removal rate is higher with vegetation. Basin 1 also shows higher efficiency than basin 2. This might be due to greater surface area in basin 1. A similar study conducted in Nepal by Bista and Khatiwada revealed that removal efficiency in subsurface flow constructed wetlands for Ortho-phosphate was around 75%, which is also close to the range of removal efficiency found in this study.

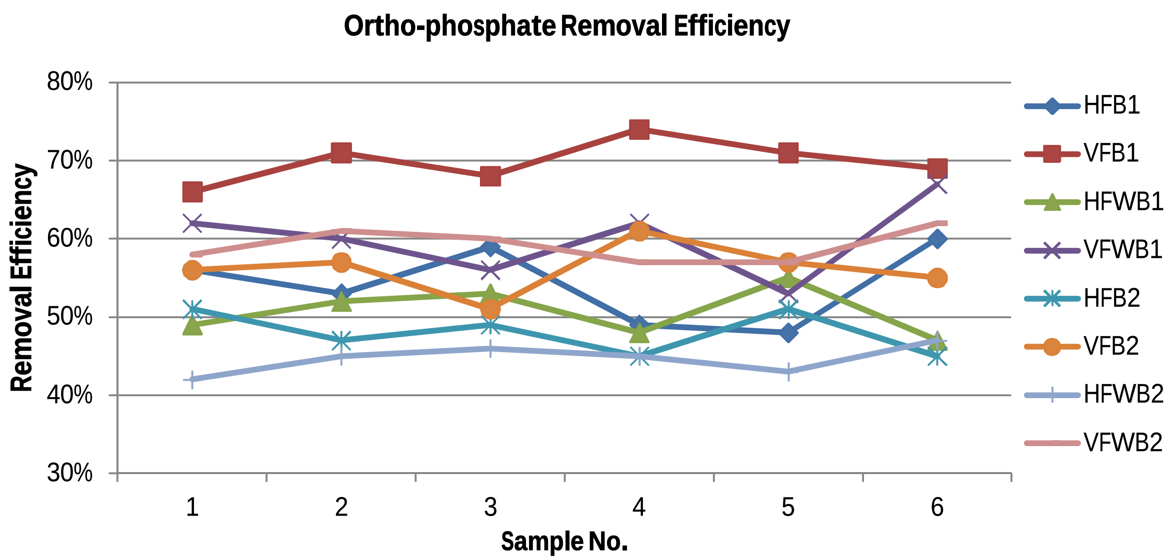

Figure 32: Ortho-Phosphate removal efficiency of constructed wetlands in two different basins with and without vegetation.

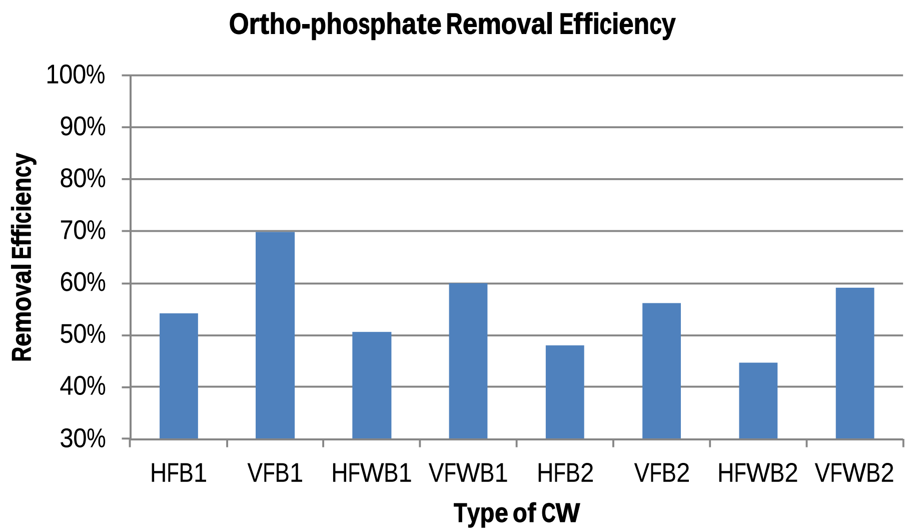

Figure 33: Comparison of Ortho-Phosphate removal efficiency of different types of constructed wetlands in two different basins with and without vegetation.

4.6 Removal of Fecal Coliform

Removal of Fecal Coliform (FC) was found very high in the constructed wetlands under all four scenarios. The removal efficiency of FC in basin-1 and basin-2 is shown below for all four types of scenario in each basin from Table 16 to 23. It can be said from the results that except for few cases, the constructed wetlands, regardless of the vegetation condition, can remove 90% Fecal Coliform from grey water.

Table 16: Removal of fecal coliform by horizontal flow system in basin 1 with vegetation.

Basin Type	Sample		% removal
	Inflow	Outflow	
HFB1	660	46	93%
	730	12	98%
	370	4	99%
	990	19	98%
	TNTC	31	-
	580	7	99%

Table 17: Removal of fecal coliform by horizontal flow system in basin 2 with vegetation.

Basin Type	Sample		% removal
	Inflow	Outflow	
HFB2	340	31	91%
	370	49	87%
	840	13	98%

Basin Type	Sample		% removal
	Inflow	Outflow	
	TNTC	12	-
	120	22	82%
	290	18	94%

Table 18: Removal of fecal coliform by vertical flow system in basin 1 with vegetation.

Basin Type	Sample		% removal
	Inflow	Outflow	
VFB1	640	23	96%
	290	35	88%
	810	36	96%
	880	8	99%
	370	6	98%
	630	24	96%

Table 19: Removal of fecal coliform by horizontal flow system in basin 2 with vegetation.

Basin Type	Sample		% removal
	Inflow	Outflow	
VFB2	910	14	98%
	130	9	93%
	620	16	97%
	470	5	99%
	410	8	98%
	590	10	98%

Table 20: Removal of fecal coliform by horizontal flow system in basin 1 without vegetation.

Basin Type	Sample		% removal
	Inflow	Outflow	
HFB1W	530	17	97%
	280	16	94%
	260	3	99%
	170	24	86%
	590	32	95%
	TNTC	8	-

Table 21: Removal of fecal coliform by horizontal flow system in basin 2 without vegetation.

Basin Type	Sample		% removal
	Inflow	Outflow	
HFB2W	480	9	98%
	TNTC	17	-
	680	23	97%

Basin Type	Sample		% removal
	Inflow	Outflow	
	940	10	99%
	TNTC	27	-
	190	13	93%

Table 22: Removal of fecal coliform by vertical flow system in basin 1 without vegetation.

Basin Type	Sample		% removal
	Inflow	Outflow	
VFB1W	230	12	95%
	480	3	99%
	720	9	99%
	780	24	97%
	750	15	98%
	330	39	88%

Table 23: Removal of fecal coliform by vertical flow system in basin 2 without vegetation.

Basin Type	Sample		% removal
	Inflow	Outflow	
VFB2W	700	4	99%
	120	13	89%
	280	11	96%
	740	6	99%
	140	17	88%
	290	12	96%

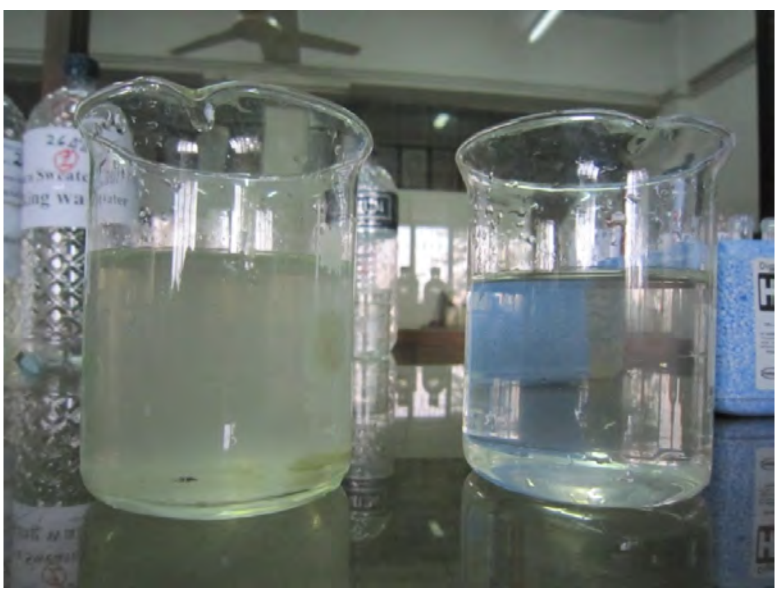

Figure 34: Grey water before and after treatment in the laboratory.

4.7 Sizing of the Constructed Wetland

Effect of different length-width ratio on removal efficiency was also tested in the study. From Table 24 to 27, removal performance of two types of basin with two different lengths are shown. The results show that in almost all the cases, except for removal of Fecal Coliform in horizontal flow without vegetation, the basin with longer (basin 1) dimension in length has greater removal efficiency than the basin with shorter dimension (basin 2).

Table 24: Comparison of removal efficiency in horizontal flow system with vegetation between basin 1 and basin 2.

Horizontal Flow With Vegetation			
Sl. No	Parameter	Average Removal Efficiency (%)	
		Basin 1	Basin 2
1	Total Suspended Solids	80	73
2	BOD (5)	97	94
3	COD	95	91
4	Total Ammonia	93	87
5	Orthophosphate	54	48
6	Fecal coliform	97	90

Table 25: Comparison of removal efficiency in horizontal flow system without vegetation between basin 1 and basin 2.

Horizontal Flow Without Vegetation			
Sl. No	Parameter	Average Removal Efficiency (%)	
		Basin 1	Basin 2
1	Total Suspended Solids	85	73
2	BOD (5)	95	93
3	COD	91	88
4	Total Ammonia	79	72
5	Orthophosphate	51	45
6	Fecal coliform	94	96

Table 26: Comparison of removal efficiency in vertical flow system with vegetation between basin 1 and basin 2.

Vertical Flow With Vegetation			
Sl. No	Parameter	Average Removal Efficiency (%)	
		Basin 1	Basin 2
1	Total Suspended Solids	89	83
2	BOD (5)	99	97
3	COD	97	95
4	Total Ammonia	95	92
5	Orthophosphate	70	56
6	Fecal coliform	95	97

Table 27: Comparison of removal efficiency in vertical flow system without vegetation between basin 1 and basin 2.

Vertical Flow Without Vegetation			
Sl. No	Parameter	Average Removal Efficiency (%)	
		Basin 1	Basin 2
1	Total Suspended Solids	91	84
2	BOD (5)	97	95
3	COD	93	91
4	Total Ammonia	84	75
5	Orthophosphate	60	59
6	Fecal coliform	96	94

At household level, to treat grey water, calculation for the sizing of the basin is often found a complicated task as the sizing depends on influent. Basin-1 with effective dimensions of 9' X 6' X 4.5' (surface area = 5 m^2), showed higher efficiency than basin-2 which has effective dimensions of 6' X 6' X 4.5' (surface area = 3.75 m^2).

Since the top layer of the filter media is find sand, which has the lowest rate of permeability among the three types of materials used, the rate of water passing through this layer would govern the condition for the basin getting flooded or inundated. Two types of flows tried in the study, 10-12 L/min and 18-24 L/min, by controlling the inlet pipes, have shown that if 1,000 L of grey water is allowed to flow into the basin in 1.5 hour, it will not cause flooding at top layer. But when the flow rate was increased to 18-24 L/min, it caused temporary flooding, This indicates that in a day, almost 18,000 L grey water can be treated through these basins, for the materials used in the basin, after sedimentation. But this may not be true for other areas where the composition of materials, especially sand, is different, hence the permeability would be different. Therefore, it is suggested that the surface area should be calculated considering the permeability of sand layer in the constructed wetland.

However, the findings from the study indicate that a basin with surface area less than 5 m^2 is good enough to treat almost 18,000 L of grey water per day. Normally, 60-70% of total wastewater from households is grey water. Assuming that per capita use per day is 225 L, then a constructed wetland with surface area of 5 m^2 would treat grey water of almost 20 families (6 person/family). But it should be noted that the flow rate should not be above 12 L/min. In that case a sedimentation tank should be designed to control the flow rate into the basin.

4.8 Cost Estimation

Constructed wetlands has been known as low-cost solution for treatment of waste water, which has also a very low operation and maintenance cost. Comparing to other technologies, it needs very small initial investment too. The major part of the cost belongs to sedimentation tank, lining and filter materials. The vegetation can be selected locally. The total cost of the basin used in the study is shown in Table 28. The total cost of construction for basin 1 was around BDT 52,000. It should be noted that this basin can be used to treat grey water produced by 20 families by increasing the sedimentation tank size.

If the basin is placed on ground, then excavation cost will not be needed. Considering the aesthetics of the households, other components can be added for beautification too.

Table 28: Estimated cost for a constructed wetland of dimensions similar of basin-1.

Item No	Item Description	Cost (BDT)
1	Excavation	2,000
2	Brick lining	20,000
3	Filter materials	13,500
4	Sedimentation tank	10,000
5	Pipe and joint fixtures	5,000
6	Vegetation	2,000
Total		**52,500**

Chapter 5: Conclusion and Recommendation

5.1 Conclusion

Water insecurity is gradually becoming a major threat with the rapidly growing rate of urbanization. It is necessary now to explore ways of using resources in a sustainable way to reduce pressure on centralized systems. Through recycling and reuse of generated wastewater, this crisis could definitely be reduced which also will make the currently available fresh water sufficient for the city. This study on sub-surface flow constructed wetland systems tested whether it can be a reliable treatment method for recycling grey water (typically 60-70 % of total wastewater) which have low organic and nutrient loading. The performance of sub-surface constructed wetlands to treat grey water at household level is tested in this experiment where the removal efficiency of BOD_5, COD, ammonia (total), orthophosphate, TSS, and fecal coliform were tested. From the test results, the major findings are:

- Removal efficiency of TSS varies in the range from 70-95%. Vertical flow constructed wetlands are more efficient in removing suspended solids from water. It was also experienced in the study that removal rate is higher without vegetation, which is likely because of the absence of root zones that help water to travel easily into the deeper layers of the filter media. Basin 1 also shows higher efficiency than basin 2 that indicates that higher travel time and length increase removal efficiency.
- Removal efficiency of BOD is more than 90%. Vertical flow constructed wetlands are more efficient in reducing BOD from water than horizontal flow systems. It was also experienced in the study that BOD removal rate is slightly higher with vegetation. Basin 1 also shows higher efficiency than basin 2 that indicates that higher travel time and length increase removal efficiency.
- Removal efficiency of BOD is more than 90%. Vertical flow constructed wetlands are more efficient in reducing COD from water than horizontal flow systems. It was also experienced in the study that removal rate is higher with vegetation. Basin 1 also shows higher efficiency than basin 2 that indicates that higher travel time and length increase removal efficiency.
- Removal efficiency of total ammonia varies in the range from 70-95%. Vertical flow constructed wetlands are more efficient in reducing total ammonia from water than horizontal flow systems. It was also experienced in the study that removal rate is higher with vegetation. Basin 1 also shows higher efficiency than basin 2 that indicates that higher travel time and length increase removal efficiency.
- Removal efficiency of orthophosphate varies in the range from 45-70%. Vertical flow constructed wetlands are more efficient in reducing Ortho-Phosphate from water than horizontal flow systems. It was also experienced in the study that removal rate is higher

with vegetation. Basin 1 also shows higher efficiency than basin 2 that indicates that higher travel time and length increase removal efficiency.

- Removal of Fecal Coliform (FC) has been found high in constructed wetlands. On average, the wetlands remove more than 90% fecal coliform from grey water.

The findings from the study indicate that a basin with surface area less than 5 m^2 is good enough to treat almost 18,000 L of grey water per day. Normally, 60-70% of total wastewater from households is grey water. Assuming that per capita use per day is 225 L, then a constructed wetland with surface area of 5 m^2 would treat grey water of almost 20 families (6 person/family). But it should be noted that the flow rate should not be above 12 L/min. In that case a sedimentation tank should be designed to control the flow rate into the basin.

Constructed wetlands with horizontal or vertical sub-surface flow are a viable alternative for grey water treatment for organics, nutrients and suspended solids removal. Removal of organics (BOD_5 and COD) is very high of these systems. Removal of color and suspended solids also show good results for both the systems. As the concentration of nutrients is not very high in greywater, the removal efficiency of total ammonia and orthophosphate were also found very high for both the systems. The high removal of total ammonia indicates the passage through the roots for oxygen under the soil layer.

5.2 Recommendation

For treating grey water where total ammonia and orthophosphate concentration are low, soil can be considered with the used emergent plants where the soil will help adsorption of phosphorus and also will help keeping the normal landscape condition. Being simple in construction, operation and maintenance, sub-surface flow constructed wetlands can be adopted in towns, institutions and households. Any of these two systems could be used to recycle and reuse grey water and mitigate the water crisis of Dhaka and any other cities of Bangladesh.

Water shortage in Dhaka will be a key issue for its sustainable development in the future. Treated grey water can play a major role in substituting and supplementing DWASA's water supply. The potential of potable water savings can be substantial by using these proposed grey water treatment systems. It is clear that efficient, cheap, reliable and sustainable on-site grey water treatment system is indispensable in order to avoid technical problems and public health risk as well as promotion of public acceptance. There are numerous ways that water reuse can be implemented to supplement current water supplies. Bangladesh Govt. may launch water reuse projects through public utility departments such as the Water Administration and Sewerage Authority (WASA), the City Corporations, the Department of Public Health Engineering (DPHE) etc. Private organizations like operational and campaigning NGOs can work parallel. They can deal with water quality and water rights requirements for such projects, and address other important issues such as human health, environmental impacts, economics and project funding. To promote grey water reuse in the country, building codes should be modified, and standards and guidelines should be adopted. The standards should be realistic, enforceable, and sensible to public and environmental health.

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