Escola de Camins
Escola Tècnica Superior d'Enginyeria de Camins, Canals i Ports
UPC BARCELONATECH

DISSERTATION

Title

NATURAL SYSTEMS FOR WASTEWATER TREATMENT IN WARM CLIMATE REGIONS

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Date

JULY 2014

ABSTRACT

Water scarcity and the high costs of new water supply technologies are the two major factors responsible for the increasing recognition of the importance to conserve water resources by wastewater treatment, reuse or reclamation.

Sustainability of sanitation systems should be related to low cost, low energy consumption and operation and maintenance requirements, especially for small communities in developing countries. Hence, natural systems for wastewater treatment seem to be a suitable solution.

In this study, a review of two natural systems, constructed wetlands and stabilization ponds, was carried out. Thereby, strengths and weakness of both systems have been analysed.

Furthermore, this dissertation evaluates the robustness of a pilot-scale hybrid constructed wetland system to cope with a heavy rain episode, a characteristic phenomenon in tropical climate regions.

During three months (from June to September 2013), the pilot plant operated under an input flow of 33 l/hour (0.27 m/day HLR in vertical CWs). Under these conditions, the system showed good mass removal rates for all contaminants, (96.6% for TSS, 95.5% for BOD_5, 77.6% for COD and 90.8% for NH_4-N). These results were compared to that obtained in previous studies carried out during the coldest months of the year.

In September 2013, a heavy rain episode was simulated. The pilot plant operated under an input flow of 330 l/hour (300l/h of potable water plus 33 l/h of real wastewater) for one hour. The removal rates were high (above 77.6% for all parameters), and the contaminants concentrations seemed to return to the normal values about 7 hours after the campaign.

In summary, it can be concluded that the technology of constructed wetlands is a valid solution for wastewater treatment for small communities in warm climate regions. Likewise, these systems can cope with sharp fluctuations in flow to be treated.

Keywords: BOD, COD, constructed wetlands, free water surface, horizontal subsurface flow, hydraulic load, natural systems, NH_4-N, stabilization ponds, TSS, vertical subsurface flow, wastewater stabilization ponds, and wastewater treatment.

ACKNOWLEDGEMENTS

This research has been funded by the European Commission for the financial support of the NaWaTech project (Grant Agreement N°: 308336).

I would like to be thankful to all those that helped me, especially Carlos, my laboratory partner, and Javi.

Also thanks to my tutors Anna and Joan for helping and guide me in this work.

To Victor, for his patience and support

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1. GLOSSARY

Abiotic: non-biological processes

Adsorption: adherence by chemical or physical bonding of a pollutant to a solid surface (U.S. EPA, 2000).

Aerobic treatment: wastewater treatment processes that take place in the presence of dissolved oxygen.

Anaerobic treatment: wastewater treatment processes that take place in the absence of dissolved oxygen and other terminal electron acceptors (nitrates, sulphates, etc.)

Anoxic treatment: wastewater treatment processes that take place in the absence of dissolved oxygen.

Chelation: substances that bind metal ions.

Denitrification: biotic conversion of nitrate to nitrogen gases.

Detritus: loose, dead leaves and stems from dead vegetation (U.S. EPA, 2000).

Diffusion: chemical mechanism of transport due to gradient concentration

Flocculation: agglomeration of colloidal particulates into larger, settleable solids which can be removed by sedimentation process.

Humic acids: organic acid produced by the biodegradation of dead organic matter.

Hydrolysis: is a chemical process involving the breaking of a bond in a molecule using a molecule of water. One fragment of the molecule gains a hydrogen ions and the remaining collects the hydroxyl group.

Liquor: mixture of raw wastewater and activated sludge

Mineralization: process in which an organic substance is converted or oxidized into an inorganic substance and derivatives.

Nitrification: biotic conversion of ammonium nitrogen to nitrite and nitrate.

Periphyton: aquatic organisms that grow on submerged substrates.

Photic zone: is the depth of the water in a lake or ocean that is exposed to enough sunlight for photosynthesis to occur.

Photolysis: is an abiotic chemical process in which a chemical compound is broken down and degraded by photons.

Phytoplankton: algae that is microscopic in size which floats or drifts in the upper layer of the water column and depends on photosynthesis and the presence of phosphorus and nitrogen in water (U.S. EPA, 2000).

Protoplasm: are the living contents of a cell, composed of a mixture of small molecules such as ions, amino acid, water and macromolecules such as nucleic acids, proteins and lipids.

Wetland hydraulics: refers to the movement of water through constructed wetlands, including volumes, velocities, flow patterns and other characteristics.

2. ACRONYMS

AnP	Anaerobic Pond
COD	Chemical Oxygen Demand
CW	Constructed Wetlands
BOD	Biochemical Oxygen Demand
BOD_5	5th day Biochemical Oxygen Demand
DO	Dissolved Oxygen
EPA	Environmental Protection Agency
ET	Evapotranspiration
FP	Facultative Pond
FWS	Free Water Surface systems
HLR	Hydraulic Loading Rate
HRT	Hydraulic Residence Time
HUSB	Hydrolytic Upflow Sludge Blanket reactor
MP	Maturation ponds
NH_4-N	Ammonia-Nitrogen
OLR	Organic Loading Rate
O&M	Operation and Maintenance
PAH	Poly-aromatic Hydrocarbons
P.E.	Population Equivalent
PPCP	Pharmaceutical and Personal Care Products
SSF	Subsurface Flow Systems
TSS	Total Suspended Solids
UASB	Upflow Anaerobic Sludge Blanket reactor
VS	Volatile Solids
WSP	Wastewater Stabilization Ponds
WW	Wastewater
WWTP	Wastewater Treatment Plant

3. INTRODUCTION

Wastewater engineering is the branch of environmental engineering that tries to solve the issues associated with wastewater treatment and reuse. Its goal is to protect the public health and at the same time, to ensure environmental, economic, social and political sustainability (Metcalf and Eddy, 2003).

Water scarcity in tropical and subtropical regions and the high costs of new water supply technologies are the two major factors responsible for the increasing recognition of the need to conserve water resources by wastewater treatment, re-use or reclamation (Mara, 1976).

Wastewater (WW) may be purely domestic in origin or it may contain some industrial or agricultural WW as well. It is hazardous in content, mainly because the number of disease-causing (pathogens) organisms present in sewage (Mara, 1976). It has to be treated before its disposal in a receiving watercourse in order to reduce diseases spreading caused by pathogenic organisms and to prevent pollution of surface and groundwater.

On May 1991, the EU Council Directive 91/271/EEC concerning urban WW treatment was adopted. This Directive takes into account the collection, treatment and discharge of urban WW and the treatment and discharge of WW from certain industrial sector, in order to protect the environment. According to this directive, all Member States shall ensure that urban WW from agglomeration from 2000 P.E. has to be subject to at least a secondary treatment before discharge. Moreover, any agglomeration of less than 2000 P.E. has to treat WW before discharge. Against a backdrop of rise in energy price and labour costs, natural WW treatment systems have become an attractive alternative for some communities, especially for developing countries and small rural communities. Small communities cannot afford the cost of advanced and specialized systems, which require trained and qualified personnel.

The high cost of some conventional treatment processes and the implementation of environmental policies have produced economic pressures in order to search for cost-effective, aesthetic and environmental friendly ways to control water pollution. Under these conditions, natural systems for WW treatment have become popular.

Natural systems for WW treatment are a biological system in which water treatment depends on natural compounds and does not depend on external energy sources exclusively to maintain the major treatment responses (Crites et al., 2006). There are three main categories: aquatic, terrestrial and wetland systems.

In tropical and subtropical regions of developing countries capital is scarce but labour is plentiful and relatively cheap. In this context labour intensive schemes are economic and more advantageous from a social point of view (Mara, 1976). Hence, natural systems to treat WW are a feasible solution.

In this study a review of natural systems for WW treatment for tropical and subtropical climate regions was carried out. It was focused on constructed wetlands and waste stabilization ponds.

Constructed wetlands (CWs) are engineering systems that have been designed and constructed to utilize natural processes involving wetland vegetation, soils and the associated microbial assemblages to assist in treating WW (Vymazal, 2005). There are basically two types of CWs: free water surface (FWS) systems and subsurface flow (SSF) systems. The latter can be subdivided into horizontal or vertical systems, depending on the water circulation pattern. Various CWs are usually combined in order to increase their treatment efficiency or reduce their hydraulic residence time (HRT). When various CW configurations are combined, they are called hybrid systems.

Waste stabilization ponds (WSPs) are large shallow basins in which sewage is treated by entirely natural processes involving both algae and bacteria. In WSPs, the biodegradable organic matter is stabilized by micro-organisms and the number of disease-causing agents is reduced significantly, mainly because of the long retention time required for stabilization. There are three types of pond: anaerobic, facultative and maturation ponds (Mara, 1976). Just like CWs, they can be combined in order to obtain a better quality effluent.

In order to improve the quality of the final effluent, natural systems are usually preceded by a primary treatment. Anaerobic biological reactors, such as Upflow Anaerobic Sludge Blanket Reactor (UASB) have been becoming popular because of its higher degree of solids removal in comparison with a conventional settler (Pedescoll et al., 2011). The concentrations of organic matter and suspended solids are reduced drastically, improving the rates of system performance.

Furthermore, in this study the robustness of an experimental hybrid CW pilot system was assessed during three months. Moreover, a heavy rain episode, a characteristic phenomenon of tropical climate regions, was simulated, in order to assess the appropriateness of this system for warm climate regions.

4. OBJECTIVES

Main objective:

Assessing the performance of natural systems for WW treatment in warm climate regions.

Specific objectives:

- ✓ **Review of natural systems for WW treatment in warm climate regions**

A review of natural systems for WW treatment, specifically CWs and WSPs was carried out. It focused on the mechanisms for pollutants removal and their performance rates in warm climate regions.

- ✓ **Evaluating the performance of a hybrid CW system at pilot scale**

The performance of a hybrid CWs system at pilot scale was evaluated. The system was operated under an input flow of 800 L d^{-1} of real WW.

- ✓ **Evaluating the performance of the hybrid CW system under heavy rain episodes**

Heavy rain episodes where simulated to assess the robustness and the buffer capacity of the CW system. The heavy rain episode was simulated by increasing the input flow up to 10 times during 1h, mixing the usual hydraulic loading rate with clean water.

5. NATURAL SYSTEMS FOR WASTEWATER TREATMENT

Natural systems for WW treatment are a biological system in which purification is carried out without energy input, and therefore reactions responsible for water purification occur very slowly (Salas et al., 2007).

For this reason, hydraulic retention time in these systems can be even 100 times higher than in conventional treatment systems. This is the reason why they required larger land areas to treat the same water flow than in conventional systems.

The main features of natural treatments are:

- Reliability: natural systems are very reliable in extreme operating conditions. They can treat a variety range of WW and they work under a wide range of weather conditions (W.P.C.F., 1990)
- Environmental benefits: aesthetic and wildlife are insured
- Simplicity of the plants design: maintenance can be carried out by low skilled workers
- Low operation and maintenance costs

Natural WW systems are simple, cost-effective and efficient methods to treat WW. They are usually applied as secondary or tertiary treatment, allowing the removal of most of the bacteria, microorganism and organic matter.

In tropical and subtropical developing countries where capital is scare but labour plentiful and relatively cheap, labour intensive schemes are economically and socially more advantageous (Mara, 1976). Hence, natural systems to treat WW are a feasible solution. Moreover in these regions, generally sufficient land is normally available. Nevertheless, few studies assessed the robustness of these systems during heavy rain episodes.

Concept	Capital Cost ($/m^3·d)	O&M Cost ($/m^3·d)
Slow rate infiltration[1]	800-2000	0.10-0.20
Rapid infiltration[2]	450-900	0.05-0.10
Overland flow[3]	600-1000	0.08-0.15
CWs[4]	500-1000	0.03-0.09
WSPs[5]	500-1000	0.07-0.13

[1] Includes an allowance for pre-treatment and storage, flow rate ≈ 400m^3/d
[2] With pre-treatment to primary, flow rate ≈ 400m^3/d
[3] Pre-treatment: screening or settling, flow rate ≈ 400m^3/d
[4] FWS type, pre-treatment: screening or settling, flow rate ≈ 400m^3/d
[5] No pre-treatment, flow rate ≈ 400m^3/d

Table 1: Typical construction and O&M costs for natural systems (W.P.C.F, 1990)

6. PRELIMINARY AND PRIMARY TREATMENT

6.1. SCREENING

The first stage of WW treatment is the removal of large floating objects and heavy mineral particles. These materials can damage the equipment used, clog the pipes and accumulate on the surface of the secondary treatment systems.

Coarse solids are usually removed by screens. The spacing between bars is usually 15-25 mm. At small works, screens are raked by hand and, in order to facilitate this, the screens are inclined, commonly at 60° to the horizontal.

6.2. UPFLOW ANAEROBIC SLUDGE BLANKET REACTOR

A primary treatment is strongly recommendable in order to reduce the organic loading and suspended solids of the effluent.

Recently, anaerobic reactors have been becoming popular as primary treatment, including Upflow Anaerobic Reactors, Imhoff tanks and Hydrolytic Upflow Sludge Blanket reactors.

UASBs are high-rate anaerobic digesters. They were developed in the 1970's by Professor Lettinga, and they have been extensively tested at full-scale in tropical and subtropical regions, particularly in Brazil, Colombia and India (Mara, 2003). They have been used for the primary treatment of domestic and mixed WW and high-strength biodegradable industrial and agro-industrial WW.

UASBs are reinforced-concrete structures with a short hydraulic retention time, of the order of 6-12 hours. As shown in figure 1, the raw WW is distributed across the base of the reactor and flows upwards through the sludge layer, what ensures the thorough contact between the WW and the anaerobic bacteria in the sludge. The liquor rises through the reactor, and during this time, the biodegradation of organic matter occurs, and reaches the "phase separator". This is the important characteristic of this type of anaerobic reactor: it divides the reactor into its two constituent zones, the lower digestion zone and the upper settling zone. As the liquor rises through the settling zone, its Upflow velocity decreases due to the inclined surface of the phase separator, and the suspended sludge particles settle out. Finally the weight of the accumulated sludge particles exceeds the frictional force that keeps them on the inclined surfaces, and the settle down to the sludge layer. The phase separator enables an effluent with a very low suspended solids concentration to be discharged from the reactor.

Biogas bubbles are collected under the phase separator, from where the gas is easily extracted and can be re-used. Deflectors are placed between the phase separators units to prevent any biogas bubbles entering the settling zone where they would hinder sedimentation.

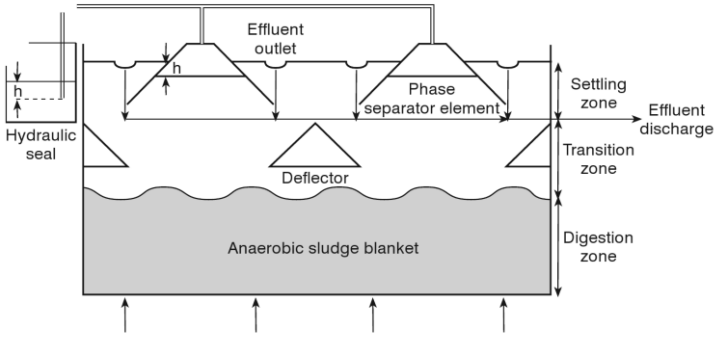

Figure 1: Schematic diagram of an UASB (Van Haandel and Lettinga, 1994)

UASBs produce quite large amounts of waste sludge, 0.2kg of sludge/kg of BOD removed. This is much less than in conventional activated sludge plants, but much more than in anaerobic ponds. In warm climates, UASB waste sludge can be simply dewatered on drying beds.

Hydrolytic upflow sludge blankets (HUSB) are essentially UASB reactors operated at a lower HRT, from 2 to 5 hours, in order to avoid methanogenesis reaction wherever possible. In general, solids retention time in HUSB reactors is maintained for over 15 days in order to achieve high hydrolysis rates of WW solids. HUSB reactors have been recently investigated as a suitable primary treatment for CWs, mainly because they can provide effluents with lower TSS and COD concentration than standard primary treatments (Pedescoll et al., 2011).

7. CONSTRUCTED WETLANDS

7.1. HISTORY AND INTRODUCTION:

Wetlands are land areas that are wet during part or all of the year. They are frequently transitional between uplands and continuously or deeply flooded systems (Kadlec and Wallace, 2008).

In many coastal plain areas of the southeastern U.S. and in the poorly drained marshes and fens of the north, natural wetlands have historically been used as convenient receiving waters for WW discharges. Several of them, including the *Houghton Lake* fen in Michigan or the *Florida cypress dones* designed for WW management, were extensively studied in the U.S. and it has been recognised that their treatment capacity is quite unknown due to their variability and changes over time (W.P.C.F., 1990).

The first attempts to use the wetland vegetation to remove various pollutants from water were in early 1950s. The first full-scale FWS was built in The Netherlands to treat WW from a camping site during the period 1967-1969. In late 1980s, soil was replaced with coarse materials (washed gravel) and this set-up has been successfully used since then (Vyzamal, 2005).

Through the 1980s, a more thorough understanding has developed the specific strengths and weakness of CWs as treatment systems. This process has mirrored the increasing acceptance of the use of upland systems for WW renovation (W.P.C.F., 1990) and this treatment technology rapidly spread around the world.

In 1990s, the increased needed for nitrogen removal from WW led to more frequent use of vertical flow CWs which provide higher degree of oxygenation in the bed, and the consequent removal of ammonia via nitrification. Few years later, in order to produce simultaneously nitrification and denitrification to remove total nitrogen lead to the use of hybrid systems (Vyzamal, 2005).

The success of these new systems is probably due to the change of the last years towards a sustainable development and the more concern about the resource exploitation. For thirty years, WW treatment plants based in CWs have found greatest popularity in some areas (central and northern Europe) to treat WW of small communities, as shown in Figure 2. Nowadays, this system is used all around the world, including northern countries as well as southern countries (Garcia and Corzo, 2008). In developing countries, CWs are an attractive alternative to conventional WW treatment technologies.

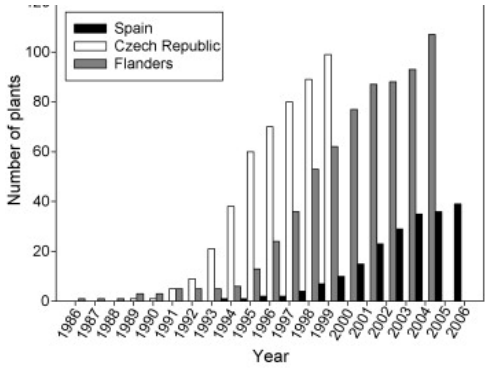

Figure 2: Cumulative number of WW treatment plants based in CWs over the last years in various European regions (Puigagut et al., 2007)

Wetlands have properties that make them unique among the major ecosystems groups. Ample water is important for most forms of biological productivity, and wetlands plants are adapted to take advantage of this abundant supply. Because of this, wetlands are among the most biological productive ecosystems on the planet (Kadlec and Wallace, 2008). Thereby, as they have a higher rate of biological activity than most ecosystems, they can transform many of the common pollutants that occur in conventional WW (Kadlec and Wallace, 2008).

CWs are designed to take advantage of many processes that occur on natural wetlands but in a more controlled environment (Vymazal, 2005). They may be used to treat municipal WW, domestic, animal, mine water and industrial WW, as well as leachate and runoff.

CWs may have several advantages compared to conventional and advanced WW treatment systems. Some of these advantages are:

- ✓ Low cost of construction and maintenance.
- ✓ Low energy requirements.
- ✓ "Low-technology" system, it can be run by relatively low-skilled personnel.
- ✓ Flexible systems and less susceptible to variations in loading rate than conventional treatment systems.

The main disadvantage of CWs is the increased land area required, compared to conventional systems and their possible decreased performance during winter in temperate regions (Moshiri, 1993). Thus, CWs are a viable and advisable solution for tropical developing countries.

7.2. DESIGNS OF CONSTRUCTED WETLANDS:

CWs can be classified based on the water flow pattern. CWs are classified in FWS systems with shallow water depths and SSF system with water flowing literally through the sand or gravel (U.S. EPA, 1988).

There are subcategories under subsurface flow category which depend on the water flow, horizontal or vertical.

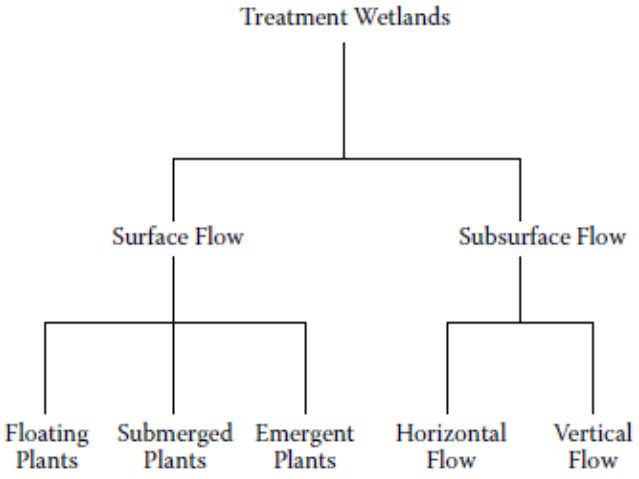

Figure 3: Treatment wetland types (Kadlec and Wallace, 2008)

There are many general functions of vegetation in wetlands, especially with regards to chemical processing and removal. This vegetation may be categorized by their growth habit with respect to the wetlands water surface as (Kadlec and Wallace, 2008):

- Emergent soft tissue plants
- Emergent woody plants
- Submerged aquatic plants
- Floating plants
- Floating mats

Emergent soft tissue macrophytes are the dominant life in wetlands and marshes, growing within a water-table ranging from 50 cm below the soil surface to a water depth of 150 cm or more (Moshiri, 1993).

In general, these macrophytes have an extensive root and rhizome system, and the depth penetration of the root system as well as the sediment volume is different for different species. Some of the most common plant species used in CW design includes cattails (*Typha*), bulrushes (*Scirpus*) and common reed (*Phragmites communis*) (W.P.C.F., 1990). Oxygen is transported through the gas spaces to the roots and rhizomes by diffusion and/or by convective flow of air. Part of the oxygen may leak from the root systems into the surrounding rhizosphere, creating oxidized conditions

in the otherwise anoxic sediment and stimulating both decomposition of organic matter and growth of nitrifying bacteria (Moshiri, 1993).

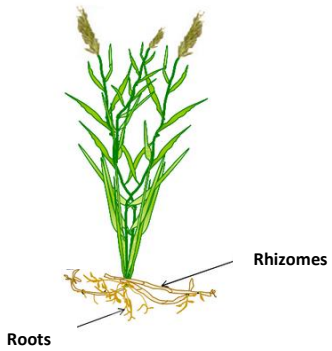

Figure 4: Schematic drawing of common reed, Phragmites communis (Garcia and Corzo, 2008)

The main advantages of the two main designs are listed below.

FWS Systems	SSF Systems
✓ Lower installation cost	✓ Greater cold tolerance
✓ Simpler hydraulics	✓ Minimization of the insect vectors and odour problems
	✓ Greater assimilation potential per unit of land area

Table 2: Main advantages of the different CWs designs (W.P.C.F., 1990)

7.2.1. Free Water Surface System (FWS)

FWS systems can be understood as a modification of natural lagoons with a water depth of 0.3 and 0.4 meters, and plants (Garcia and Corzo, 2008).

WW is directly exposed to the atmosphere, and water flows mainly through the leaves and stems of the plants. The selection of plant species type does not appear to be overly critical to assimilation capacity. This is because of the major role in assimilation played by the microbes that are attached to the plants and present in the wetland surface (W.P.C.F., 1990). The percentage of plant cover appears to be more important than the actual species composition but at the same time, this can suppose a problem with mosquito populations.

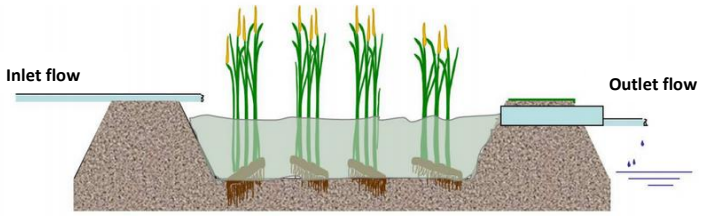

Figure 5: Cross section of a FWS CW (Garcia and Corzo, 2008)

Generally, the distribution of plants in a FWS wetland is not homogeneous, there may be open water zones with submerged plants and other zone fully vegetated. This

heteroigenity may mean in practice different propierties and removal mechanisms in each zone.

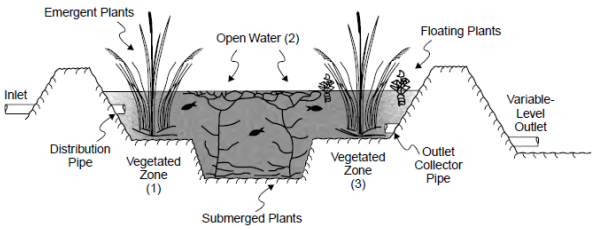

Figure 6: Profile of a three-zone FWS CW cell (U.S. EPA, 2000)

The most common application for FWS wetlands is for advanced treatment of effluent from secondary or tertiary treatment processes (Kadlec and Wallace, 2008).

7.2.2. Subsurface Flow System (SSF)

In SSF wetlands, water flows through the soil medium in contact with the roots and rhizomes of the plants. The water depth is from 0.3 to 0.9 meters.

The biofilm grows adhered to the soil medium and the roots and rhizomes, and it is fundamental to the pollutants removal (Garcia and Corzo, 2008). In SSF systems, bulrush and common reed have the best properties for use due to their root development and sediment aeration potential (W.P.C.F., 1990).

The main advantages of SSFs over FWSs are: bigger treatment capacity (they accept higher OLRs), lower risk of contact between WW and persons, and lower risk of mosquitoes appearance. Nevertheless, they are less useful for environmental restoration projects due to the lack of accessible water sheet (Garcia and Corzo, 2008).

7.2.2.1. Horizontal Subsurface Flow System

In these systems, water flows horizontally through the granular bed. The water layer is 0.3-0.9 meters deep, and it flows 0.05-0.1 meters under the soil surface. They are characterized by working permanently flooded and with OLRs of 6g DBO/m^2·day (Garcia and Corzo, 2008).

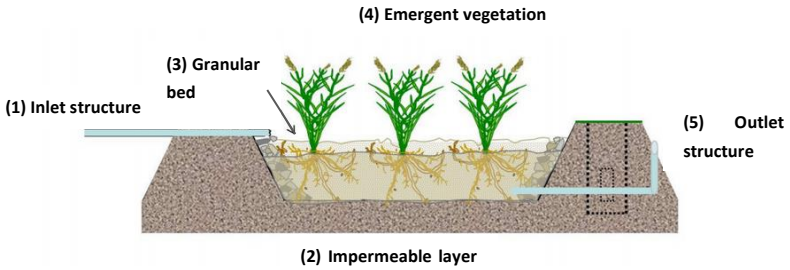

Figure 7: Cross section of a Horizontal SSF system (Garcia and Corzo, 2008 adapted)

- **Waterproofing:** it is necessary to underlay the bed by an impermeable membrane to prevent seepage and groundwater contamination. Depending on the local conditions, it can be enough to compact the soil, in other cases, it will be necessary a clay layer or a synthetic liner (U.S. EPA, 1988).
- **Inlet and outlet structures:** CWs are systems that require a good distribution and collection of the water in order to achieve the estimated outputs. Thus, inlet and outlet structures have to be carefully designed and constructed.

  The WW from the primary treatment has to be collected into a sink, and then water will be homogenously distributed to the wetland. The effluent is collected at a perforated pipe which is at the bottom of the wetland and is connected to an inverse "L"-shaped pipe with adjustable level for water level control in the wetland (Garcia and Corzo, 2008). Outlet structure controls must be able to control depth of water in the wetlands especially for winter ice conditions where deeper wetland conditions are required to maintain treatment levels (U.S. EPA, 1988).
- **Granular bed:** the inlet and outlet are often filled with coarse gravel in order to distinguish these structures from the granular bed. The bed must be clean (exempt from fines), homogeneous, hard, durable and able to keep its shape in long-term. Moreover, it has to support the growth of the emergent vegetation and biofilm. Diameters of 5-8 mm achieve good results.

  The performance of the system depends also on some hydraulic parameters, such as the hydraulic conductivity which determine the stream that can flow through the soil (Garcia and Corzo, 2008).
- **Vegetation:** the most used specie is common reed *Phragmites australis* (Moshiri, 1993). The vegetation has five important functions in the process:
  - ✓ Increase and stabilize the hydraulic conductivity of the soil (Moshiri, 1993).
  - ✓ Supply oxygen to the heterotrophic microorganisms in the rhizosphere: around the roots, there are aerobic microenvironments where microbial processes take place, such as nitrification and aerobic removal of organic matter (Moshiri, 1993).
  - ✓ Roots and rhizomes create a suitable surface for the growth of the biofilm (Garcia and Corzo, 2008).
  - ✓ Temperature variation dampening: when plants grow, they reduce the light intensity incident on the soil surface, avoiding temperature gradient that may affect some processes. Moreover, vegetation protects from freezing (Garcia and Corzo, 2008).
  - ✓ Uptake of nutrients: modest contribution to the nutrients removal in urban WW, but it is higher in diluted WW (Garcia and Corzo, 2008).

7.2.2.2. Vertical Subsurface Flow System

In these systems, water flows pulsed and vertically through the granular bed. Hence, the granular medium is not permanently flooded. The water layer is 0.5-0.8 meters deep, and they work with OLRs of 20g DBO/m^2·day (Garcia and Corzo, 2008).

The vertical wetlands have a higher treatment capacity than the horizontals (they require less treatment surface to treat the same OLR), but they are more liable of clogging (Garcia and Corzo, 2008).

During the loading period, air is forced out of the soil; during drying period, atmospheric air is drawn into the pore-spaces of the soil and diffusive oxygen transport is enhanced, thus increasing soil oxygenation. This operational regime provides alternating oxidizing and reducing conditions in the substrate, stimulating nitrification-denitrification and phosphorus adsorption (Moshiri, 1993).

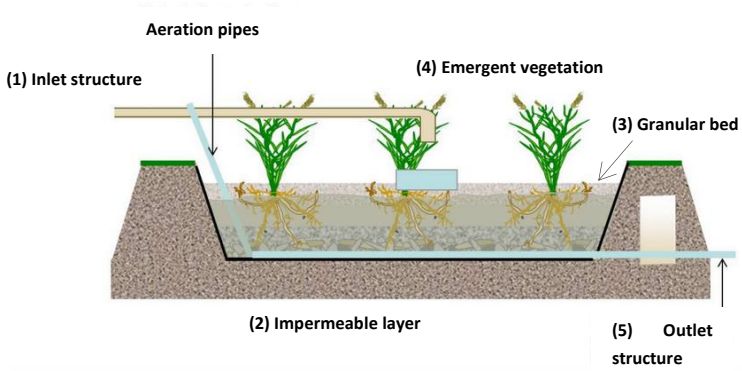

Figure 8: Cross section of a Vertical SSF system (Garcia and Corzo, 2008 adapted)

Vertical wetlands usually include aeration pipes. The aspects of waterproofing and vegetation are exactly the same as in horizontal wetlands.

- **Inlet and outlet structures:** water is distributed through a piping net set out over the surface. Owing to the discontinuous flow, on cold climates, the piping net is buried 0.05-0.1 meters under the surface in order to prevent from freezing. The effluent is collected at a perforated piping net which is at the bottom of the wetland (Garcia and Corzo, 2008).
- **Granular bed:** The bed must be heterogeneous, consisting of three horizontal layers with different gradation, which has to increase with the depth (coarse sand, gravel, and coarse gravel at the bottom) in order to prevent from a low/high flow speed (Garcia and Corzo, 2008).
- **Aeration pipes:** these elements are used to air the bottom of the bed to enhance the aerobic degradation processes and the nitrification. In general, it is recommended the installation of 1 aeration pipe every 4 m2 (Kadlec et al, 2000).

7.2.3. Hybrid Systems

Recently, in order to increase the efficiency of the treatment, especially for nitrogen, various CW configurations have been combined, so-called hybrid systems.

To increase the performance rate of nitrates removal, vertical and horizontal SSFs are combined. While vertical SSFs have proven to provide good conditions for nitrification, horizontal systems provide the anaerobic conditions needed for denitrification.

Hence, combining the strengths and weakness of both systems, it is possible to restore the balance and obtain an effluent low in nitrogen and BOD. Many combinations are possible, using multiple wetlands types, such as vertical SSFs followed by horizontal SSFs and these by FWS wetlands.

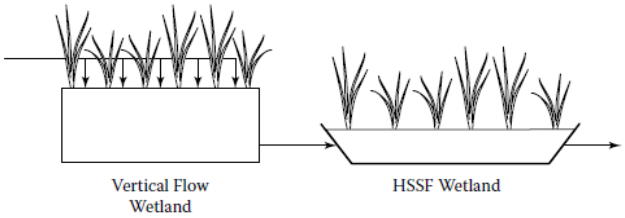

Figure 9: An hybrid wetlands system: vertical SSF followed by an horizontal SSF (Kadlec and Wallace, 2008)

7.3. HYDROLOGICAL PARAMETERS

The hydrology of wetlands is considered one of the most important factors for the design and maintenance of these systems (U.S. EPA, 2000).

Important hydrologic factors include hydroperiod, hydraulic loading rate (HLR), HRT, hydraulic conductivity, infiltrative capacity, evapotranspiration (ET) and the overall water balance (W.P.C.F., 1990)

- Hydroperiod includes the depth and duration of flooding
- HLR is the WW loading on a volume per area basis (cm/d), is defined as the ratio of the volumetric flow rate to the wetland surface area.
- HRT is the average residence time of a typical molecule of water within its confines, is defined as the ratio of useable wetland water volume to the average wetland flow.
- Hydraulic conductivity is especially important in SSF wetlands. It is extremely sensitive to porosity, so when clogging of horizontal SSF occurs the conductivity will decrease. For example, if one third of the pore space is blocked, the hydraulic conductivity will decrease by a factor of ten (Kadlec and Wallace, 2008).

- Infiltrative capacity is a measure of the net water transferred through the wetland sediment (only important in FWS systems if they are not lined with an impermeable layer)
- ET is the combined water loss from a vegetated surface caused by plant transpiration and evaporation from the water surface (especially important in FWS systems)
- Wetland water budgets are dominated by surface inflows and outflows, ET, and precipitation.

The first step in the design of a CW is estimate and overall wetland water balance, because it may quantify the storage capacity. The sources of water are WW inflow, precipitation and direct runoff from the wetland catchment, while water losses may be the outlet flow, ET, infiltration and bank storage.

Wetland ET may increase the HRT by removing water, and can concentrate certain pollutants. For other component, such as BOD, an increase in the HRT may enhance the removal rate.

In FWS wetlands, it is difficult to measure specific ET rates since they may vary from those in open waters to those in fully vegetated zones. Indeed, it is assumed a rate of 70 to 75% of the pan rate (U.S. EPA, 2000). Infiltration may occur if the system is not lined, as ET, it will increase the HRT and increase the potential for removal. However, infiltration tends to diminish with time because of bed clogging.

Water depth is an important physical measure for the design, operation and maintenance of FWS CWs. The ability to vary water depth in FWSs allows the operators to manipulate wetland performances. The actual water depth at all locations is not easy to calculate due to basin bottom irregularities and plants. Operation depths are ranged from 0.15—0.6m, depending if there are emergent plants or submerged plant, which have larger depths.

The main advantage of a SSF system over a FWS system is the isolation of the WW from vectors, animals and humans. Thus, it is crucial to avoid “surfacing”. This phenomenon consists of a portion of the WW flows on the top of the media, creating all the undesirable conditions of FWS and reducing HRT. In tropical climates with extended periods of precipitation, as monsoon season, the runoff from the total catchment area that drains into the SSF has to be estimated.

7.4. REMOVAL MECHANISMS IN CONSTRUCTED WETLANDS:

CWs rely upon natural microbial, biological, physical and chemical processes to treat WW. The pollutants are removed by the interaction between water, soil, microorganisms, vegetation and even fauna.

The most important removal mechanisms in CWs are listed in table 3.

WW constituent	Removal mechanisms
Suspended Solids	Sedimentation/filtration Microbial degradation (aerobic and anaerobic)
BOD	Sedimentation (accumulation of organic matter/sludge on the sediment surface) Ammonification followed by microbial nitrification and denitrification
Nitrogen	Plant uptake Ammonia volatilization
Phosphorus	Soil sorption (adsorption-precipitation reactions with aluminium, iron, calcium and clay minerals in the soil) Plant uptake (Phosphine production)
Pathogens	Sedimentation/filtration Natural die-off UV radiation Excretion of antibiotics from roots of macrophytes

Table 3: Removal Mechanisms in CWs (Moshiri, 1993)

Wetland environments support a wide variety of bacteria, fungi, algae and macrophytes. The growth, death and decay of plant biomass are an important biogeochemical cycle in treatment wetlands and impose a seasonal cycle on many internal processes. During the growing seasonal cycle, macrophytes remove pollutants by directly assimilating them into their tissue, and providing surface and suitable environment for microorganisms to transform pollutants and reduce their concentrations (Moshiri, 1993). At the end of this season, nutrients are returned to the system after the emergent portion of the plants die back.

CWs have been used to treat a variety of WW including urban run-off, municipal, industrial, agricultural and acid mine drainage. Usually, some degree of pre- or post-treatment are required in conjunction with the wetland to meet the stream discharge or reuse requirements since it is not advisable to put raw WW into the CW (U.S. EPA, 2000).

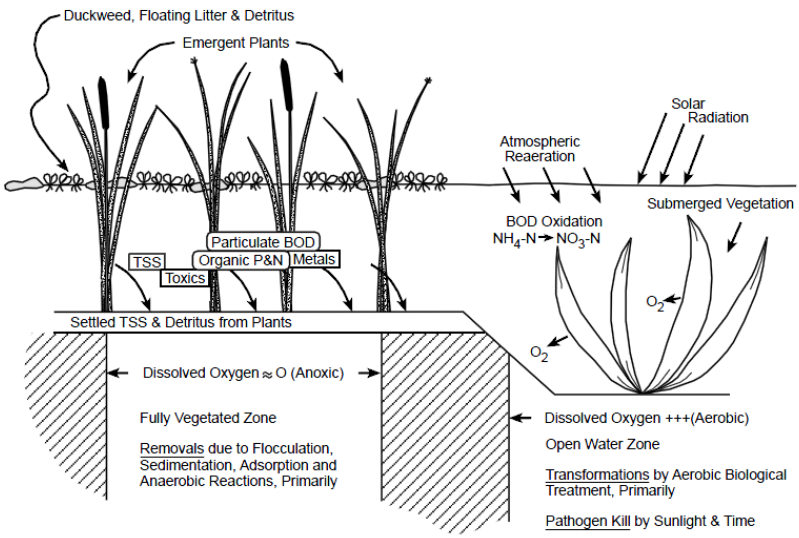

Figure 10: Removal mechanisms that dominate in FWS CWs (U.S. EPA, 2000)

Assuming that CWs are preceded by a primarily treatment, as UASB or HUSB reactor, the main removal mechanisms and performances rates are going to be discussed below.

7.4.1. Mechanisms of Suspended Solids

According to Standard Methods (1998), the TSS are defined as those solids retained on a standard glass fiber filter that typically has a nominal pore size of 1.2 µm. As a result, TSS may include settleable solids (>100 µm) to supracolloidal solids (1-100 µm). Based again on Standard Methods, solids are also classified as volatile (those that ignite at 550°C) or fixed.

The composition of these solids is varied, but UASB and primary effluents will normally contain neutral density colloidal and supracolloidal solids emanating from food waster, faecal materials, and paper products (U.S. EPA, 2000).

Low water velocities, coupled with the presence of plant litter (in FWS wetlands) or sand/gravel media (in SSF wetlands), promote settling and interception of solid materials (Kadlec and Wallace, 2008).

7.4.1.1. Suspended Solids in Free Water Surface Wetlands

TSS are removed and produced by natural processes in FWS CWs. The predominant physical mechanisms for suspended solids removal are sedimentation and interception. Whereas, suspended solids production may occur due to death of invertebrates, fragmentation and detritus from plants, formation of chemical precipitates and production of plankton and microbes in the water column.

After the suspended material reaches the wetland, it joins large amount of internally generated suspended materials, and both are transported across the wetlands. Sedimentation and trapping, and resuspension, occur en route, as does "generation" of suspended material by activities both above and below the water surface.

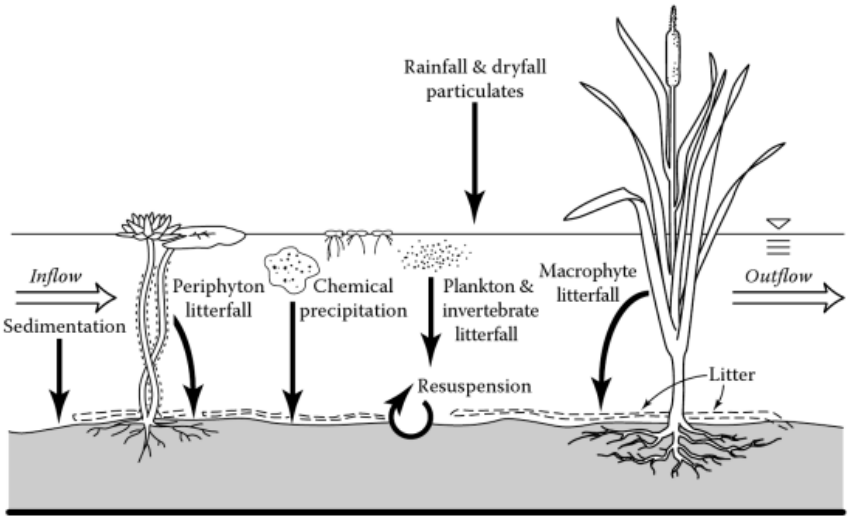

Figure 11: Processes affecting particulate matter removal and generation in FWS wetlands (Kadlec and Wallace, 2008)

In wetland systems, the larger and denser particles may be removed in the primary zone of the wetland based on simple discrete settling theory, but the smaller and neutral-density particles which are a significant fraction, are no likely to remove by this mechanisms.

Filtration is not likely to be significant in FWS wetlands, instead, interception and adhesion of particles on plant surfaces is more significant mechanism for removal. The efficiency of particle collection would depend on particle size, velocity, and characteristics of the particle and the plant surfaces that are impacted. In wetlands, plant surfaces in the water column are coated with an active biofilm of periphyton (U.S. EPA, 2000). This biofilm can adsorb colloidal and supracolloidal particles as well as absorb soluble molecules. Depending on the nature of the suspended solids, they may be metabolized and converted to soluble compounds, gases, and biomass or may just stay physically adhered (U.S. EPA, 2000).

In FWS wetlands, velocity induced resuspension is low or even minimal. Water velocities are too low to resuspend settled particles from the bottom and from plant surfaces (U.S. EPA, 2000). Furthermore, fully vegetated wetlands provide excellent stabilization of sediments by virtue of sediment detritus and root mats. However, in open water areas, wind action and the oxygen generated by algae and submerged plants, nitrogen gas from denitrification, or methane formed in anaerobic process may cause flotation of particulates (Kadlec and Knight, 1996). Owing to the heterogeneity

of the distribution of the plants, aerobic and anaerobic conditions may occur at the same but in different zones of the system.

Several chemical reactions can produce particulate matter within wetlands under the proper circumstances. Some of the more important are the oxyhydroxides of iron, calcium carbonate under aerobic conditions, and sulphites under anaerobic conditions. However, these reactions are closely linked to pH, redox and chemical composition. The iron oxyhydroxides are typically flocs with the possibility of coprecipitates. They may form under conditions of elevated dissolved ferric iron and oxygen/rich water. This set of reactions forms the basis for phosphorus removal by addition of ferric chloride to WW, and the accompanying co-precipitation of the phosphorus. Aluminium oxyhydroxides are also typically flocs with the possibility of co-precipitated. They may form under circumneutral pH conditions and do not require oxygen. This set of reactions also forms the basis for phosphorus removal (Kadlec and Wallace, 2008).

Trapped TSS, plus material generated within the wetland will result in a measurable increase in bottom sediments near the inlet structure. However, no FWS treatment wetland has yet required maintenance because of the sediment accumulation (U.S. EPA, 2000).

7.4.1.2. Suspended Solids in Subsurface Flow Wetland

The main mechanisms of TSS removal in these systems are flocculation and filtration in the granular bed. These mechanisms are relatively effective because the relatively low velocity and high surface area of the bed. These processes are enhanced by the adhesion forces between solids which tend to form bigger particles (Garcia and Corzo, 2008).

SSF systems act like gravel filters and thereby they provide opportunities for TSS separations by gravity sedimentation, physical capture, and adsorption on biomass film attached to gravel and root system (U.S. EPA, 2000).

The importance of vegetation in these systems has been debated for some time, and several studies have compared pollutant removal performance of planted and unplanted CWs and have shown no significant difference (Young et al., 2000). However, the main role of plants in SSF is to provide thermal insulation to the WW during cold weather.

In Horizontal SSFs, most of the TSS are removed at near the inlet structure, and its concentration decreases exponentially along the bed. In general, nearly all the TSS removal takes place on the 1/4-1/3 of the total length of the bed. While in Vertical SSF,

TSS retention takes place on the firsts centimetres of the bed; and it concentration decreases exponentially along the depth (Garcia and Corzo, 2008).

Reaction chemistry as noted previously for FWS wetlands can also occur in horizontal SSF wetlands. One use of horizontal wetlands has be sulphate-reducing systems to induce the precipitation of coppers, nickel and other metals (Egger, 1992).

The performance of TSS removal is consistently high, around 90% producing outlet effluent with HLRs below 20mg/L (Garcia and Corzo, 2008). Removal efficiency for TSS is also closely related to input concentration, with lower efficiencies measured at low input concentrations. The critical HRT for achieving TSS removal efficiencies above 70% appears to be about 5 days (W.P.C.F., 1990).

As mentioned above, most organic matter is removed in the inlet zone. This is the zone of the heaviest biosolids accumulation, where the greatest reductions in hydraulic conductivity occur. This zone can be termed the *biosolids clogging distance*. Clogging of the filter media is matter of concern because the bed may end up functioning, especially with high TSS loading (>50 mg/L) (Garcia and Corzo, 2008).

7.4.2. Mechanisms for Organic Matter

One of the major constituents of raw and treated WW is organic matter (W.P.C.F., 1990), its removal is quite complex because is the result of the interaction of several physics, chemicals and biological mechanisms that occur simultaneously. The diverse array of sources of organic matter make characterization difficult, the total organic carbon and volatile solids (VS) measure the total amount of organic matter, the chemically oxidizable organic matter is measured as COD, and the biodegradable organic matter is determined by the BOD (U.S. EPA, 2000). Most regulatory agencies establish WW discharge permit limits based on BOD_5 values (W.P.C.F., 1990).

7.4.2.1. Organic Matter in Free Water Surface Wetlands

The mechanisms that regulate dissolved organic matter removal in wetlands include biodegradation, sorption and photolysis (U.S. EPA, 2000). The end products will depend on the presence or absence of oxygen. Areas of the wetland populated with dense emergent macrophytes can supply only a small fraction of the needed oxygen and most of the water column is anoxic, even though small microsites containing oxygen may be found adjacent to active plant roots. On the other hand, open areas of wetlands, containing submerged plants, have aerobic conditions throughout the wetland depth.

The effluent form the primary treatment contains some particulate organic matter as well as dissolved and colloidal fraction. This influent particulate organic matter may be entrapped within biofilm attached to emergent plant surface or accumulated on the wetland floor. In addition, the organic matter deriving from dead plant may accumulate also on the floor of the wetlands. The separation of the particulate organic matter would occur by the same mechanisms as those described for TSS (U.S. EPA, 2000).

The soluble organic matter is removed by a number of separation processes, such as adsorption and absorption. This soluble material is more likely sorbed onto plant surface biofilm, and may be metabolized by organisms associated to this biofilm. The metabolic pathway and the end products of this metabolism will depend on the presence or absence of oxygen. The degree of sorption and its rate depend on the characteristics of the organic and the solid. Volatilization may also account for the loss of certain organics. However, organic matter entering a wetland receiving primary treatment will not contain significant quantities of VSs (U.S. EPA, 2000).

Biological processes change the concentration and composition of organic matter in wetlands; these reactions include oxidation/reduction processes, hydrolysis and photolysis (U.S. EPA, 2000). Organisms will consume organic matter (and inorganic matter) to sustain life and to reproduce. The organic matter in WW serves as an energy source.

Aerobic metabolism is the most efficient conversion of biodegradable materials to mineralized end products, gases, and biomass. Anoxic reactions use nitrates, carbonates, or sulphates as terminal electron acceptor, producing end products such as nitrogen oxides, free nitrogen, sulphur, etc. These reactions are typically less efficient than aerobic reactions and will not result in the reduction in BOD unless hydrogen or methane is produced (U.S. EPA, 2000).

Much of the particulate organic matter will be hydrolysed, producing lower molecular weight organic compounds that are more soluble in water. In the presence of oxygen, these compounds will be oxidized by microbes to CO_2, oxidized forms of nitrogen and sulphur and water. Under anaerobic conditions, these compounds will be converted to low molecular weight organic acids and alcohols. Under strict anaerobic conditions, methanogenesis will occur whereby these compounds are converted to gaseous end products of CH_4, CO_2 and H_2. In the presence of sulphates, sulphur-reducing microbes will convert these low weight organic compounds to CO_2 and sulphides.

It has been observed that the removal efficiency depends on the input concentration. The low efficiency at low input concentration appears to be related to the internal

production of BOD_5 in wetlands and possibly to insufficient substrate for microbes at these low concentrations (W.P.C.F., 1990). The rates of degradation are temperature dependent. Thus sediment organic matter may accumulate during the colder months and be more rapidly degraded in the spring and summer.

7.4.2.2. Organic Matter in Subsurface Flow Wetlands

In horizontal SSFs, oxygen sources will be limited to some small amount of surface aerobic and plant-mediated transport, the predominant biological mechanism is likely to be anaerobic, while in vertical wetlands it seems to be more important the aerobic degradation of organic matter (Garcia and Corzo, 2008).

Particulate organic matter is retained by filtration near the inlet structure in horizontal SSF wetlands, and near the bed surface in vertical SSFs and removed by similar mechanisms to suspended solid separation. This particulate matter will be converted to smaller particles by abiotic fragmentation, and then they can be hydrolysed by extra-cellular enzymes. These enzymes will be excreted by aerobic heterotrophs bacteria or fermentative bacteria. The smaller particles and the dissolved organic matter will be hydrolysed. Hydrolysis will generate lower molecular weight organic compounds that can be directly oxidized by the aerobic heterotrophs bacteria or fermentative bacteria (Garcia and Corzo, 2008).

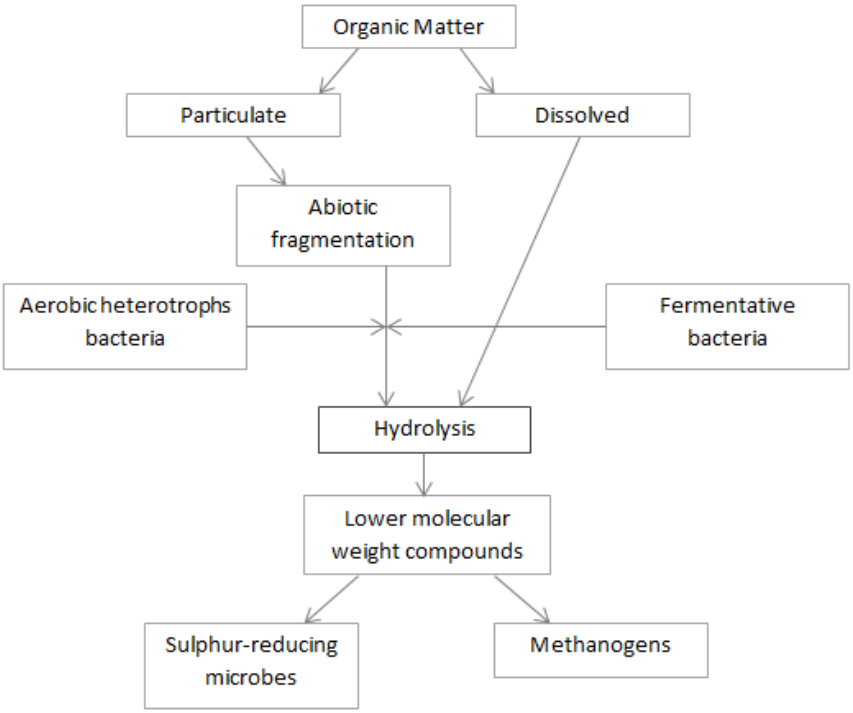

Figure 12: Simplified scheme of organic matter removal processes (Garcia and Corzo, 2008)

Moreover, it must be taken into account that some of the dissolved matter will be retained by adsorption onto the granular bed or the organic particles.

In the horizontal systems, aerobic degradation can occur near the water surface and around the roots, but the oxygen released by the roots is not enough to remove completely the organic matter. Hence the predominant metabolic pathways are most likely anaerobic. The fermentative bacteria produce fatty acids and alcohols, which are substrate for the sulphur-reducing microbes and methanogens (both anaerobic). In the vertical systems, oxygen has been found along the bed depth, what suggests that aerobic degradation is the predominant metabolic pathway since the existence of oxygen in the bed inhibits any anaerobic reaction (Garcia and Corzo, 2008).

Residual effluent from vertical systems is likely more consistent than that form the horizontal system, which shows that the aerobic conditions have better performances than anaerobic ones. Under both conditions, the removal performances vary between 75-90% producing outlet effluents with a BOD_5 concentration below 20 mg/L (Garcia and Corzo, 2008). Moreover, residual effluent from SSF systems are likely more consistent than that from the FWS system because of the present of less plant matter in the water column (U.S. EPA, 2000).

7.4.3. Mechanisms for Nitrogen

Nitrogen compounds are among the principal constituents of concern in WW because of their role in eutrophication. The wetland nitrogen cycle is very complex, and control of even the most basic chemical transformations of this element is a challenge in ecological engineering (Kadlec and Wallace, 2008).

In WW, it is common to find nitrogen, normally in ammonia and organic nitrogen form. Nitrite and nitrate forms concentration are not usually significant (Garcia and Corzo, 2008). Organic nitrogen in WW includes proteins, peptides, nucleic acids and urea; these may be found in both soluble and particulate forms, while the other nitrogen species are water soluble. NH4-N may be found in the un-ionized form, NH_3, or the ionized form NH_4^-, depending on the water temperature and pH. Due to normal wetlands conditions (pH=7 and 25°C), the ionized form is predominant (U.S. EPA, 2000).

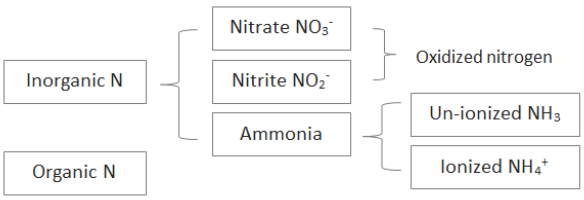

Figure 13: Common nitrogen species present in WW

The discharge of nitrogen to surface and groundwater sources is of concern for a number of reasons. Excessive accumulation of nitrogen will cause eutrophication, high concentration of ionized ammonia species are toxic to fish and other aquatic life, while nitrate and nitrite nitrogen are a public health concern (U.S. EPA, 2000). That is why is so important to evaluate the removal performances of all the species of nitrogen.

Because of toxicity of un-ionized ammonia in receiving aquatic ecosystems, this nitrogen species is often singled out for regulation. The fraction of un-ionized ammonia depends upon water temperature as well as total dissolved ammonia (Kadlec and Wallace, 2008).

Physical Separation

Even though physical separation is not the main removal process, there are a number of separation processes that will affect nitrogen species in wetlands. Organic nitrogen associated to suspended solids may be removed by the same processes described earlier for the removal of TSS. Sorption of both particulate and soluble organic nitrogen and ammonia, because the positive charge, may occur on biofilms but this is a reversible process and as soon as the local conditions change, the organic nitrogen will be released again (Garcia and Corzo, 2008). As it was mentioned before, un-ionized ammonia (NH_3) concentration is low at neutral pH, but during photosynthesis in open water zones pH may rise and its concentration too (U.S. EPA, 2000). If surface turbulence is high due wind action, un-ionized ammonia can be volatized.

Biological Separation

Almost half of the municipal WW nitrogen content is in the ammonia and organic nitrogen form. In wetlands, the main removal mechanism is a microbial process, which consists of a nitrification followed by a denitrification. However, nitrification can be also followed by plants uptake. In wetlands, the nitrogen cycle is coupled with the carbon cycle, mainly through the denitrification process. Below, these three processes are thorough described.

$$NH_4 \longrightarrow NO_2^- \longrightarrow NO_3^- \longrightarrow NO \longrightarrow N_2O \longrightarrow N_{2(g)}$$

- Nitrification

Nitrification is the principal transformation mechanism that reduces the concentration of NH4-N in the water column. In the presence of dissolved oxygen (DO), microorganisms in the water column or onto the biofilm may convert ammonia to nitrite and nitrate nitrogen in a two-steps process. Ammonia oxidation is carried out by autotroph bacteria under aerobic conditions, with ammonia as the electron donor and oxygen as the electron acceptor. Nitrification process requires 4.6g of O_2 per 1g of

ammonium nitrogen to oxide it to nitrate and 7.14g of alkalinity as $CaCO_3$ is consumed, in order to maintain the proper pH.

$$(1)\ 2NH_4 + 3O_2 \longrightarrow 2NO_2^- + 2H_2O + 4H^+$$

$$(2)\ 2NO_2^- + O_2 \longrightarrow 2NO_3^-$$

Nitrate is not immobilized by soil minerals and remains in the water column or in the pore water of the sediments. It may be absorbed by plants or microbes in assimilatory nitrate reduction (converted to biomass via ammonium) or may be consumed by heterotroph bacteria and converted to nitrogen gas (denitrification).

The presence of heavy metals in the water column may inhibit completely the reaction. The optimal pH range observed for nitrification is between 7.2 and 9 (Metcalf and Eddy, 1991).

- Denitrification

Denitrification or nitrate reduction is carried out by heterotroph bacteria under anoxic conditions, with organic carbon as the electron donor and nitrate as the terminal electron acceptor (U.S. EPA, 2000). The reaction occurs in the absence of oxygen and requires an organic carbon source; the minimum carbon to nitrate-nitrogen would be about 1g C/g NO_3-N. The products of the reaction will be N_2 and N_2O gases which will exit the wetland.

$$(3)\ CH_3COOH + 2NO_3 \longrightarrow 5/2\ CO_2 + N_2 + 2OH + + 3/2H_2O$$

The process is temperature and pH dependent (U.S. EPA, 2000). A pH below 5 and an oxygen concentration above 0.3-1 mg/l in the water column may inhibit completely the reaction.

- Plant Uptake (Assimilation)

Wetlands plants can remove nitrogen by assimilating ammonia or nitrate as an important part of their metabolism, and convert it to biomass. They can reduce inorganic nitrogen forms to organic forms that are used for plant structure. During the growing season, there is a high rate of nitrogen uptake by emergent and submerged vegetation from the water and sediments. Nonetheless, during senescence, nitrogen is released to the water column, that is why is recommended lopping the vegetation before the senescence.

7.4.3.1. Nitrogen in Free Water Surface

In FWS systems, each of the previously mentioned processes occurs but in different zones. Nitrification requires DO and therefore, there are limited areas of the wetlands where oxygen is available. There may be some nitrification occurring next to plants rhizomes were oxygen leaks from the plants, as well as, in the water column near the surface in open areas. The important variable in sizing an open water zone of FWS systems for nitrification is organic and nitrogenous loading because nitrification will not start until majority of the organic compounds have been removed. Thus, nitrification in the water column would not be expected in the initial settling zone. Under anaerobic conditions and in presence of organic matter, microbes associated with biofilms or suspended in the water column may convert nitrates to nitrogen gases via denitrification. Some nitrate will also diffuse into the sediments where it is available for plant uptake or can be denitrified as well. In open water zones of FWS systems, elevated pH and water temperature may enhance the NH3-N volatilization (U.S. EPA, 2000).

Generally, designers have to be concerned with achieving removal by nitrification. This may be achieved at low loading (oxygen demand) with sequencing closed and open wetland areas (US EPA, 2000). Temperature affects both nitrification and denitrification, and performance rates can significantly decrease during the cooler months.

7.4.3.2. Nitrogen in Subsurface Flow Wetlands

As described in section *7.4.2.2 Organic Matter in Subsurface flow wetlands*, depending on the direction of the flow, the local conditions differ. In vertical wetlands, aerobic conditions prevail while in horizontal wetlands anaerobic conditions are more common.

In vertical systems water flows pulsed. This operational regime provides alternating oxidizing and reducing conditions in the substrate, stimulating nitrification-denitrification (Moshiri, 1993). However, in practice, it has been observed that in vertical wetlands aerobic conditions are more prevalent, that is why they achieve high performances in the conversion of ammonia to nitrate. In general, the nitrification is total. Denitrification permits remove the nitrate formed during the nitrification by converting it to nitrogen gas. However, this reaction only occurs under anoxic conditions and the presence of organic matter because this reaction is carried out by heterotroph bacteria. That is the reason why vertical wetlands are usually combined with horizontal wetlands. In horizontal wetlands the oxygen transfer is low and there

are few aerobic zones, so the nitrification process is not significant. But, they achieve high performance in denitrification as long as there is enough organic matter. It has been observed, that in horizontal wetlands, the processes of nitrification and denitrification occur coupled, thus, the formed nitrate is quickly reduced and converted to nitrogen gas (Garcia and Corzo, 2008).

As described in the FWS systems, the ammonia can also be adsorbed, but this is a reversible process and as soon as, local conditions change, it will be release into water again. Plants can also remove the nitrogen by assimilating the ammonia or nitrates, and incorporate it to the biomass. But during the senescence, nitrogen can be release to the wetlands. Volatilization is not significant (Garcia and Corzo, 2008).

7.4.4. Mechanisms for Phosphorus

Phosphorus occurs in natural waters and WW mainly as phosphates. They may be in solution or particulate form. Organic phosphates are produced mainly by biological processes, while inorganic phosphates come from fertilizers.

The removal mechanisms can be biotic o abiotic. Biotic includes the assimilation by plants or microorganisms, while abiotic is mainly adsorption by the granular bed. Indeed, it is estimated that the proportion of phosphorus uptake by microflora and microfauna can be about 50% (Richardson, 1985).

Particulate phosphate is usually associated to suspended matter, then, by removing suspended solids, particulate phosphate will be removed.

The removal of phosphates is quite complex in CWs, including FWS and SSF systems. Experiences of the last three decades indicate that FWS wetlands can fulfil a useful role in phosphorus reductions in many situations. Improvements in water quality for secondary and tertiary effluents are possible, but there is perhaps an even greater role in controlling nutrients in urban and agricultural runoff. In contrast, SSF wetlands are rarely designed with phosphorus retention as a primary performance objective (Kadlec and Wallace, 2008). In general, performance rates are below 20% (Garcia and Corzo, 2008).

7.4.4.1. Phosphorus in Free Water Surface

In FWS, soluble phosphates may be sorbed onto plant biofilms in the water column, or onto the wetland sediments. The exchange of soluble phosphates between sediment pore water and the overlying water column by diffusion and sorption is the major pathway (U.S. EPA, 2000). In the sediment pore water, organic form may be precipitated as the insoluble ferric, calcium and aluminium phosphates or adsorbed

onto clay particles. The precipitation as calcium phosphates occurs at pH values above 7 and may occur within the sediment pore water or in the water column near the phytoplankton. The sorption of phosphorus on clays involves negative charged phosphates and positive charges clay, and the substitution of phosphates for silicates in the clay matrix. However, phosphate can be released from the metal complex depending on the redox potential (Eh) of the sediment. Under anoxic conditions, phosphates may also be released from ferric and aluminium phosphates by hydrolysis.

Plant uptake will only occur with dissolved inorganic phosphorus. That is why, dissolved organic phosphate and insoluble inorganics and organic phosphate may be transformed to a soluble inorganic form. This transformation may take place in the water column by suspended microbes and biofilms on the plants. Plant uptake is rapid, and following plant death, phosphorus may be quickly recycled to the water column or deposited in the sediments. Uptake by macrophytes occurs in the sediment pore water by the plant root system.

The absence of vegetation lessens removal capability (Kadlec and Wallace, 2008).

Temperature and vegetation growth patterns are two factors that may modify phosphorus uptake over the course of a year. Water temperature would be expected to modify microbial processes, which are involved in phosphorus uptake. However, the examination of wetland phosphorus removal data for warm climates shows only minimal seasonal effects.

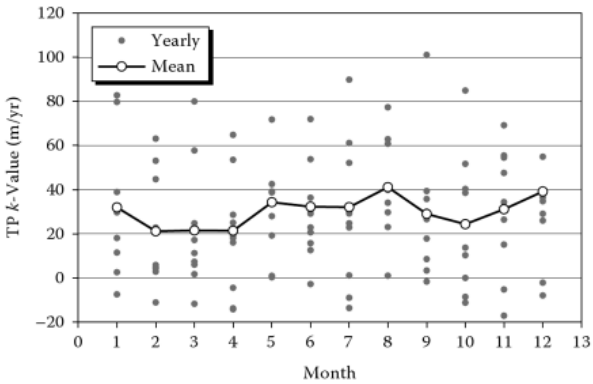

Figure 14**: Phosphorus removal rate constants for the Orlando Easterly Wetlands, Florida (Kadlec and Wallace, 2008)**

7.4.4.2. Phosphorus in Subsurface Flow Wetlands

Removal of phosphorus occurs mainly as a consequence of adsorption and precipitation with aluminium, iron, calcium and clay minerals in the bed matrix.

It has been noticed that phosphorus concentrations produced in SSF wetlands are a function of three primary variables: area, HLR and influent concentrations. However,

the sorption characteristics of the media are important factors that can be dominant mechanisms in treatment performance over the initial stage of operation until the sorption capacity is saturated (Kadlec and Wallace, 2008). At the start-up of the CWs, the phosphate removal will be high owing to the initial reaction with the soils of the wetland, but later on this rate is reduced quickly. This is because the clean granular bed has adsorption capacity but this is lost quickly.

In vertical SSF, the wet and dry periods may enhance the fixation of phosphorus in the matrix (Cooper et al., 1996).

Total phosphorus removal efficiency increases with higher input concentration and with higher HRTs (W.P.C.F., 1990).

Currently, there is a lot of research coming on to develop new mechanisms to remove phosphates, but for the moment, it seems that the best mechanism is the precipitation of phosphates by adding aluminium sales. However, using iron sales to precipitate phosphates can lead to back colour water (Garcia and Corzo, 2008).

7.4.5. Mechanisms for Pathogens

Pathogens are present in untreated domestic WW as well as in runoff waters from animal sources. Pathogens including helminths, protozoans, fungi, bacteria, and viruses are a great concern in assessing water quality. The density of these organisms in raw WW varies geographically.

In order to evaluate the removal of pathogens, indicator organisms are used. The most common indicators of level of pathogen contamination are the faecal coliforms. Faecal streptococci analysis may also be used as an additional indicator of faecal pollution.

Separation of pathogens and indicators, from the water column does not mean that the organisms are no longer viable. They may be released from the matrix to the water column and become available again. The true removal of pathogens is only achieved by making them nonviable.

The efficiencies of conventional treatment technologies that reduce pathogens have been studies thoroughly and WWTPs regularly add processes to accomplish necessary removals (Metcalf and Eddy, 1991). The most common disinfection processes are chlorination, ozonation and ultraviolet irradiation. Meanwhile, natural treatment technologies have the potential to reduce populations of enteric pathogens because of natural die-off rates and hostile environmental conditions. Wetlands have been found to reduce pathogen populations with varying but significant degrees of effectiveness.

The removal of microorganisms is a complex process because depends on many processes such as filtration, adsorption and predation. Pathogens (and indicators) may be present in suspended o dissolved form. Suspended form would be separated from the water column by the same mechanisms of TSS (sedimentation, interception or sorption). As intestinal organisms, they will normally require a rich substrate and high temperature, and some of them will die in this competition with the other organisms. They will also be destroyed by predation or by UV irradiation. The high sunlight UV exposure is the may reason why FWSs work better than SSFs. Near open water surface, some organisms will be removed by UV irradiation (U.S. EPA, 2000).

UV radiation is a potent agent for killing bacteria in open surface FWS wetlands, however, the fraction of the incoming solar radiation that is in the UV range is small (Kadlec and Wallace, 2008). Solar disinfection depends on the sunlight reaching and penetrating into the water column. Dense vegetation intercepts sunlight.

Most pathogens are food for nematodes, rotifers and protozoa (Decamp and Warren, 1998). Among these, rotifers and protozoa have been implicated as important contributors to the reduction of bacteria in treatment wetlands. However, it is complex to predict the concentrations of these predators in the wetlands waters and the quantification of this removal mechanism.

Settling and filtration also play important roles in pathogen reduction. In wetlands, submersed plant parts and their associated biofilms form “sticky traps” for particles, including all sizes of microorganisms. There may be an optimal plant density that allows light and provides the necessary surfaces for biofilm growth.

Generally, faecal coliform removal in SSF wetlands is enhanced under the following conditions:

- Longer nominal HRT or lower HLR
- Finer bed materials
- Warmer water temperatures
- Shallower bed depths

Moreover, the presence of plants has a beneficial effect on pathogens reduction. It is not clear whether this is due to the greater surface area for biofilm development or because they provide a habitat for microorganisms which may be predators for pathogens. The degree of removal in horizontal and vertical is similar and ranges between 1 and 2 logarithmic units/100 ml approximately (Garcia and Corzo, 2008).

It is important to emphasize that CWs by themselves, are unlikely to consistently meet effluent faecal coliform permit level. To produce effluents suitable for agricultural irrigation is recommended to provide the system with WSPs or FWS wetlands, as well

as chlorination. However, to obtain drinking water is required a disinfection system of the effluents prior to discharge, such as UV disinfection.

7.4.6. Mechanisms for Metals

In addition to the pollutants discussed earlier, WW typically contain many other substances than can cause problems when discharged to receiving water. These additional materials include heavy metals.

While some metals are required for plant and animal growth in trace quantities, these same metals may be toxic at higher concentrations. Other metals have no known biological role and may be toxic at even very low concentrations, (W.P.C.F., 1990). Influent WW may carry metals as soluble or insoluble species. Iron, aluminium and manganese are ubiquitous in wetlands, but they are at elevated concentrations when CWs treat acid mine drainage.

Metals entering wetlands as insoluble suspended solids are separated from the water column in a manner similar to TSS. Depending on the pH and Eh, they may be released. The most important removal mechanism for metals include, cation exchange and chelation with wetland soils and sediments, binding with humic materials, precipitation as insoluble salts and uptake by plants (U.S. EPA, 2000).

Metal	Symbol	U.S. EPA Freshwater CCC[1] (µg/l)	U.S. EPA Freshwater CMC[2] (µg/l)	U.S. EPA Human Health(µg/l)
Aluminium	Al	750	87	-
Cadmium	Cd	2	0.25	-
Chromium	Cr (IV)	16	11	-
Iron	Fe	-	1000	300
Manganese	Mn	-	-	50
Zinc	Zn	120	120	7400

[1] CCC: Criterion Continuous Concentration
[2] CMC: Criteria Maximum Concentration

Table 4: Guidelines for Metal Concentrations in WW (U.S. EPA, 2002a)

Iron is a metal that may occur at trace to high concentration. Wetlands interact strongly with iron in a number of ways, and thus are capable of significant metal removal. The three main removal mechanisms are binding to soils, sediments and particulates, precipitation as insoluble salts, principally sulphides and oxyhydroxides

and uptake by plants, including algae. Iron is not typically monitored in municipal WW wetlands; however, FWS treatment wetlands for AMD have become widely used in the United States and United Kingdom. The removal percentages are high for mine waters and leachates, but lower for waters with low iron content.

Aluminium also occurs naturally in surface water. Its precipitation is rare in natural waters but is of interest in treatment processes that rely upon addition of aluminium chlorides or sulphates for purposes of phosphorus removal. Alum or polyaluminium chloride additions are designed to form a floc of insoluble $Al(OH)_3$, which in turn adsorbs phosphorus. As mentioned in section *7.4.4 Mechanisms for Phosphorus*, the phosphorus adsorption process is a temporary mechanism, which is exhausted when the soil is saturated. That is the reason why there is a growing interest in the addition of aluminium, especially using aluminium sludge from a water treatment plant. However, in wetlands without previous coagulation, the flocs settle slowly or not at low, leaving the particulate aluminium and phosphorus in suspension, which may be removed by the same mechanisms explained in suspended solids.

7.4.7. Mechanisms for Other Contaminants

Currently, there is an increasing interest in emergent contaminants. Over the past few years due to the increasing concentration found in municipal WW, it is being carried out researching into this topic. These emergent contaminants include pharmaceutical products and personal care products, organic chemicals, and pesticides. Hydrocarbons and pesticides concentrations are caused by urban and agricultural runoff.

Pharmaceutical and personal care products (PPCPs) are an emerging class of aquatic contaminate that have been increasingly detected in natural water. There are several indications that photochemical degradation may be one of the potentially significant removal mechanisms for PPCPs in aquatic environments (Kee Ong et al., 2008). Many of PPCPs have functional groups such as aromatic rings, heteroatoms, phenol, and nitro that can absorb solar radiation.

There is considerable information on the use of treatment wetlands in the petroleum industry. The major routes for removal of hydrocarbons from wetland waters include: volatilization, photochemical oxidation, sedimentation, sorption, biological degradation and plant uptake. Three types of microbial processes can contribute: fermentation, aerobic respiration and anaerobic respiration.

BTEX constituents are volatile and may be easily lost from water, especially in shallow water bodies such as FWS wetlands. Wetland sediments typically have high organic content, and therefore sorption may be an important first step in overall removal, as well as plant uptake and biodegradation.

Polycyclic aromatic hydrocarbons (PAHs) are fused ring aromatic compounds. They are not volatile, so the main removal mechanisms are photodegradation, biodegradation and sorption. PAHs may resist biodegradation, especially as the number of fused rings increases. Low molecular weight PAHs are degraded more readily while high MW PAHs are difficult to be degraded by normal enzymatic degradation (Kee Ong et al., 2008). A variety of bacteria can degrade certain PAHs completely; however, degradation by anaerobic bacteria has not been very successful. Notwithstanding, PAHs can undergo fairly rapid transformations in aqueous solutions when exposed to UV light, implying photodegradation is an important removal process. Peat soils adsorb PAH compounds quite effectively. Wetlands have been tested for PAHs removal, and they present mixed results.

In many existing petrochemical plant applications, wetlands have been accompanied by pre-treatment. As a case of study, in a former oil refinery in United Kingdom, horizontal SSF were tested for the ability to reduce hydrocarbons, especially Diesel Range Organics. Reductions of 40-64% were achieved in less than one day's retention time. Gravel-based beds performed better than soil-based beds for this component removal (Kadlec and Wallace, 2008).

The list of pesticides is very long. A distinction may be drawn between the persistent chemicals used prior to the 1950s and the more degradable substances used since that time. Possible retention factors are adsorption to soil particles and organic matter, sedimentation of particles, photodegradation, plant uptake and biodegradation (Kadlec and Wallace, 2008).

Many of the "old" pesticides, such as DDT, are very persistent in the environment and it is doubtful that wetlands can provide any effective removal mechanisms but they can act as a trap for particulates that carry most of the load. However, modern pesticides degrade more readily and wetlands have been found to generally reduce levels of many of these compounds. Perhaps, the most commonly used pesticide in agriculture is atrazine.

The atrazine-wetland interaction is very complex, including removal by hydrolysis and sorption on wetland sediment and litter. Atrazine transport and sorption were studied at the Des Plaines River CW site, and in accompanying laboratory work. Sorption was effective for soil and sediments, but the more organic materials showed a stronger affinity for atrazine than the mineral base soils of the wetlands (Alvord and Kadlec, 1995, 1996). Atrazine was found to degrade in those sediments with a half-life of 40 to 90 days. However, degradation was faster on cattail litter, with an estimated half-life on the order of 5 days. Studies have shown that planted systems show high removal rates that unplanted systems and no large difference among plant species.

7.5. POTENTIAL HAZARD FOR MOSQUITO DEVELOPMENT

CWs provide a perfect habitat for mosquitoes that can be nuisance pests and transmit pathogens such as arboviruses and malaria (Russell , 1999). Disease transmission depends on mosquito species and abundance, and extent of contact with humans as well.

Generally, the nature of the habitat will influence which mosquito species will colonise it. Newly flooded, vegetated habitats without predators may provide suitable conditions for some pest species and produce large numbers of mosquitoes in few weeks (McDonald and Buchanan, 1981) whereas more permanently flooded habitats with established invertebrate fauna usually produce fewer mosquitoes although may mean a greater range of mosquito species (Russell, 1993).

The characteristics and siting of wetlands determine hazards, shallow water and dense vegetation promote mosquito production while open water produces fewer mosquitoes (Russell, 1999FWS CWs are of concern because densities of larval populations can greatly exceed those in natural wetlands (Tenessen, 1993). Thus, it is vitally important to plan this potential problem during the design of FWS systems. Nevertheless, SSF systems are free of this type of pest because water passes through the bed medium, flow is subsurface.

The prevention of mosquito in FWS CWs is quite complex and should be studied case by case. The way problem is set out should be a management policy instead of an eradication policy.

Ideally, FWS CWs should be allocated away from the community and beyond the flight range of local mosquito species. Shallow (<30cm) vegetated water typically supports more mosquito breeding (Russell, 1999). Steep concrete edges have shown good results in the prevention of vegetation in WSPs, but these measures are no often acceptable from an aesthetical point of view in CWs (Russell, 1999).

Water quality is another important factor. Stormwater and not heavily polluted with organic matter may create less mosquito problem that sewage or heavily pollutes WW (Schaefer et al., 1983; Carlson and Knight, 1987; Whelan, 1988; Kramer and Garcia, 1989).

Maintaining water movement through the wetlands will reduce the mosquito populations, as well as aerating, because it raises the oxygen level and improves water quality.

Vegetation provides a good environment by protecting larvae from predators and physical disturbance.

To sum up, water and vegetation management can reduce mosquitoes: aeration and sprinkler systems, flooding and drainage regimes can reduce larval densities; vegetation thinning can assist mosquito predators. Such measures may appear incompatible with the operations of wetlands, but mosquito management has to be an integral objective of CWs design and maintenance in order to reduce health hazards.

Chemical control methods can be quickly applied but they are not a good long-term strategy because the prolonged used may lead to a development of resistance in mosquito populations.

8. WASTE STABILIZATION PONDS

8.1. HISTORY AND INTRODUCTION

Ponds have been used for centuries to store and treat animals and household WW (Gloyna, 1971). However, only within the last sixty years have specific design criteria been developed in terms of volumetric requirements, organic loadings rates, and HRT.

Today, over 8,000 WSPs, comprising more than 50 percent of the WW treatment facilities in the United States are stabilizations ponds. In Europe, they are used to treat WW generated by small communities, while in New Zealand, Australia and Africa there are large pond systems (Mara, 2003).

Over the last years, they have been without doubt the most important method of sewage treatment in warm climates when required land is available and the climate conditions are favourable (Mara, 1976).

WSPs are large shallow basins in which WW is treated by entirely natural processes involving algae and bacteria (Mara, 1976). The main objective of WSPs is the reduction of the influent organic matter concentration (Von Sperling, 2007). Since these processes are not aided by man, the rate of oxidation is rather slow, thus, long HRT are required even under the favourable conditions in tropical regions (high temperatures and solar radiation intensity).

The long HRT for the removal of organic matter has an important indirect advantage: WW remains for a sufficient long period to achieve complete removal of helminthic eggs and a high removal efficiency of faecal coliform, ensuring a high hygienic quality final effluent (Von Sperling, 2007).

There are three major types of ponds: anaerobic ponds (AnP), facultative ponds (FP) and maturation ponds (MP). An anaerobic pond is essentially a digester, an aerobic pond is one in which aerobic bacteria break down the wastes and algae, through photosynthetic processes, provide sufficient oxygen to maintain an aerobic environment. Finally, the main function of the MP is to reduce the number of disease-causing microorganism, and it may also be used to rear fish.

In order to obtain a good performance and, at the same time, to minimize the HRT, various WSPs configurations are combined, and they operate in series. The first pond which receives that raw WW with high BOD_5 load may become predominantly anaerobic, that is why they are called AnP, mainly removing the organic matter by settling on the pond bottom. The AnP is followed by a partially aerobic pond, named FP and finally in order to obtain a higher quality effluent there is the MP which is predominantly aerobic.

Notwithstanding, over the last decades, a large number of pre-treatment high rate anaerobic treatment systems have been implement in order to enhance the performance and quality of the outflow, specially UASBs (Campos, 1999).

When anaerobic pre-treatment is applied, the concentrations of organic matter and TSS are drastically reduced and the removal of the residual concentrations in ponds becomes easier, shortening so much the HRT that the needed time to remove the pathogenic organisms and/or nutrients is sometimes bigger (Von Sperling, 2007). Thus, the ponds are referred to as *secondary WSPs*.

The major disadvantage of ponds is that they require much larger areas of land than others forms of WW treatment. However, in many countries, especially tropical developing countries, this is rarely a disadvantage since sufficient land is normally available at relatively low cost. Some advantages of ponds are (Mara, 1976):

- ✓ They can achieve any required degree of purification at a low cost and the minimum of maintenance by unskilled operators
- ✓ As shown in table 1, WSPs and CWs are the cheapest form of sewage treatment, including construction and O&M costs.
- ✓ The removal of pathogens is greater than that in other natural methods for WW treatment
- ✓ The effluent from a series of three ponds usually contains less than 5,000 FC/100 ml, whereas the final effluent from a conventional treatment (humus tank effluent) typically contains about 5,000,000 FC/100 ml. Cysts and ova of intestinal parasites, which are commonly present in conventional effluents, are not found in MP effluents (Mara, 1976).
- ✓ They can cope with organic and hydraulic shock loads
- ✓ Long HRT ensure that there is always enough dilution available for short shock overloads.
- ✓ They can effectively treat a wide variety of industrial and agricultural wastes
- ✓ Wastes which are biodegradable have been successfully treated. For strong waster, AnPs are used, and in the event of heavily polluted sewage, anaerobic pre-treatment is utilized.
- ✓ They can be easily re-designed so that the degree of treatment is readily altered
- ✓ By designing the pond outlet structure, the top water level can be varied. Hence, the retention time too, and the degree of treatment is altered.
- ✓ The method of construction is such that, if at some future date the land is required for some other purpose, it is easily reclaimed

- ✓ All that is required is the removal of the inlet and outlet structure and level the ground.
- ✓ The algae produced in the pond are a potential source of high-protein food, which can be conveniently exploited by fish farming.
- ✓ Fish have been successfully grown in MPs. The sale of fish can bring in substantial revenue (Mara, 1976).

8.2. CLIMATE, PHYSICAL AND BIOLOGICAL FACTORS:

- Climatic factors

Physical, chemical and biochemical reactions that occur in WSPs are highly dependent on temperature (Rodríguez, 2008). The rate of degradation increases with temperature.

The absorption of solar radiation plays a major part because it has an influence on: WW temperature, the photosynthetic activity and the removal of pathogens (CENTA, 2008).

Light is essential to photosynthetic activity. As light intensity varies along the year, the algae growing rate varies too. This phenomenon has too effects: the variation of DO and pH in the water column (U.S. EPA, 2001).

The wind has an important role because it leads the water column to be mixed, and ensures a uniform distribution of BOD, oxygen dissolved, bacteria and algae, so it enhances the degree of stabilization. The absence of mixing leads to stratification (CENTA, 2008).

- Interaction between Bacteria and Algae

In aerobic ponds, the presence of both algae and bacteria is essential for the proper functioning of the ponds (U.S. EPA, 2011). Bacteria break down the complex organic components into simple ones, which are then available for uptake by the algae (U.S. EPA, 2011). In turn; algae produce the needed oxygen for the survival of the aerobic bacteria.

- Biochemistry of the ponds

In WSPs, the carbonate buffering system has an important role. Its equilibrium is affected by the rate of algal photosynthesis. In photosynthetic metabolism, CO_2 is removed from the dissolved phase, decreasing the hydrogen ion and increasing the pH. Because of the close correlation between pH and photosynthetic activity, there is a diurnal fluctuation in pH when respiration is the dominant metabolic activity (U.S. EPA, 2011).

The intensity and spectral composition of light penetrating a pond surface affects significantly the microbial activity. In general, activity increases with the increasing light intensity until it becomes light saturated (U.S. EPA, 2011). The quality and quantity of light penetrating the pond depend on the presence of dissolved and particulate matter. The organisms as algae, contribute themselves to water turbidity, limiting the depth of light penetration.

Temperature at or near the surface determine the aquatic species. The major source of heat is solar radiation and there is a temperature gradient with depth (U.S. EPA, 2011). However, there is another heat source, which is the temperature of the influent water. In sewerage systems, the influent temperature is higher than that in the ponds. Thus, ponds may be prone to streaming, that is why is important a proper mixing (U.S. EPA, 2011).

- Pond nutritional requirements

In order to function as designed, the WW has to provide enough nutrients for the microorganism to grow and populate the system adequately. These nutrients include nitrogen, needed to algae uptake and bacterial action, phosphorus, which is most often the growth-limiting nutrient in aquatic environments, sulphur and carbon (CENTA, 2008).

8.3. DESIGNS OF STABILIZATION PONDS:

Ponds are designed to enhance the growth of natural ecosystems, and they can be anaerobic, aerobic or FP, which is a combination of the two ones.

The WSP system may comprise one pond only (FP) or several types of pond in series (AnP, FP and MP), even in parallel operation. There are many possible pond layouts.

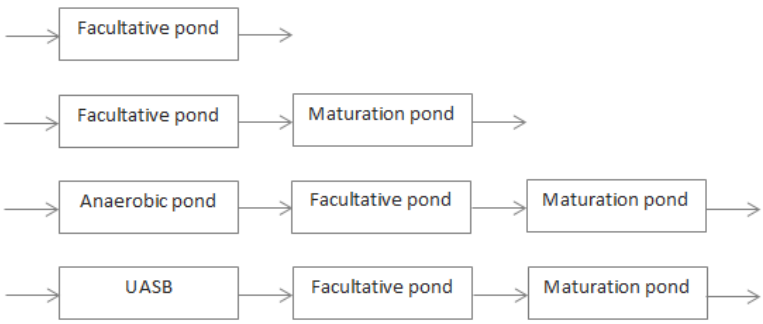

Figure 15: Typical pond layouts systems (adapted from Gloyna, 1971)

8.3.1. Anaerobic ponds (AnP)

These ponds are designed to receive such a high organic loading and they are completely devoid of DO. They are neither aerated nor mixed. They are used to pre-treat strong sewage which has high solids content. AnPs reduce the BOD load on the FP and change the nature of the settleable solids in the sludge layer, which will have a reduced fermentation potential.

Anaerobic breakdown in septic tanks, UASB, and AnPs appears to be identical.

The solids settle to the bottom where they are digested anaerobically, while the partially clarified supernatant liquor is discharged into the FP for a further treatment. During the anaerobic degradation process, there two main stages, each one carried out by a specific group of bacteria, the acid-forming and the methanogenic bacteria. The successful operation of AnP depends on the delicate balance between the acid-forming bacteria and the methanogenic bacteria. Thus, a temperature higher than 15°C is needed and the pond pH must be over 6 (Mara, 1976). Ideally, temperature should be maintained within the range of 25-40°C, and the pH value should range from 6.6 to 7.6 (U.S. EPA, 2011). Under these conditions, sludge accumulation is minimal.

Because AnPs are deep and generally have a relatively longer HRT, solids may settle, retained sludge is digested and organic matter concentration is reduced. Raw WW enters near the bottom of the pond and mixed with the active microbial mass in the sludge blanket.

- **Depth:** AnPs are usually deeper than the other type of ponds; common depths are about 2-4 meters. Higher depths prevent from atmospheric oxygen diffusion.
- **Retention times:** AnPs have detention times of 5-50 days. In the tropics, a liquid detention time of 1-5 days is recommended; longer detention may cause the upper layers of the pond to become aerobic (Gloyna, 1971).
- **Hydraulic surface loading:** is a particularly important parameter affecting sedimentation. Hence, this rate has to be lower than the settling velocity of solids and pathogens or aggregates of pathogens. So, the hydraulic surface loading should be less than 2 m/d, which is approximately the settling rate of helminthic ova (Shilton, 2005).
- **Microbiology:** anaerobic microorganisms convert organic materials into stable products, such as CO_2 and CH_4. The degradation process involves two separate but interrelated phases: acid formation, by “acid formers” bacteria, and methane production, by “methane formers” bacteria. During the acid phase, bacteria convert complex organic compounds to simple organic compounds,

mainly short-chain volatile organic acids. Then, bacteria convert the short-chain organic acids to acetate, hydrogen gas and CO_2. Finally, those components are converted into methane by methanogenic bacteria.

$$CH_3COOH \rightarrow CH_4 + CO_2$$

$$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O$$

When the system is working properly, these two phases of degradation occur simultaneously in dynamic equilibrium. However, the rate of degradation can be affected by the fluctuations of temperature and pH; even though the performance of acid-forming bacteria is the more tolerant to pH variation (U.S. EPA, 2011).

- **Loading:** "acid formers" bacteria do not cope well with shock loads.
- **Mosquito breeding:** in order to prevent the mosquito breeding, the pond must be kept free of vegetation. However, during the winter months when temperature decreases and removal too, it may appear an unsightly thick scum that promotes the fly-breeding. This problem can be overcome by increasing the maintenance (Gloyna, 1971).

In AnPs, 80-90% of BOD_5 removal can be expected, the sludge removal is rarely needed, and the energy requirements to run the plant are low or none.

The main disadvantages of these ponds are the odour that they give off, and the extra maintenance that they require. The biochemical reactions in an AnP produce hydrogen sulphide and other odorous compounds. The relationship between the odour development and organic loading is now well understood, and can be minimised at the design stage.

8.3.2. Facultative ponds (FP)

These are the most common ponds. They are usually used to treat the settle effluent from septic tanks and anaerobic pre-treatment ponds. The term "facultative" refers to a mixture of aerobic and anaerobic conditions; the aerobic conditions are maintained in the upper layers while anaerobic conditions exist towards the bottom.

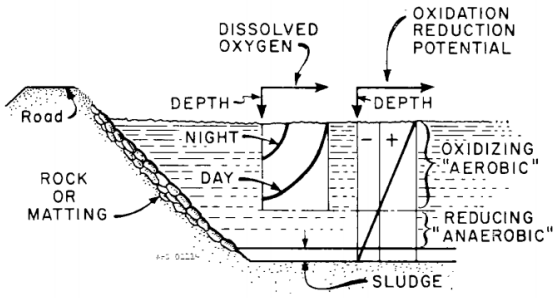

Figure 16: Typical cross-section of a FP (Gloyna, 1971)

Aerobic treatment processes in the upper layer avoid odour problems, and provide nutrients and some BOD removal. Meanwhile, anaerobic fermentation processes, such as sludge digestion, denitrification and some BOD removal take place in the lower layer.

Most of the oxygen required to keep the aerobic conditions in the upper layer, is supplied by the photosynthetic activity of the algae which grow naturally in the pond, the other amount, comes from re-aeration through the surface. Indeed, the amount of algae is so high that the ponds are green in colour. The bacteria in the pond use the oxygen produced by the algae to oxidize the organic matter. The key to successful operation of this type of pond is the O_2 production. One of the major end-products of bacterial metabolism is carbon dioxide which is used by the algae during photosynthesis since the demand for CO_2 exceeds its supply from the atmosphere. Thus, there is an association of mutual benefit, symbiosis, between the algae and bacteria (Mara, 1976).

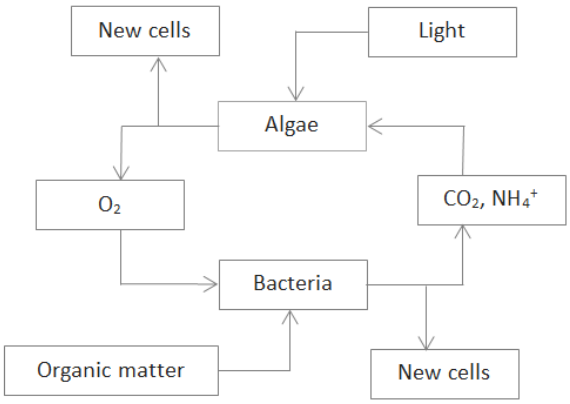

Figure 17: Symbiosis of algae and bacteria in FP and MP (Mara, 1976)

Since photosynthesis is a light-dependent activity, there is a diurnal variation in the amount of DO present in the pond and a similar fluctuation in the level of the *oxypause*, the point below the surface at which the DO concentration becomes zero, occurs. The pH also follows a daily cycle increasing with photosynthesis to a maximum which may be as high as 10. This happens because at peak demand algae remove CO_2 from the solution more rapidly than it is replaced by bacteria respiration. These high pH conditions are favourable for ammonia removal via volatilization. The O_2 in the upper layers is used by aerobic and facultative bacteria to stabilize organic matter. Anaerobic fermentation which takes place in the absence of oxygen is the dominant activity in the bottom layer of the pond.

- **Depth:** on the one hand, depths less than 1 meter may contribute to the emergence of vegetation and this must be avoided, otherwise, the pond becomes an ideal breeding ground for mosquitoes. On the other hand, with depths greater than 1.5 meters, the oxypause is too near the surface and the pond is predominantly anaerobic, which is undesirable (Mara, D., 1976).
- **Sludge layer:** as the WW enters the pond, most of the solids settle to the bottom to form a sludge layer. At temperatures >15°C, intense anaerobic digestion of the sludge solids occurs; as a result, the thickness of the sludge is rarely more than about 250 mm and often much less. Desludging is required once every 10-15 years. At temperatures >22°C the evolution of methane gas is sufficiently rapid to buoy sludge particles up to the surface, forming a mat. This must be removed, together with any other floating debris, so they do not prevent the penetration of light into the photic zone, which usually comprises only the top 150-300 mm (Mara, 1976).
- **Climatic influences:** a warm climate is ideal for pond operation. Solar radiation is intense and as a result, pond temperatures are high and there is enough intensity of light. The long daylight hours enable algal photosynthesis to occur for extended periods and so provide a reserve of DO for using during the night. However, there is usually a month of seasonal cloud cover and the light intensities are enough for algae activity, but not enough for algal and bacterial growth. That is the reason why the mean temperature of the coldest months is usually used as the design temperature (Mara, 1976).
- **Mixing:** wind and heat are the two factors which influence the degree of mixing that occurs in a pond. Mixing minimizes the stagnant regions and it ensures reasonably the uniform vertical distribution of BOD, algae and oxygen and ensures that the non-motile algae are brought into the photic zone. In tropical areas when the wind velocity is low, the differential heating is the cause of mixing. In the absence of mixing thermal stratification quickly occurs. The warm upper layers are separated from the cold lower layers by a thin static region of abrupt temperature change. The non-motile algae settle and instead of producing oxygen, they exert an oxygen demand, creating quickly anaerobic conditions (Mara, 1976).
- **Retention time:** recommended detentions vary from 5-50 days in warm climates.

The main advantages include infrequent need for sludge removal, effective removal of settleable solids, BOD_5, pathogens and faecal coliforms. They are easy to operate and require little energy. However, the main disadvantage is the higher sludge accumulation.

8.3.3. Maturation ponds (MP)

MPs are used as a following stage to FPs; they maintain DO throughout their entire depth. Their main functions are the destruction of pathogens and provide a high-quality effluent. They have shown to be one of the processes more efficient in the pathogens destruction (Yánez, 1995).

The principal factor in the design of MP is detention time, but for efficient reduction of the pathogens it is essential that the pond is arranged in series (Gloyna, 1971). Faecal bacteria and viruses die off reasonably quickly owing to what to the inhospitable environment. The cyst and ova of intestinal parasites have a relative density, and as a result of the long retention times they settle to the bottom of the pond where they eventually die. The removal of BOD_5 in MPs is low (Mara, 1976).

- **Depth:** MPs are wholly aerobic and are able to maintain aerobic conditions at depths from 0.3 meters up to 3 meters. However, the depth of MPs and FPs are the same, around 1-1.5 meters. This is because of the destruction of viruses is better in shallow ponds than in deep ones because light penetrates better. But on the other side, they are not too shallow in order to prevent aquatic plant colonization.
- **Mixing:** it is often provided, keeping algae at the surface to maintain the maximum rates of photosynthesis and O_2 production and supplying added nutrients to the surface.
- **Retention times:** detention time is typically 2-6 days.

These ponds are appropriated for treatment in warm, sunny climates; mainly because they are used to destroy pathogens by UV radiation. However, the effluent will contain high TSS unless the algae are removed.

The retention time, as well as the number of ponds, is determined primarily by the degree of bacterial purification required. The effectiveness of MPs in removing pathogens is conveniently assessed by the removal of faecal coliforms. With a proper design, rates of removal achieved may be greater than 99.99 per cent (Mara, 1976). In order to produce an effluent with a BOD_5<25 mg/l, it has been found that two MPs in series, each with a retention time of 5-7 days are required (assuming that the FPs effluent is less than about 75 mg/l) (Mara, 1976).

8.4. REMOVAL MECHANISMS IN STABILIZATION PONDS:

8.4.1. Mechanisms for Suspended Solids

TSS present in the column water can be organic or inorganic mater. A fraction of these particles may settle out by their own weight, while other may settle because microorganisms may adhere to the surface of particles forming flocs that will settle out. This will lead progressively to an accumulation of sludge on the bottom of the ponds, susceptible to be degraded by microorganism than live within it.

However, in aerobic ponds due to the symbiosis of algae and bacteria, some of the organic matter present in the water may be assimilated by the algae, increasing the content of suspended solids.

In FPs, the removal rate of TSS varies over the year, being really low or even negative during the season of mass growth of algae (spring and summer).

In MPs, the presence of protozoa and small crustaceous contributes slightly to the removal of particulate organic matter (CENTA, 2008).

The occasional high concentration of TSS in the final effluent can be the major operational challenge for pond systems. The solids are composed primarily by algae and other pond detritus, not WW solids. These high concentrations usually occur during summer. In order to remove this TSS, different methods have been used, such as intermittent sand filters, recirculating sand filters, rock filters, coagulation-flocculation and dissolved air flotation.

Because of this dissertation is focused on natural treatment systems which require low or even no energy. It would be recommendable to use the intermittent sand filters which have demonstrate their capability of polishing pond effluents at a relatively low cost (U.S. EPA, 2011). Intermittent sand filters are similar to the practice of slow sand filtration in potable water treatment, as the effluent passes through the bed, TSS and other organic matter are removed through a combination of physical straining and biological degradation processes. The accumulation of matter finally clogs the surface of the filter and prevents effective infiltration. At that time, the bed is taken out of service and cleaned. The typical HLRs range from 0.37-0.56$m^3/m^2/d$, but could be lower is the TSS concentration exceeds 50mg/l. Algae removal is almost totally a function of the sand size used.

Algal TSS may be used as a nutrient for use in agriculture or as a feed supplement (Grölund, 2002).

8.4.2. Mechanisms for Organic Matter

The main mechanisms for organic matter removal are biological processes, both aerobic and anaerobic oxidation, but the sedimentation of TSS is also a way to remove organic matter.

The aerobic process requires a continuous supply of free DO and is the most efficient method for reducing the organic content of dilute liquid wastes (Gloyna, 1971). Under aerobic conditions, aerobic micro-organisms have the ability to synthesize new cell material from wastes containing complex organic compounds. Thus, some of the organic matter is used to produce protoplasm and some of the waste is degraded into low-energy compounds. Oxygen must be supplied constantly because is the terminal electron acceptor of the redox process.

The anaerobic digestion occurs through three phases: hydrolysis, acid-forming and methane-forming; as it was explained in section *8.3.1 Anaerobic ponds (AnP).* First, the complex organic compounds are degraded into smaller compounds by hydrolysis, then, the facultative heterotrophs degrade organic matter into fatty acids, aldehydes and alcohols; and then the methane bacteria convert the intermediate products to methane, ammonia, carbon dioxide and hydrogen.

Approximately 50-70% of the solids in municipal WW are readily settable. These solids typically contain 25-40% of the BOD_5 load (Metcalf and Eddy, 2003). Hence, sedimentation of suspended solids may represent an important mechanism to remove the organic matter bound to the solids, providing a BOD_5 mass removal up to 75-80% (Rodríguez, 2008).

AnPs act in several capacities. Firstly, they provide an adequate detention time for “primary” settlement of solids. Secondly, these settle solids are anaerobically digested in the sludge layer at the bottom of the pond. And lastly, in addition to sludge digestion, these ponds may also provide some anaerobic biological degradation of the fine solids that remain suspended and the dissolved organic matter (Shilton, 2005).

As organic matter enters the FP, the settleable colloidal matter settles to the bottom to form a sludge layer where organic matter is decomposed anaerobically. The remainder dissolves organic matter is absorbed and consumed by aerobic and facultative bacteria. Fine suspended organic matter is also hydrolysed and consumed by such bacteria (Shilton, 2005). The key contribution of algae to this treatment is the production of oxygen to maintain the aerobic conditions. As bacteria breakdown organic matter, they release nitrogen and phosphorus compounds and carbon dioxide which the algae consume.

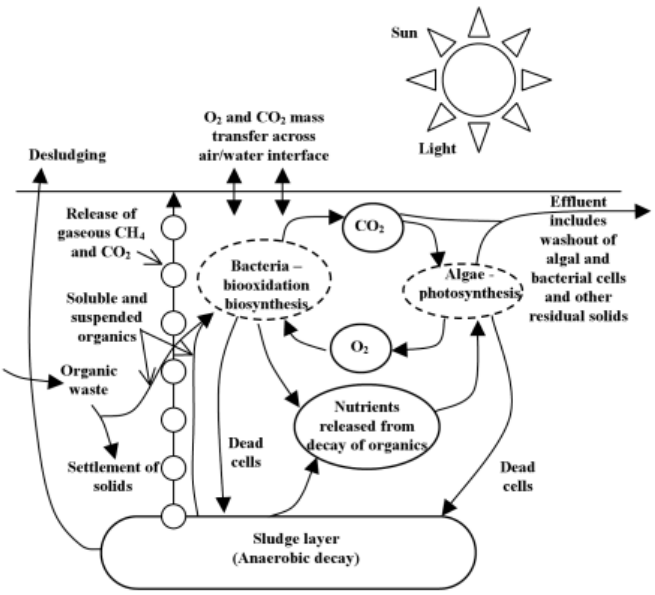

Figure 18: Basic biological interacions in a FP with emphasis on solids and organics transformations (Shilton, A., 2005)

MPS provide a low removal of organic matter and solids. Actually BOD_5 and suspended solids may increase across the MP due to algal growth (Shilton, 2005).

It has been reported a BOD_5 reduction ranging from 50 up to 95%. A very rapid reduction occurs during the first five to seven days, and they are generally much lower during winter and early spring, because the weather conditions (U.S. EPA, 2011).

8.4.3. Mechanisms for Nitrogen

Discharge of WSP effluent containing high concentrations of nutrients can have several adverse impacts on receiving water:

- Elevated nitrogen and phosphorus concentration may cause eutrophication and proliferation of nuisance plants.
- Unionized ammonia NH_3, even at low concentrations, is potentially toxic to fish and other aquatic life.

Until the early 1980's there was no agreement on the removal mechanisms. The following processes where thought to be responsible: gaseous NH_3 stripping to the atmosphere, NH_4^+ assimilation in algal biomass, NO_3^- uptake by floating vascular plants and algae, biological nitrification-denitrification, adsorption to bottom sludge and sedimentation of insoluble organic N. Research suggests that a combination of factors may be responsible, with the dominant mechanism under favourable conditions being volatilization losses to the atmosphere (Middlebrooks, 1999).

The removal of nitrogen takes place mainly in the FPs, because the anaerobic and aerobic conditions and the presence of algae.

The dominant forms of nitrogen are organic N, NH_3 and ammonium ions (NH_4^+).

Nitrogen removal in FPs can occur through the following three processes: sedimentation, ammonia assimilation in algal biomass and bacteria, and volatilization.

Organic N may be removed from the incoming WW through sedimentation of WW solids, or converted into ammoniacal-N (NH_3-N) by microbial activity, which then may be removed by algae assimilation.

Another mechanism of nutrients removal is algal/bacterial assimilation. As mentioned before, there is a symbiotic relationship between algae and bacteria. Bacteria metabolise organic waste for growth and energy, producing new bacterial biomass and releasing CO_2 and inorganic nutrients. Algae then utilise the CO_2 through photosynthesis, assimilating the nutrients into algal biomass and releasing O2. Assimilation of nutrients into algal and bacterial biomass depends on the cell density, growth rate and composition, and is affected by organic load, nutrient concentration, retention time and WW physical characteristics (Shilton, 2005). Algal growth is unaffected by inorganic nitrogen source. However, nitrate and nitrite must be reduced to ammonia-N before assimilation, as only unionized ammonia can be assimilated by algae (Abeliovich and Azov, 1976; Chevalier and de la Noüe, 1985; Shilton, 2005).

Ammoniacal-N may be lost through the WSP surface through volatilisation of ammonia gas. The rate of gaseous ammonia losses to the atmosphere depends mainly on the pH value, temperature surface to volume ration and the mixing conditions. Ammonia volatilisation can be a dominant process of nitrogen removal in WSPs reaching a mass removal about 75-98% of total N if the pH ranges at 7 to 9 and temperature ranges from 22°C to 28°C (Pano and Middlebrooks, 1982; Somiya and Fujii, 1984; Reed, 1985; Pearson et al., 1996; Shilton, 2005).

Ammoniacal-N may be also removed by adsorption to pond sludge.

Nitrification-denitrification process does not play a major role in nitrogen removal (Pano and Middlebrooks, 1982; Reed, 1985; Mara, 1997; Shilton, 2005). Nitrification is inhibited by high levels of solar-UV and bacteria grow better when attached to plants than suspended in the water column. Thus, the inconsistency of nitrification may be attributed to the variable DO, temperature and pH, and lack of aerobic attachment surface in FPs surface water resulting in a low performance (Shilton, 2005).

Under ideal conditions, up to 95% nitrogen removal can be achieved from FP (Middlebrooks, 1999).

Nitrogen removal by nitrification can be improved as much as 50% in WSPs containing biofilm (McLean et al., 2000). Incorporation of biofilm attachment surface (baffles and

geotextile supports) into the aerobic surface waters of FPs promotes the establishment of nitrifying bacteria and has been shown to enhance rates of nitrification (Craggs et al.,2000; McLean et al., 2000; Shilton, 2005).

8.4.4. Mechanisms for Phosphorus

Phosphorus is present at concentrations in municipal WW that stimulate algal growth but has to be reduced to control eutrophication. Phosphorus is present in WW as particulate and organic solids, dissolved organic matter and dissolved inorganic compounds.

Phosphates are the only form of phosphorus assimilated by algae. They can comprise up to 50-70% of the total phosphorus in domestic WW and may be released by decomposition of organic phosphorus compounds and hydrolysis of polyphosphates by phosphate enzymes (Shilton, 2005).

The main physical mechanisms are adsorption, coagulation and precipitation.

Organic phosphorus may be removed from the incoming WW through sedimentation of solids. Inorganic phosphates and ammoniacal-N may be removed by adsorption to pond sludge or, at high pH, to ferric oxyhydroxide, aluminium hydroxide and calcium carbonate crystals, which are formed in significant amounts on the surface of sludge in ponds with high Fe^{2+}, Al^{3+} and Ca^{2+} concentrations (Diaz et al., 1994; Shilton, 2005).

Phosphates may be removed through precipitation of insoluble complexes with cations, such as Ca^{2+}, Mg^{2+}, Al^{3+}, Fe^{3+}. Precipitation is dependent upon the pond pH, temperature, and phosphate and cations concentration. Phosphate precipitated at high pH during the day may be subsequently released at night when pH declines to <8 (Diaz et al., 1994).

The uptake of phosphorus by organisms in metabolic functions as well as for storage, can contribute to its removal. However, algae discharged in the final effluent may introduce organic phosphorus to receiving waters.

In AnPs, under highly anaerobic conditions small amounts of phosphine (PH_3), an odorous, toxic and combustible gas maybe produces from decomposing WW sludge by microbial metabolism (Glindemann et al., 1996; Shilton, 2005).

In MPs, both nitrogen and phosphorus removal can be enhanced by assimilation into algae biomass and elevation of the pond surface pH, which promotes both ammoniacal-N removal by volatilization and phosphate removal through precipitation with cations.

Phosphorus removal efficiency is generally lower than that of nitrogen removal and typically ranges from 40 to 50% (Shilton, 2005; CENTA, 2008).

Natural compounds containing alumina-ferric compounds, calcium and magnesium can be added to WSPs to provide cations for phosphorus precipitation (Sakadevan and Bavor, 1998; Shilton, A., 2005). Hence, P removal could be improved by increasing the water hardness, and the use of flocculants and polyelectrolytes. However, these applications increase considerably the operation costs. There may be a need to develop a low-cost technology to remove phosphorus content.

8.4.5. Mechanisms for Pathogens

WSPs are remarkably efficient and effective at removing a great variety of pathogenic organisms (Shilton, 2005). A large number of factors may influence this process, and these factors are summarised in table 5.

Factor	Likely mechanisms	Micro-organisms affected[1]	Ponds where active[2]
Temperature	Affects rates of removal processes	B, V, P, H	A, F, M
HRT	Affects extent of removal	B, V, P, H	A, F, M
Algal toxins	Algal exudates are toxic to certain bacteria	Mainly B	F, M
Sedimentation	Settlement of infectious agent (e.g. ova,, cysts) or settlement of suspended solids and the attached pathogens	H P,H	A, F, M A, F, M
Biological disinfection	Ingestion by higher organisms	B,V	F, M
Sunlight	DNA damage by solar UV radiation or photo-oxidation	B	F, M

[1] Micro-organisms: B – bacteria, V – viruses, P – protozoan parasites, H – helminth worms
[2] Ponds: A – anaerobic, F – facultative, M – maturation

Table 5: Factors that have been proposed to cause or influence disinfection in WSPs (Shilton, 2005)

Temperature itself is only lethal to micro-organisms at high values above 45°C, so it should be considered as a secondary factor. HRT should be regarded, as temperature, as a secondary factor (Shilton, 2005).

Some researchers have suggested a contribution to disinfection by certain algae in WSPs that produce extracellular materials toxic to faecal bacteria. *Otfdou et al.* (2001) reported that cyanobacteria occurring in ponds were toxic to *E. coli, Salmonella* and a number of other bacteria.

Sedimentation is believed to be the dominant mechanism for the removal of helminthic ova (Maynard et al., 1999). Bacteria and viruses may also be removed by sedimentation if sorbed onto settleable solid. However, cysts and eggs can survive for long periods in pond sludge so any sludge disturbance may be expected to release these pathogens (Maynard et al., 1999).

Predation is another removal process that has to be taken into account. WSPs are inhabited by a diverse range of micro-fauna that obtain nutrition by ingestion of WW solids including microbes.

A major difficulty with statistical approaches is that physic-chemical conditions in WSPs are highly variable with time, particularly on diurnal cycle, and seasonally with varying insolation. For this reason it is very difficult to statistically separate the variables and identify those causative of microbiological removal (Shilton, 2005).

Researches indicate that sunlight exposure is the single most important factor causing disinfection in WSPs (Maynard et al., 1999; Shilton, 2005). Sunlight inactivation, mainly by UV wavelengths, is rapid near the surface, but sunlight action over the water column may be diminished because of strong light-attenuation in the water column (Curtis et al., 1994). As shown in table 6, inactivation can be caused by at least three mechanisms: photo-biological DNA damage, photo-oxidative damage (primarily to DNA) and photo-oxidative damage to external structures.

Mechanism	Contributing wavelength	Absorbed by	Primary target	Oxygen dependence	Repairable
Photo-biological BDN damage	UV-B (290-320 nm(	DNA	DNA	No	Yes (bacteria)
Photo-oxidative damage	UV-B (290-320 nm) UV-A (320-400)	DNA + other cell constituents	DNA	Yes	Yes (bacteria)
Photo-oxidative damage	290-550 nm	Humic organic solids	Cell membrane	Yes	No

Table 6: Features of the three main mechanisms of sunlight disinfection (adapted from Shilton, 2005)

Mechanism 1 involves the absorption of solar UV-B (290-320 nm) by DNA causing direct damage to the DNA and preventing successful growth by the micro-organisms. This process is independent of oxygen and other conditions in the external medium.

Mechanism 2 involves the absorption of short solar UV-B and some UV-A (320-400 nm) wavelengths by cell constituents, including DNA but also other cellular constituents. The activated photosensitizers react with oxygen to form highly reactive photo-

oxidising species that, in turn, damage internal targets within the cell or viral particle. This mechanism depends on the DO in the external medium.

Mechanism 3 involves absorption of a wide range of ultra-violet and visible wavelength (400-700 nm) in sunlight by humic organic solids, causing direct damage the cell membrane. The activated photosensitizers react with oxygen to form highly reactive photo-oxidising species that, in turn, damage external targets, including the membrane of bacterial cells. This mechanism is also dependent on DO.

Davies-Colley et al. (2000; 1999) showed that enterococci (part of the faecal streptococci group) were more rapidly inactivated by sunlight (by photo-oxidative mechanism) than *E. coli*, expect at elevated pH (above about 9) under which conditions accelerated inactivation of *E. coli* make it less persistent. Generally it appears that *E. coli* are the better indicator except at very elevated pH (>9.5) when this bacteria is more rapidly removed than enterococci and some pathogens (Shilton, 2005).

Removal of infectious worm parasite eggs from domestic WW is especially important in developing countries where community infection levels are often high (Mara, 2001). Multiple-pond systems are capable of efficient removal of helminthic eggs mainly by the process of sedimentation to the sludge. Then, the sedimentation of these eggs transfers the concern from the water to the sludge. However, 100% removal efficiency is not always guaranteed.

Protozoan pathogens are persistent in the environment owing to their formation of resistant cysts. Experimental data suggests however, that, despite their environmental resistance, protozoan cysts are effectively removed within WSPs (Shilton, 2005; CENTA, 2008).

Although disinfection by WSPs is generally really good and much better than in mechanical treatment plants (George et al., 2002), the final effluent quality is still variable. Hence, some further disinfection treatment may be needed to meet the stringent standards.

8.4.6. Mechanisms for Heavy Metals:

Heavy metals may be removed from WSPs by a variety of processes, including: sedimentation of solids, adsorption to algal/bacteria biomass and bottom sludge, bioaccumulation into algal/bacteria biomass, chelation and precipitation.

Most heavy metals are associated with particulate matter and therefore, they settle out.

Adsorption of heavy metals onto the surface of algae and bacteria cell is a rapid process. Adsorption involves attraction of the positively charged metal ions to the

numerous negatively charged sites on the surface of algae and bacteria cells, what results into the displacement of divalent or monovalent cations.

Algae and bacteria are also known for their capacity to accumulate heavy metals since they are requires as essential micronutrients (Shilton, 2005). The accumulated metal ions are usually compartmentalized within the cell or converted to less toxic forms by binding or precipitation (Gadd, 1990). However, at high concentrations, they can inhibit the growth of algae and bacteria and may even cause death (Gadd, 1990).

Moreover, many algae and bacteria release extracellular secretions that act as chelating agents. These chelating agents from complexes with free heavy metals ions and, hence bioaccumulation will be reduced as well as toxicity. However, these heavy metal chelates are only stable at high pH (Gale and Wixon, 1979).

Heavy metals are most toxic in their free ionic form; therefore, toxicity decreases as pH is high due to the formation of insoluble precipitates (Rai et al., 1981). They may precipitate under both anaerobic and aerobic conditions. Under anaerobic conditions, heavy metals precipitate with sulphites. Whereas, under aerobic conditions and at high pH, heavy metal cations combine with anions such as hydroxide and phosphate (Rai et. Al, 1981; Shilton, A., 2005).

There is little information on heavy metals removal in WSPs. Most removal occurs in primary ponds, AnP or FP, and is due to sedimentation of solids to which heavy metals are sorbed (Toumi et al., 2000; Shilton, 2005).

The following table 7 summarizes the removal rates of the contaminants depending on the type of WSP.

	TSS	BOD_5	COD	N	P
AnP	50-65	40-50	40-50	5-10	0-5
FP	0-70	60-80	55-75	30-60	0-30
MP	40-80	75-85	70-80	35-80	10-60

Table 7: Removal rates of contaminants (%) at the different types of WSPs (CENTA, 2008)

8.5. ALGAE CONTROL:

Algal overgrowth is a matter for concern because it can cause the depletion of oxygen during the respiration phase, and may increase the TSS concentration in the pond final effluent.

Algal overgrowth is prevalent in the areas where there are a high number of sunny days during the year, long HRT, shallow pond depths, abundant nutrients, warm water

and sunshine. The problem is that during the night, algae and aerobic bacteria will utilize oxygen during the respiration process, potentially depleting the DO in the column water and causing incomplete treatment. However, on the other side, the high concentration of algae at the surface will reduce sunlight penetration and may slow the growth rate.

On the other side, control of algae in WW treatment ponds effluents has been a major concern throughout the history of the use of these systems. As it was mentioned in section *8.4.1 Mechanisms for Suspended Solids*, algae grow in MPs increases the TSS in the final effluent.

It has been established that few, if any, of the solids in the final pond effluent are faecal matter or material entering the pond system (U.S. EPA, 2011). This has led to a discussion about the need to removal algae from the effluent, because when algae die, settle out and decay, they do create some O_2 demand on the receiving stream.

Algae require light to grow, and as light penetration is reduced with increasing depth, so, increasing the depth of the MPs, up to 3-4 meters, the algae growth will be reduced. Without mechanical mixing, thermal stratification occurs in ponds, providing an excellent environment for algae to growth. Disturbing stratification, and reducing light transmission, will help to reduce the rate of growth (U. S. EPA, 2011).

8.6. ODOUR RELEASE AND CONTROL:

The release of offensive odours from AnPs occurs when the volumetric loading on the pond is greater than 400 g $BOD_5/m^3 \cdot d$ (Mara, 1976). Thus, even for a strong sewage (BOD_5=1 000mg/l), odour release is unlikely to be a problem when the retention time is 5 days. However, a high concentration of sulphates in the water, especially if it is agricultural or industrial waste, may cause odour problems. In this case, odour control is required, and this may be achieved by:

- Raising the pH of the pond to about 8, so most of the sulphide will exist as the odourless bisulphide ion, HCl^-.
- Recirculating the effluent from the FP or MPs to the AnP inlet in the ratio 1 to 6 (1 volume of the effluent, with higher DO, to 6 volumes of raw sewage) (Mara, D., 1976). This provides a thin aerobic layer at the surface of the AnP, which prevents odours from escaping into the air.
- A cover may also use to contain odours.

8.7. UASBs VS. ANAEROBIC PONDS:

Even though this dissertation is focused on a pond layout with an UASB as a primary treatment followed by a FP and MP, it is a matter of interest discuss if an AnP would be a better option.

A 6h UASB achieves a 70% removal of BOD, but this is also achieved by a 1d AnP at 25°C (Mara, 2003). The UASB is clearly smaller: it has only one quarter of the volume of the AnP. However, it costs more to construct a 6h UASB in reinforce concrete, even reinforced brickwork, than it costs to construct a 1d AnP (Mara, 2003). Moreover, the saving in land area is insignificant when compared with the area of the secondary FP needed to treat the anaerobic effluent and the area of the drying beds for the UASB sludge. However, the AnPs may present greater odour problems.

Hence, the choice of one or another primary treatment is not trivial. It may depend on the land available and surroundings, the design criteria, the budget and the experience of the client and duty holders.

9. WATER REUSE:

A growing world population, the unrelenting urbanization, the increasing scarcity of good quality water resources and the rising fertilizer prices are the driving forces behind the accelerating upward trend in the use of WW, excreta and greywater for agriculture and aquaculture (WHO, 2008). The principal forces driving this increased use are:

- ✓ Increasing water scarcity and stress
- ✓ Expanding population with increasing environmental pollution
- ✓ Recognition of the resource value of WW, excreta and greywater

It is estimated that within the next 50 years, more than 40% of the world's population will live in countries facing water stress or water scarcity (Hinrichsen et al., 1998). Indeed, in many cases, it is better to use WW, excreta and greywater in agriculture than to use higher-quality fresh water, because crops benefit from the nutrients they contain (WHO, 2008).

Most population growth is expected to occur in urban and periurban areas in developing countries (United Nations Population Division, 2002). The reuse of WW will be an important component of a package of coping strategies in areas affected by such change.

For agriculture use which includes irrigation of crops, sports fields and public parks, it has been established by the *recommended guidelines for unrestricted WW use in agriculture of WHO*, that the faecal coliforms concentration must be lower than 1,000 CFU/100 ml and the concentration of helminth eggs must be lower than 1 egg/litre.

Meanwhile, for aquaculture use, the faecal coliform concentration has to be lower than 1 000 CFU/100 ml and none viable eggs (WHO, 2002).

10. CASE STUDY: Influence of heavy rain episodes on removal efficiency in three stage hybrid treatment wetlands

The performance of an experimental hybrid CW pilot system was assessed during three months. Moreover, a heavy rain episode, a characteristic phenomenon of tropical climate regions, was simulated. The aim was to assess the appropriateness of this system for warm climate regions. The following sections show the description of the pilot system, as well as the results of the experiments carried out from June 2013 to September 2013**.**

10.1. DESCRIPTION OF THE PILOTE-SCALE TREATMENT WORKS:

The experimental hybrid CW pilot system belongs to the Group of Environmental Engineering and Microbiology (GEMMA). It is located at the Department of Hydraulic, Maritime and Environmental Engineering (DEHMA) of the Universitat Politècnica de Catalunya, Spain.

The pilot plant consists of:

- Preliminary treatment: raw WW tank, fine screening and stirred tank.
- Primary treatment: HUSB
- Secondary treatment: two vertical SSF CWs, and one horizontal SSF CW
- Tertiary treatment: one FWS CW

All these different elements are set up within two skids of 11 m^2 each one.

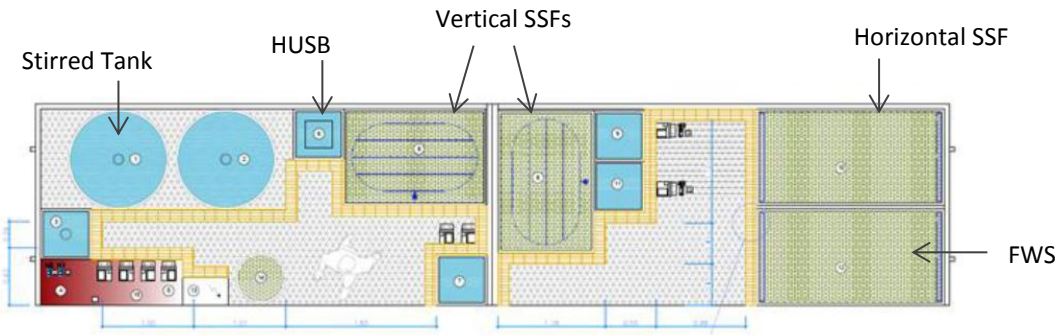

Figure 19: Top view of the pilot-scale treatment works (adapted from Donoso, 2013)

Figure 20: Studied pilot-scale treatment plant (Donoso, 2013)

10.2. DESCRIPTION OF THE TREATMENT PROCESS:

Urban WW was pumped directly from the nearest municipal sewer into the system by two pumps. Once in the system, WW was undergone a fine screening and poured into a 1.2 m^3 polyethylene stirred tank, in order to prevent any sediment settling out.

After that, WW was pumped into the HUSB by means of a peristaltic pump with an input flow rate of 800 l/d. Three flowmeters were installed at the entrance of the HUSB, the horizontal SSF and the FWS CWs. The HUSB had a nominal HRT of 5 hours for a design flow of 1200 l/d.

Once in the HUSB, the organic load of the effluent was reduced in order to enhance the performance of the CW system. The HUSB is equipped with 9 taps positioned vertically in series, starting at a height of 48 cm from the bottom and located at a distance of 20 cm from the previous one. By this distribution, it is easy to regulate the level of the mud inside the reactor. To reduce the lag phase of the microorganism in the sludge and to accelerate the stabilization of the sludge layer, the HUSB was inoculated with secondary sludge from the wastewater treatment plant of Gavà (Catalonia, Spain). Fifty litres of sludge were inoculated two times.

After that, the effluent flowed into a tank of 0.25 m^3 that regulated the amount of water pumped to the vertical SSF wetlands by means of two pumps. The two vertical CWs operated alternatively in cycles of 3.5 days; this pulsed pumping was done in order to ensure aerobic conditions within the wetland. Moreover, each one has a metal tramex plate above the floor level and a number of holes to allow passive aeration of the bed. Each pressure pump fed each of the vertical CW.

Both vertical CWs are identical, with a surface area of 1.5 m^2. They have a feeding pipe, 0.10 meters above the surface of the bed, with 5 holes with diffusers that ensure a

360° radial horizontal water pattern. The water passed through the granular bed and was recollected at the bottom where flowed into a small tanks of 0.25 m^3. This tank is covered to avoid light exposure and is necessary for the sampling of the effluent after this treatment stage.

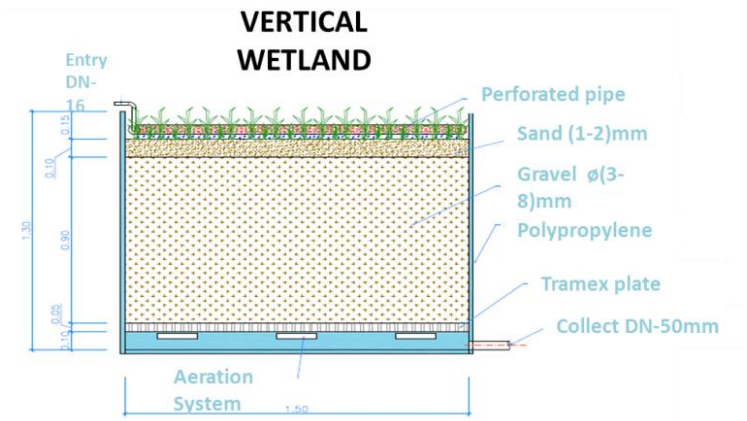

Figure 21: Cross section of the vertical SSF CW

From this tank, WW was pumped by means of a peristaltic pump to the horizontal SSF wetlands; which has a surface area of 2 m^2.

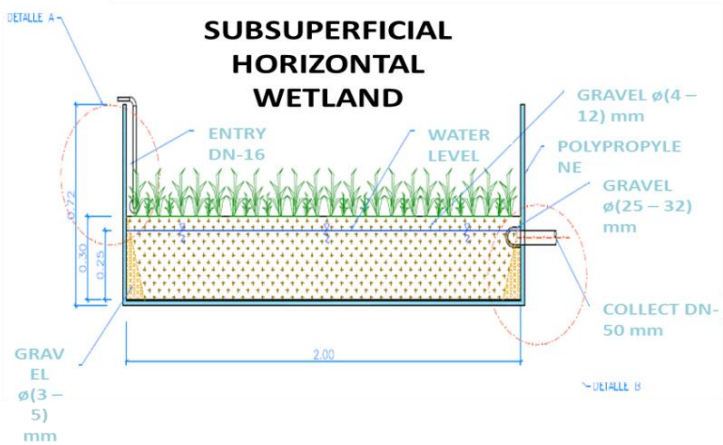

Figure 22: Cross section of the horizontal SSF CW

Finally, the horizontal SSF effluent is send to another 0.25m^3 tank that allows sampling and then pumped to the tertiary treatment which is a FWS wetland with a surface area of 2 m^3, achieving a high quality effluent ready to reuse.

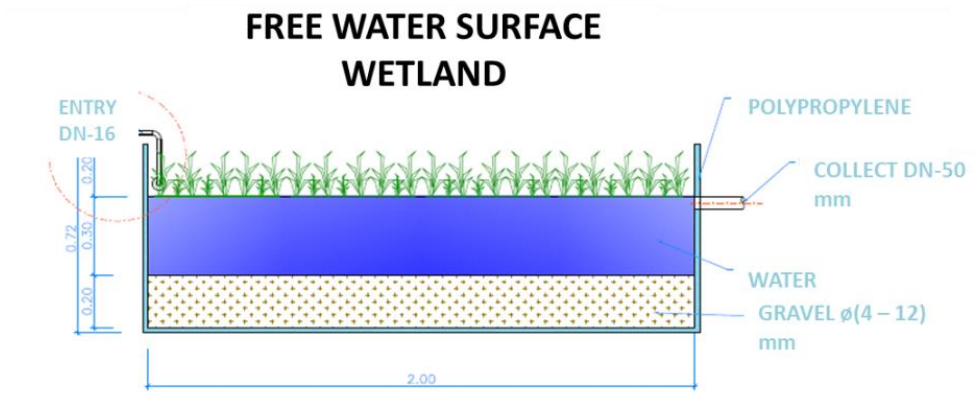

Figure 23: Cross section of the FWS CW

All CWs were planted with *Phragmites australis*. In the case of the FWS wetland, only third of the surface area was covered, in order to allow sunlight penetration and obtain mixed conditions.

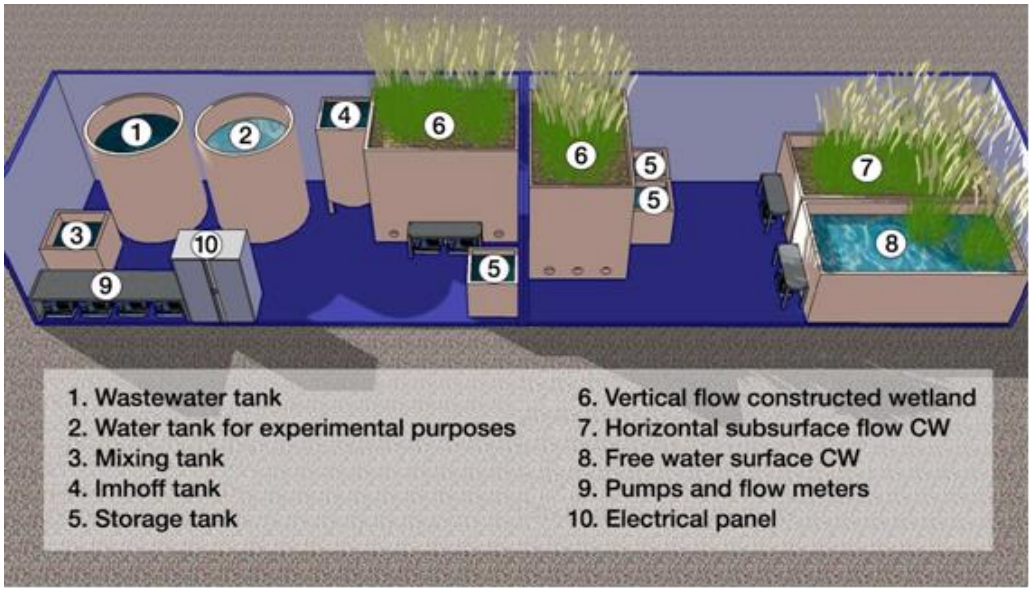

Figure 24: Hybrid CW System (Imhoff tank was remplaced by a HUSB reactor) (Avila et al., 2013)

10.3. HYDROLOGICAL PARAMETERS:

10.3.1. Under design conditions:

The plant had been operating with clean water for almost one year, but it was not until March 2013 that the whole treatment plant started to work correctly. Since then sampling campaign and analysis started.

The hybrid CWs system was evaluated under design conditions during nearly four months, the hottest months of the year, from June 2013 to September 2013.

- Influent flow: 33 l/hour
- Nominal HRT of the complete system: 24h

- Average OLR for vertical SSF: 54.7 g BOD_5 m^{-2} d^{-1} (calculated considering the surface of vertical flow wetlands (3 m^2))
- Average HLR for vertical SSF: 0.27 m/d
- Average temperature: 23.4^oC

10.3.2. Under heavy rain conditions:

Monsoons are climatological phenomena associated to the weather patterns of tropical and sub-tropical continents. Summer monsoons are large-scale sea breezes which occur when the temperature on land is significantly warmer than the temperature of the ocean, which causes a heavy rain over the land.

On September 2013, a heavy rainfall period was simulated. The WW was mixed with potable water, increasing the flow rate 10 times more than the normal influent. The treatment plant had to be adapted accordingly, and the two peristaltic pumps that feed the horizontal SSF and FWS were changed by two centrifugal pumps in order to meet the new input flow. During the Monsoon simulation, the pilot plant worked under and HLR of 330 l/h (33 litres of WW + 300 litres of potable water) during 1 h. The duration of the experiment was 10 h.

The first sampling was made just before the beginning of the storm and immediately after, and then samples were taken every 1 hour during 9 hours.

10.4. SAMPLING STRATEGY:

As mentioned before, the plant operated under an input flow of 33 l/hour and was monitored from June 2013 to September 2013. Previous assay were carried out from February 2013 to June 2013 and were compared with this last campaign in order to see any improvement in NH_4-N removal rate with higher temperatures. Grab samples were taken one a week for the analysis for the following parameters: pH, DO, Eh COD, BOD_5, TSS, NH_4-N. Sampling points are shown in Figure 25.

The sludge blanket within the HUSB reactor was sampled twice a week to ensure that VS concentration was lower than 10 g/l. Hence, the VS concentration was measured at each tap of the HUSB.

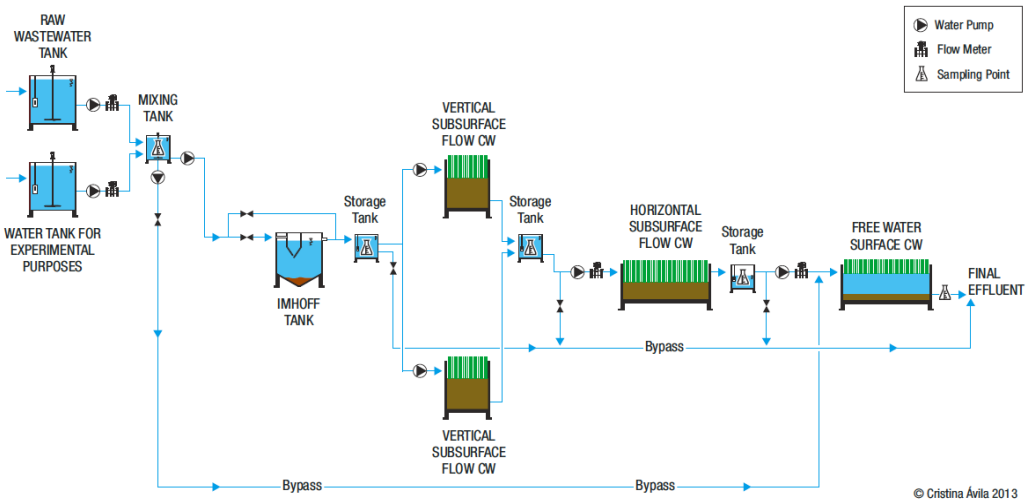

Figure 25: Plan diagram of the pilot plant with the corresponding sampling points (Imhoff was replaced by a HUSB reactor) (Avila et al., 2013)

Under the heavy rain episode, samples were taking every hour. The first sampling was made one hour before the beginning of the storm and immediately after, and then samples were taken every 1 hour during 9 hours. In total, 10 samples were grabbed during this experiment for the analysis for pH, DO, Eh, COD, TSS and NH_4-N. The other parameters could not been analysed because of the logistics.

Scope	Period	Frequency	Number of samplings	Average flow
Design Conditions	01/06/2013-19/09/2013	Weekly (ALL) Twice a week (HUSB)	7	33 l/h
Heavy Rainfall conditions	18/09/2013	Every 1h	10	330 l/h

Table 8: Sampling strategy

10.4.1. Analytical methods:

On site measurements of water temperature, DO and pH were taken by using a *Checktemp-1 Hanna thermometer*, a *Eutech Ecoscan DO6 oxymeter* and *a Crison pH-meter*, respectively. Eh was also measured in situ by using a *Thermo Orion 3 Star redox meter*. Eh values were corrected for the potential of the hydrogen electrode (Donoso, 2013).

Conventional WW quality parameters, including COD, TSS and NH_4-N were determined by using Standard Methods (APHA, 2001). BOD_5 was measured by using a *WTW® OxiTop® BOD Measuring System* and it was added and inhibitor of nitrification (2-cloro-6 (triclorometil) piridina (N-Serve), HACH Lange) (Donoso, 2013).

10.4.2. Data treatment:

Results are commonly presented as average effluent concentrations.

The results obtained in this study will be compared using average concentrations and mass removal efficiencies (Kadlec and Wallace, 2008). Mass removal efficiency is calculated as:

$$\boldsymbol{MRE = \frac{C_i Q_i * C_e Q_e}{C_i Q_i} * 100}$$

Where Ci (mgL^{-1}) is the influent concentration of a pollutant, Qi (Ld^{-1}) is the influent flow, Ce (mgL^{-1}) is the effluent concentration of a pollutant and Qe (Ld^{-1}) is the effluent flow.

10.5. RESULTS:

10.5.1. Performance of the treatment system under normal conditions:

The following section show the results of physical and chemical analysis obtained when the plant operated under an input flow of 33 l/h.

10.5.1.1. Chemical Oxygen Demand

COD concentration in the effluent of the different treatment stages is represented in figure 26. The average influent COD concentration was 226.41±100.11 mg/l, while the average concentration of the effluent was 62.43±9.56 mg/l. According to MRE equation, the COD mass removal of the entire treatment plant was 77.6%.

In figure 26, the COD concentration of each stage of the system is shown. It can be observed that COD removal occurs mostly in horizontal SSF CW, in which the influent has a concentration of 173.2±35.25 mg/l, and the effluent 69.94±14, which means a MRE of about 64.5%.

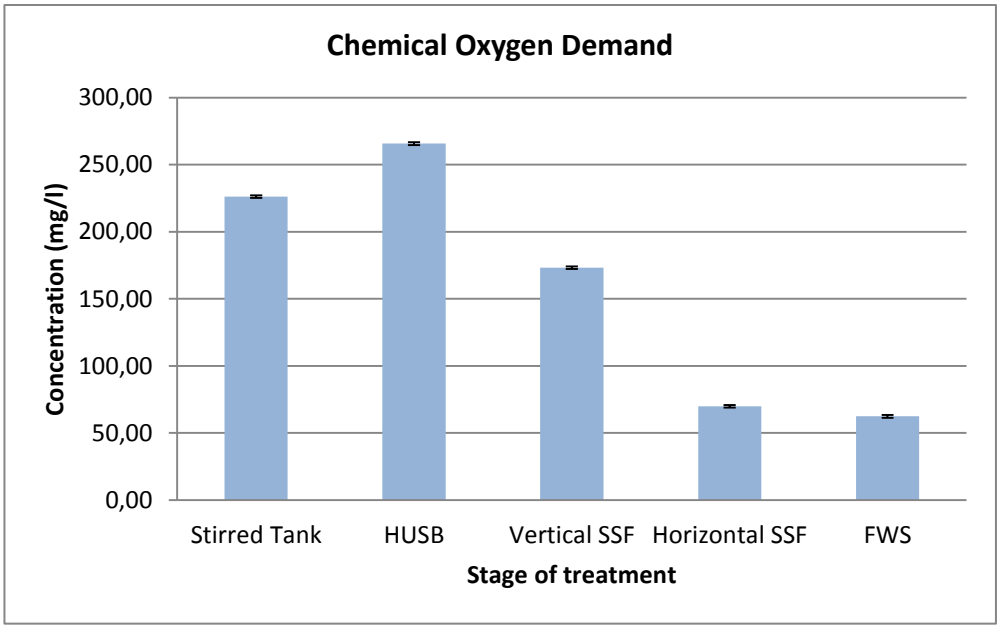

Figure 26: Average (± s.d) values of COD in the effluent of the different stages of treatment

10.5.1.2. Biochemical Oxygen Demand

In figure 27, the BOD_5 concentration on each unit of the treatment is shown. Average BOD_5 influent concentration was 154±45.06 mg/l and the average effluent concentration was 8.8±4.87 mg/l. Therefore, the removal efficiency of the entire system was 95.5%.

As shown in the figure 27, BOD mass removal occurs mainly in vertical SSF CWs, which have efficiency of 81.4%.

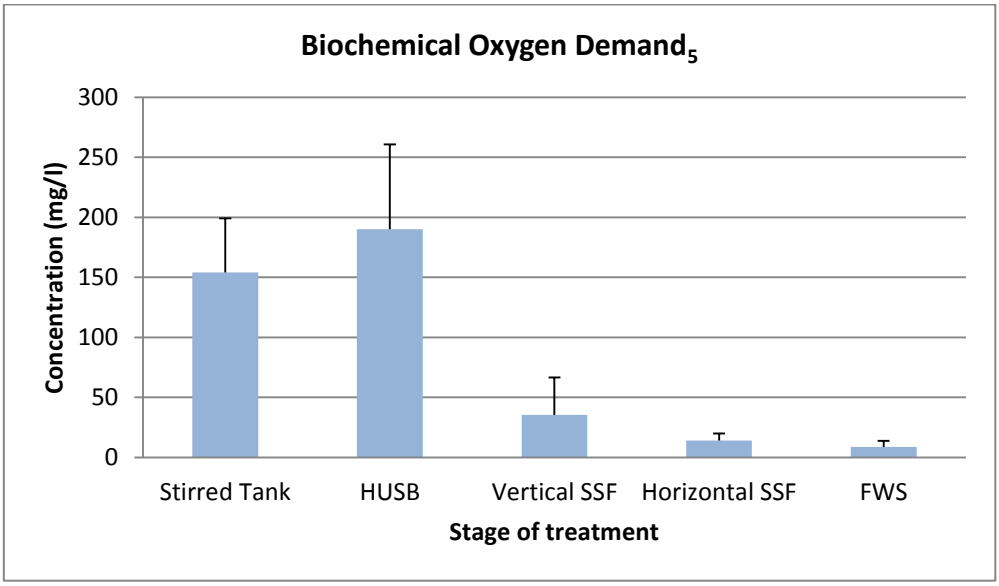

Figure 27: Average (± s.d) values of BOD_5 in the effluent of the different stages of treatment

In figure 28, COD and BOD_5 concentrations are compared. There is a noteworthy difference between removal rate of COD and BOD_5 in vertical CWs. This could be explained by the temperature conditions. In summer, high temperatures favour microbial activity, increasing the removal of organic matter by microbiological activities.

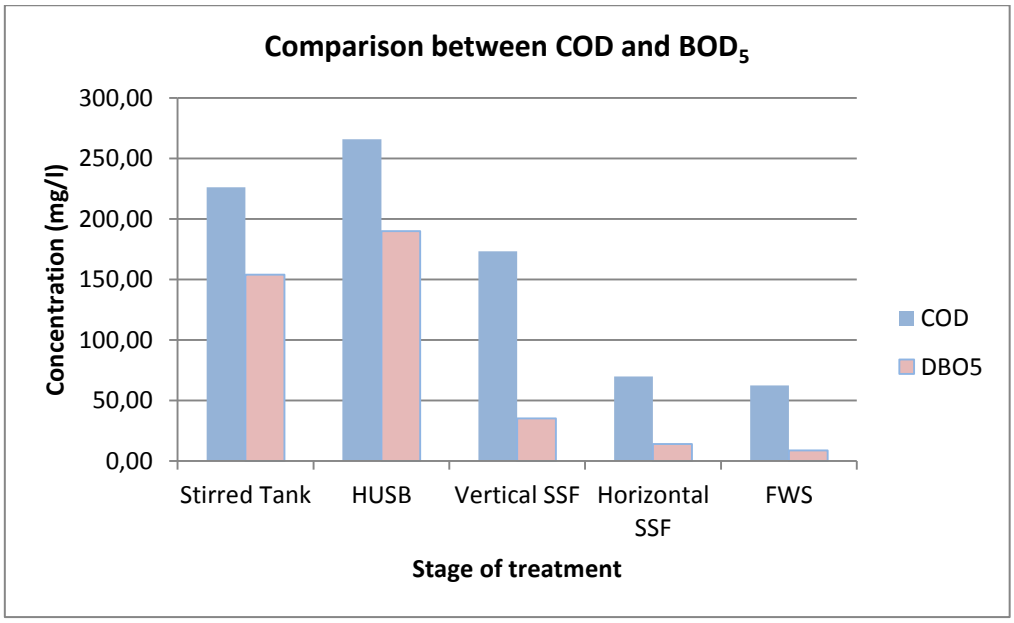

Figure 28: Comparison of the average values between COD and BOD_5 in each stage of treatment

10.5.1.3. Total Suspended Solids

Figure 29 shows values of TSS concentration in each unit of the pilot plant. Average TSS concentration in the influent was 94.95±40.65 mg/l and after the 3 wetland stages the concentration of TSS was 4.06±2.14 mg/l. Average TSS removal efficiency of the global system was 96.6%.

The increment of the TSS concentration in the HUSB comes from the sludge of the HUSB. The HUSB was fed with sludge from a WWTP in order to speed up the lag phase of the microorganism, and encourage the development of the sludge layer.

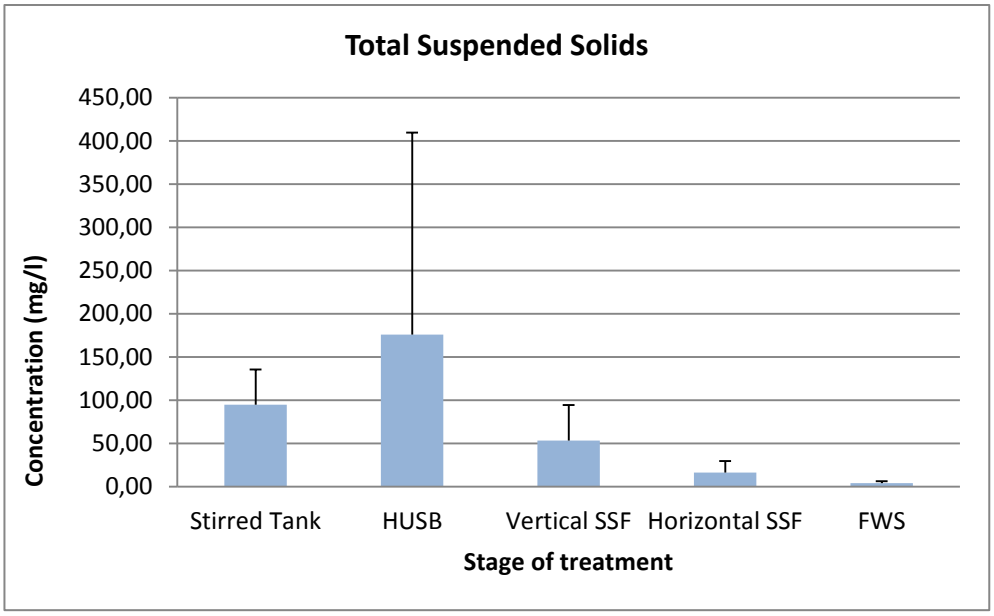

Figure 29: Average (± s.d) values of TSS in the effluent of the different stages of treatment

10.5.1.4. Ammonia-Nitrogen

Figure 30 shows the NH_4^+ concentration in the effluent of the different stages of the treatment. The influent COD concentration was 26.85±10.72 mg/l, while the final concentration of the effluent was 2.78±3.27 mg/l. Average mass removal efficiency of the treatment system for NH_4-N was 90.8%. The highest removal rate occurred in the vertical SSF CW, 67.4%.

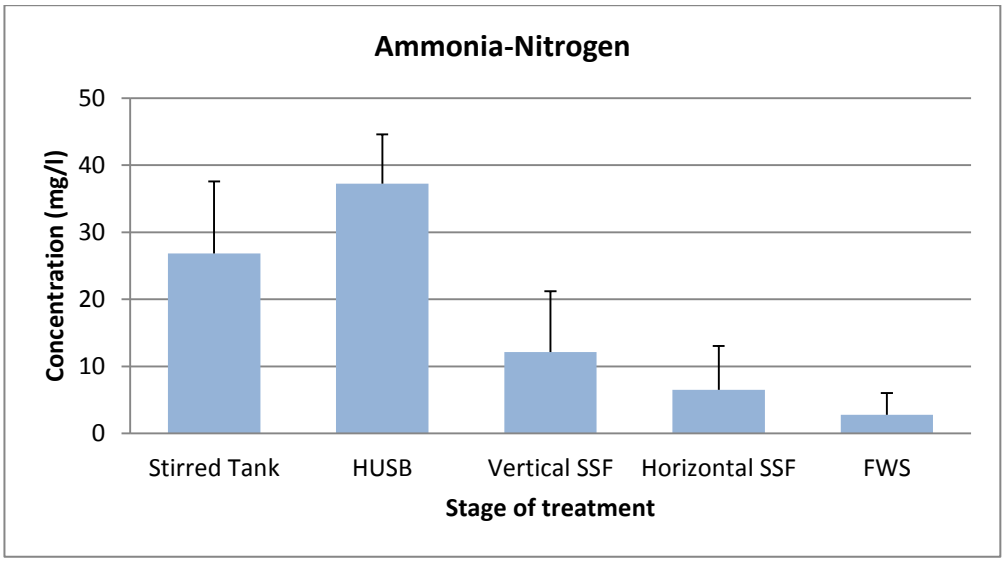

Figure 30: Average (± s.d) values of NH4-N in the effluent of the different stages of treatment

NH_4-N concentration increases in the HUSB due to the mineralization of organic matter.

One of the weaknesses of natural systems is its dependence of temperature and season conditions, especially in nitrogen removal. For this reason, it is interesting to compare the removal rates of the hottest months of the year with the removal rate of the previous campaign.

Thereby, according to the previous campaign carried out by Amigó (2013) in the same pilot-scale with the same design conditions, the removal efficiency of NH_4-N was 78% from February to March. As expected, the efficiency was higher in warm season than cold season (90.8% and 78% form warm and cold period, respectively). Figure 31, shows NH_4-N removal rate in cold season (February-March, average ambient temperature of 10ºC) and warm season (from June to September with an average temperature of 23 ºC).

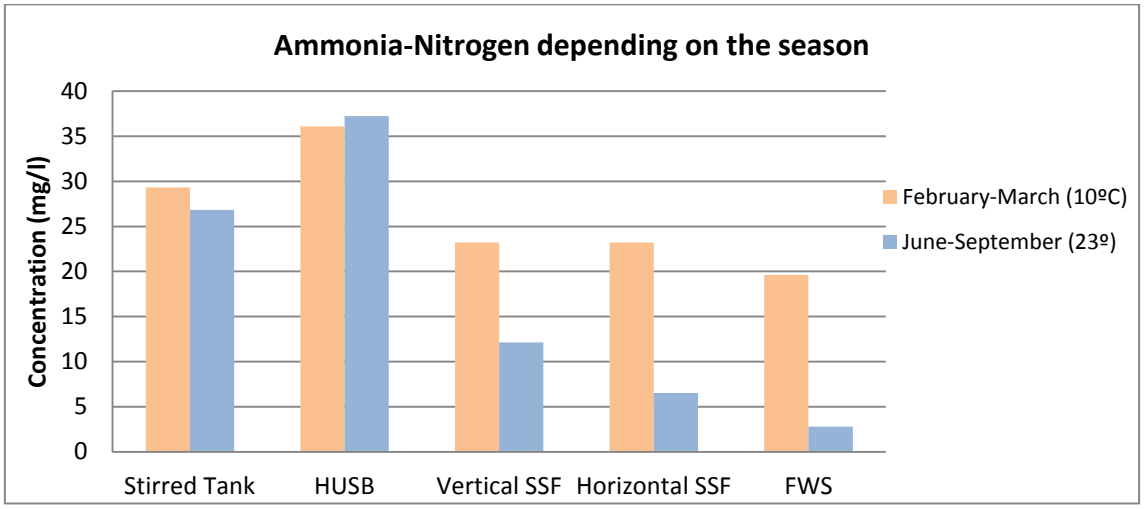

Figure 31: NH4-N concentration in cold and warm season (adapted from Amigó, 2013)

10.5.1.5. pH, Eh and DO

Table 9 shows the values of pH, Eh and DO.

	Stirred Tank	HUSB	Vertical SSF	Horizontal SSF	FWS
pH	8.09 ± 0.36	7.65 ± 0.12	7.80 ± 0.23	7.48 ± 0.36	7.39 ± 0.10
Eh (mV)	47.20 ± 33.87	- 276.64 ± 10.92	58.62 ± 33.29	62.62 ± 35.50	37.94 ± 30.30
DO (mg/l)	4.06 ± 1.05	2.13 ± 0.92	2.95 ± 1.89	1.69 ± 2.63	2.51 ± 2.53

Table 9: Average values (± s.d) of pH, Eh and DO in the effluent of the different stages of treatment

10.5.2. Under heavy rain conditions:

This section shows the results of physical and chemical analysis obtained during the heavy rain episode test, when the plant operated under an input flow of 330 l/hour.

The first sampling point was taken just before the start of the heavy rain episode simulation, so, under normal conditions. Between sample one (1h) and two (2h) the flow was increased ten times, simulating the heavy rain storm. Sample two (2h) was taken just after the storm episode simulation.

10.5.2.1. Chemical Oxygen Demand

Figure 32 shows the evolution of COD concentration in each stage during the heavy rainfall campaign.

As expected, in the stirred tank the COD decreased drastically because of the dilution of the raw WW with the income rainfall during the duration of the episode (from hour 1 to hour 2) and then, after the end of the episode (hour 2), it increased up to the normal conditions values.

In the HUSB, the process was similar; it showed a reduction of COD concentration during the first 2 hours, and remained low until hour 6 when COD concentration rose again. This turning point happened 5 hours after the rainfall episode started, which matched up with the HRT of the reactor.

The vertical SSF CWs showed a minimum in COD concentration, 50.97 mg/l, at hour 6, matching with the time the WW from the episode flowed from the CWs. The same happens in the horizontal SSF CW and FWS CW with a minimum concentration value of 7.48 mg/l and 11.44 mg/l, respectively.

The SSF CW and FWS CW showed a relatively stable concentration during the whole campaign, which indicates the robustness of the system and their capability to cope with heavy rain episodes.

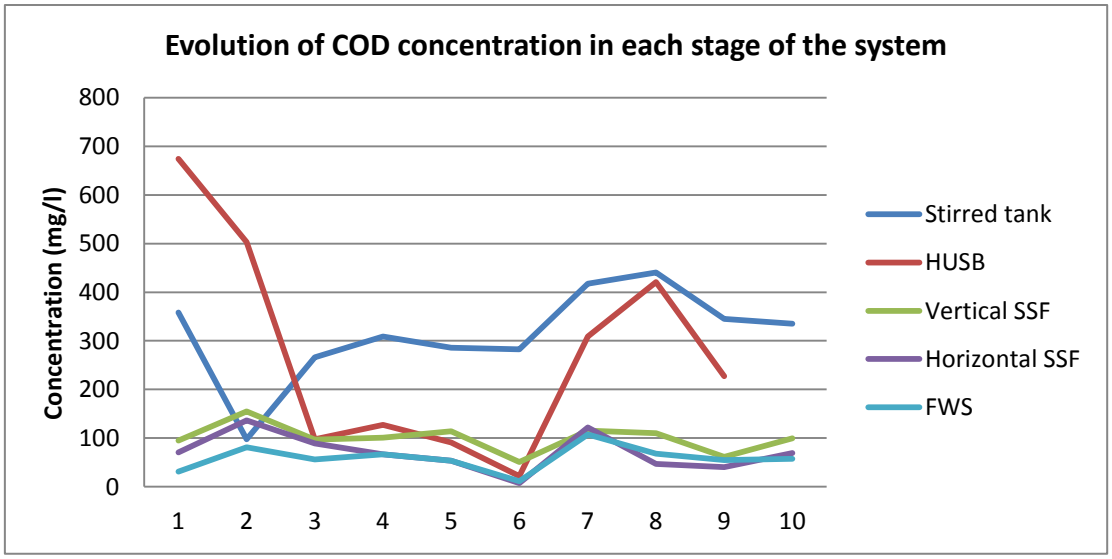

Figure 32: Evolution of the COD concentration in each stage of the system during the heavy rain campaign

10.5.2.2. Total Suspended Solids

The evolution of TSS concentration in each stage of the system during the experiment is showed in Figure 33. The evolution is really similar to COD concentration.

As expected during the rainfall episode (the interval 1-2), the TSS concentration decreased drastically in the stirred tank and HUSB. Then, the concentration in the stirred tank waved. Whereas, in the HUSB the concentration remained really low, below 25.75 mg/l, during 4 hours, increasing again at hour 6, matching up again with the HRT of the reactor.

Vertical SSF CWs showed minimum values of TSS concentration after 6 hours, matching with the time the WW from the episode flowed from the CWs. Horizontal SSF CW and FWS CW showed again constant TSS concentration, indeed, really low values, below 6.33 mg/l and 1.92 mg/l respectively.

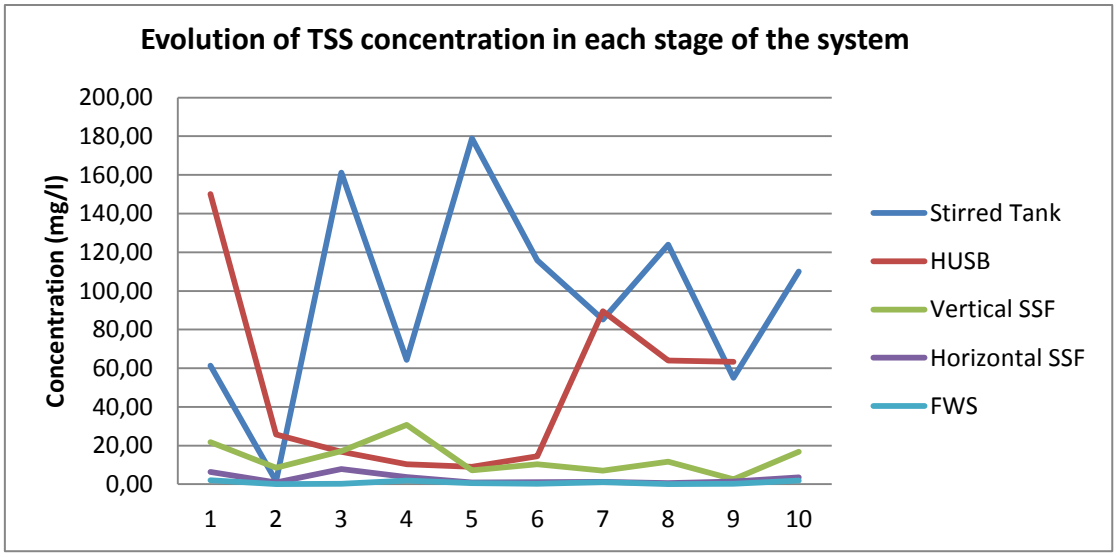

Figure 33: Evolution of the TSS concentration in each stage of the system during the heavy rain campaign

10.5.2.3. Ammonia-Nitrogen

A drastic drop of NH_4-N concentration occurred in the stirred tank and HUSB just after the rainfall episode started. Due to the dilution of the WW with the simulated rain it decreases from 22.75 mg/l to 2.45 mg/l in the stirred tank, and from 32.95 mg/l to 8.66 mg/l in the HUSB.

After the end of the storm (2h), the NH_4-N concentration in the tank returned to normal values. However, in the HUSB, values remained low during 4 hours after the start of the experiment, below 11.77 mg/l.

Both vertical and horizontal CWs showed a peak concentration about the same hour 2, 17.61 mg/l and 7.12 mg/l respectively. Then, NH4-N concentration also rose in vertical CW at hour 4, 17.03 mg/l, but after the concentration decreased gradually. Whereas in horizontal CW concentration decreased.

FWS CW showed constant concentrations during the whole campaign, below 6.66 mg/l.

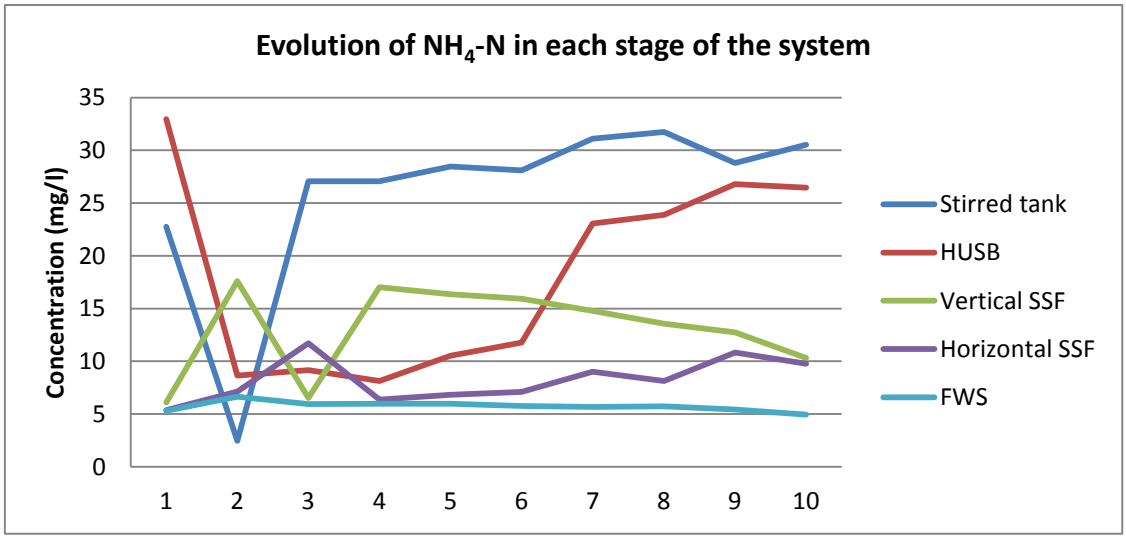

Figure 34: Evolution of the NH4-N concentration in each stage of the system during the heavy rain campaign

10.6. DISCUSSION:

10.6.1. Discussion of the results under normal conditions:

An overview of the general performance of the plant will be discussed in this section.

First of all, it is important to point out that the sewer system from which the WW was collected, was located at high income residential area with a large number of schools. Thus, there may be a great variation on WW composition especially in summer, with a great amount of WW coming from gardens and swimming pools. In short, the composition of WW could not be defined and controlled; the water quality parameters of the influent had a high variability.

In Annex I, the average values and standard deviations of the measured parameters in each stage of the treatment plant are shown.

Average influent concentrations of COD and BOD_5 were 226.24 ± 100.11 mg/l and 154 ± 45.06 mg/l respectively. These values are really low and far from the typical sewage concentrations, this could be caused by the fact that during the hottest months a great amount of WW came from garden's irrigation and swimming pools and schools from the surrounding were closed. Another matter of concern is the fact that the removal rates for COD and BOD_5 of the HUSB were -17.4% and -23.4%, so, the pre-treatment did not operate properly at all. The reason why to use a pre-treatment is to reduce the concentration of TSS, COD and BOD_5, to improve the efficiency of the secondary and tertiary treatment and prevent from clogging, but instead of that, there was an increased in concentrations. Even so, CWs showed good removal efficiency. Vertical SSF CW showed a removal efficiency of 34.8% and 81.4% for COD and BOD_5

respectively. Horizontal SSF CW provided about 64.5% and 64.7% removal of COD and BOD_5 respectively.

TSS mass removal of the HUSB was even worse, -85.34%, which is a clear warning that the HUSB is not working correctly, this mass was coming from the added sludge of the HUSB. According to Metcalf and Eddy, the normal value of TSS mass removal in physical pre-treatments may be about 50-60%. Vertical SSF CW showed a removal efficiency of 69.7% for TSS while horizontal SSF CW provided about 72.6% of TSS removal. Even so, the removal rate in FWS CW was 78.1% and the overall system reached about 96.6% of TSS removal.

NH_4-N concentration, as well as the above mentioned parameters, increased in the HUSB about 38.7%. However, the CW system showed a good removal efficiency especially the vertical SSF CW, 67.4%, and overall, the removal rate was about 90.8%. As noted before, this result was compared with the NH_4-N removal rate of a previous campaign carried out at the same pilot-scale plant during the cool season by *Amigó* (2013). As expected, the removal rate during the warm season (90.8 %) was far higher than the one obtained during the cold season (78%).

The high concentration of DO in the influent may be caused by the continuous aeration in the stirred tank before flowing to the HUSB. Once the WW was within the HUSB, DO concentration decreased because the consumption of oxygen to degrade organic matter, 2.13±0.92 mg/l, and then, as WW percolated through the bed of the vertical SSF CW, DO concentration increased, 2.95±1.89 mg/l, because the alternating aerobic and anaerobic conditions. The DO concentration in the final effluent is really low. It should be aerated before going to the receiving water. Fish needs DO≥5 mg/l.

According to studies carried out by *Vyzamal et al.* (2008) and *Barros et al.* (2008), anaerobic digester provided a COD removal about 35-65%, about 35-65% removal of BOD_5 and 50-90% removal of TSS. However, the results of this study are the complete opposite and indicate that the implementation of a HUSB reactor as a primary treatment did not enhance the treatment capacity of the system.

Despite the malfunctioning of the anaerobic pre-treatment, overall treatment efficiency range from 90.8 to 96.6% removal for NH_4-N, BOD_5 and TSS, and 77.6% removal for COD.

10.6.2. Discussion of the results under heavy rain conditions:

First of all, it has to be pointed out that the first flush and increasing OLR was not simulated due to technical limitations. Hence, the expected concentration curve of a real storm case did not occur. This curve is characterized by a peak of the concentrations occurring a little bit after the beginning of the storm, followed by a decrease of the concentrations (Avila et al., 2013a). In this case, all the water quality

parameters concentration suffered a drastic drop because of the dilution with potable water.

In Annex II, the concentration values of the measured parameters in each stage of the treatment during the heavy rain campaign are shown.

As it can be seen in Annex II, the influent COD concentration was about 300-400 mg/l. This value is quite higher than the one from normal conditions because by the time the heavy rain episode was simulated (19th of September 2013), schools were open again and people had come back from holidays or second homes. At the point 5 and 6, the vertical SSF CW showed an increase on COD concentration. This could be explained by the fact that the diluted WW was still in the HUSB, which had a nominal HRT of 5h, and the water collected in the vertical wetland was previous to the heavy rain episode. After this "breaking point" the performance of the vertical was the expected one.

The average removal rates of the treatment system for COD, TSS and NH_4-N were 83%, 99% and 80% respectively. It has to be mentioned that these values are not exact but an approximation, because to calculate the overall rate at each point it was not considered that in some units of the system the water was diluted and in others it was not, due to the different retention times of the stages. Indeed, a tracer experiment should be carried out to obtain the real HRT of each unit and for a better understanding of the results.

It was also important to study the response of the HUSB to the heavy rain episode, especially the response of the sludge. Despite of the increased flow, the HUSB did not lose much sludge. It was measured the solids concentration of the effluent the day before and the day after the test, resulting 8.88 g/l and 7.12 g/l respectively, and the following week the concentration was again stable.

To sum up, the removal rates of the treatment plant did not vary significantly from the ones obtained under normal conditions and the anaerobic digester-CW system showed a good efficiency during the experiment. The contaminants concentration seemed to return to the normal average around 7 hours after the rain episode for some units (i.e: HUSB, VF CW). On the other hand, for HF CW and FWS CW fairly constant concentrations were observed. It could be due to the higher HRT of the pilot plant compared to the duration of the experiment. As mentioned above, a tracer experiment should be carried out to obtain the real HRT of each unit and for a better understanding of the results.

Finally, the system is robust and it can handle on heavy rain episodes, which makes it a suitable water treatment engineering solution for warm climate countries.

11.CONCLUSIONS AND RECOMMENDATIONS:

The work carried out during the dissertation led to the following conclusions:

- About the literature review of natural systems:

Against a backdrop of restrictive directives in which small and rural communities have to treat their WW prior to discharge, this dissertation has tried to review and find the most suitable engineering solutions considering the environmental, aesthetic and cost aspects.

The natural systems for WW treatment presented in this dissertation are feasible solutions to treat sewage from small communities and allows to its further reuse.

Apart from a good removal rates, CW systems provide an additional value to the treatment because they allow recovering lost natural zones and ecosystems. However, FWS CWs could be prone to mosquito development.

WSPs allow to removal organic matter, nutrients and pathogens from WW with minimal O&M costs. However, the main disadvantage is the substantial increase in algae content in the final effluent, which may need a further treatment such as sand or rock filters.

Both natural systems show good removal rates for TSS, organic matter, and NH_4-N. However, phosphorus mass removal is low, and the effluent could cause eutrophication in the receiving water. Phosphorus removal could be improved by adding salts or flocculants, however this increase considerably the costs. There may be a need to develop a low-cost technology to remove phosphorus content.

The combination of anaerobic digesters and CWs or WSPs for the secondary and tertiary treatment of domestic WW is a recent and promising solution in developing countries.

- About the experiment to assess the efficiency of a three stage hybrid treatment wetlands:

The performance of pilot-scale treatment plant, which consisted of an anaerobic digester followed by a hybrid CWs system, during the warmest months of the year and its robustness under an extreme rainfall event was tested.

Under an input flow of 800 l/d, the average values of the total mass removal rates were above 77.6% for all the contaminants even though the HUSB did not work properly during the period considered.

During the heavy rainfall campaign, the total mass removal rates were even higher (above 80%). The system seemed to return to its normal average values 7 hours after

the rainfall episode for some units (i.e: HUSB, VF CW). For HF CW and FWS CW fairly constant concentrations were observed. Moreover, the sludge within the HUSB could handle on the increased flow.

In short, the system showed a very good buffer capacity under extreme rainfall events. It was proved that the system can cope with the sharp fluctuations in flow to be treated.

It can be concluded that the technology of CWs is a valid solution for WW treatment generated in small agglomerations of warm climate areas. Indeed, the experimental hybrid system showed to be highly efficient.

The study has also allowed proposing the following recommendations:

- During the design stage of FWS CWs, the potential hazard of mosquito development should be considered and minimized. That is why SSF CWs are preferred.
- In WSPs, AnPs may also be a feasible option for primary treatment, instead of an UASB. Each project is unique, that is why the choice of one or another primary treatment is not trivial and should be evaluated.
- In relation to the pilot-scale plant, it may be advisable to study the performance of the hybrid CWs system without the primary treatment. Moreover a tracer experiment should be carried out to obtain the real HRT of each unit and for a better understanding of the results obtained from the Monsoon simulation.

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