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THE USE OF CONSTRUCTED WETLANDS TO TREAT AGRICULTURAL TILE DRAINAGE

DIPLOMA THESIS

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DIPLOMA THESIS ASSIGNMENT

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Objectives of thesis

The aims of the Thesis are:

- to describe water chemistry of agricultural drainage waters
- to summarize the use of constructed wetlands to treat agricultural drainage
- to describe the wetland systems at Velký Rybník designed to treat agricultural drainage
- to evlaluate treatment efficiency of mentioned constructed wetlands with respect to nitrogen and phosphorus60

Methodology

In the first part, the review of the the use constructed wetlands to treat agricultural drainages will be prepared. In the second part, the constructed wetlands at Velký Rybník will be summarized. In the last part, the treatment performance with respect to nitrogen and phosphorus over the period of two years will be evaluated.

The proposed extent of the thesis

60 pages including appendices

Keywords

tile drainage, constructed wetlands, nitrogen, phosphorus, organics

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Recommended information sources

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DECLARATION

I hereby declare that I wrote this thesis independently under the direction of my supervisor Prof Jan Vymazal. I have listed all the literature and publications cited in this thesis.

In Prague, 29.03.2021 Lynda Odogwu

DEDICATION

This work is dedicated to God for his mercy, even in this pandemic. I also dedicate this thesis to my late Father, Tony Odogwu. I miss you.

ACKNOWLEDGEMENT

I would like to thank my supervisor Prof Jan Vymazal for his patience and understanding. I would also like to appreciate my family for their moral, financial and emotional support. My mom for her prayers and advice, my sister for her support and my brothers for being there. I would like to appreciate Dr Kumble for always having encouraging words, the project team for the data and help, my friends Anton, Bj, Jitka, Sodiq, Barnabas for their support. I am grateful. Thank you.

ABSTRAKT

V situaci globálního oteplování, změny klimatu a všeobecného znečistění životního prostředí nelze přecenit význam zkoumání způsobů, jak pomoci přírodě s překonáním znečistění a utlumit antropogenní efekty. Stoupající akumulace přírodního dusíku v životním prostředí spolu se stoupajícím množstvím antropogenního dusíku a fosforu z hnojiv vedou k rozsáhlé eutrofizaci vodních nádrží po celém světě.

V roce 2020 bylo provedeno hodnocení experimentu, probíhajícího na třech umělých mokřadech s podpovrchovým horizontálním tokem, vybudovaných pro čistění drenážních vod z povodí o rozloze 15,73 ha, z hlediska dlouhodobé účinnosti v zachycení dusíku a fosforu. Umělé mokřady byly navrženy a vybudovány v roce 2018 (Vymazal et al., 2020). Jejích monitorování a odběr vzorků byly zahájeny v roce 2018 a stále pokračují. Mokřady mají rozlohu 79, 90 a 98 m^2. Používané makrofyty jsou *Phalaris arundicacea* a *Glyceria maxima* vysázené souběžnými řádky.

Jako substrát v prvních dvou mokřadech používá se směs štěrku frakce 4-8 mm s drtí z březového dřeva v objemovém poměru 10:1. Na jednom z těchto dvou umělých mokřadů voda je udržována na úrovni 10 cm nad povrchem, na druhém – pod povrchem. Třetí mokřad skládá se z horní 20 cm vrstvy březové drti na vrstvě štěrku.

Průměrný podíl koncentrace dusíku, zachycený mokřady 1, 2 a 3 za období dvou let a čtyř měsíců, činil 56,20%, 59,60% a 61,48%. Průměrné zachycované dusíkové zatížení na umělých mokřadech 1, 2 a 3 činilo 1,92 $g/m^2 \cdot den$, 0,962 $g/m^2 \cdot den$ a 0,839 $g/m^2 \cdot den$.

Průměrný podíl celkové koncentrace fosforu, zachycený mokřady 1, 2 a 3, činil 8,27%, 1,930% a 12,21%. Dosažené výsledky ukazují na vysokou účinnost mokřadů s podpovrchovým horizontálním tokem pro dlouhodobé čistění zemědělských drenážních vod.

Klíčová slova: odvodneni dlaždic, vybudované mokrady, dusík, forfor, organické látky.

ABSTRACT

With the case of Global warming, Climate change and pollution in general, researching methods to assist nature in combating pollution and reducing the anthropogenic effects of man cannot be overemphasized. Also, the increased environmental application of natural nitrogen mixed with increased anthropogenic Nitrogen and Phosphorus from fertilizers, has led to a widespread eutrophication of water bodies around the world.

In 2020, an on-going experiment of three constructed wetlands with horizontal sub-surface flow built to treat tile drainage from a 15.73 ha watershed, was assessed to check its long-term efficiency in nitrogen and phosphorus removal. The constructed wetland sites were designed and constructed by Vymazal *et al*., 2020, in 2018. Monitoring and sample collection started in 2018 and is still ongoing. The wetlands have a surface area of 79, 90 and 98m^2 and macrophytes used are *Phalaris arundicacea* and *Glyceria maxima* planted in parallel bands.

The substrate in the first two wetlands is gravel (4-8 mm) mixed with birch woodchips (10:1 volume ratio). In one of the two constructed wetlands, the water level is kept 10cm above the surface, in the second wetland the water is kept below the surface. The third wetland has 20cm layer of birch woodchips on top of gravel. The average total nitrogen concentration removed during the two years and four months period was 56.20%, 59.60% and 61.48% for CW1, CW2 and CW3 respectively. The average load removal amount for the three CWs is 1.92g/m2d, 0.962g/m2d and 0.839g/m2d for CW1, CW2 and CW3 respectively. Also total phosphorus has an average removal concentration of 8.27%, 1.930 % and 12.21% for CW1, CW2 and CW3 respectively. The results show that the horizontal subsurface flow wetland is very efficient for long-term treatment of agricultural tile drainage.

Keywords: tile drainage, constructed wetlands, nitrogen, phosphorus, organics.

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1. INTRODUCTION

With the case of Global warming, Climate change and pollution in general, researching methods to assist nature in combating pollution and reducing the anthropogenic effects of man cannot be overemphasized. Also, the increased environmental application of natural nitrogen mixed with increased anthropogenic Nitrogen and Phosphorus from fertilizers, has led to a widespread eutrophication of water bodies around the world (Galloway *et al*., 2003; Le Moal *et al*., 2019; Smith, 2003; Withers *et al*., 2019)

Constructed wetlands act as man's way of speeding up the recovery process.

Creating a balance between pollution and repair is a more realistic method towards remediation especially with the increase in population coupled with an increase in demand for food and shelter which causes an increase in the depletion and degradation of nature resources like water, forests, animals, etc. The use of constructed wetlands to treat agricultural drainage reduces the effects of pollution.

Restored and constructed wetlands are capable of significantly reducing water-borne Nitrogen transports (e.g. Land et al., 2016, Vymazal, 2016, Vymazal and Brezinova, 2018) and have been recognized for its remediation qualities and efficiency in preventing eutrophication.

Constructed wetlands are a designed and man-made complex of saturated substrates, emergent and submergent vegetation, animal life, and water that simulate natural wetlands for human use and benefits.

2. AIMS AND OBJECTIVES

The aims of the Thesis are:

- to describe water chemistry of agricultural drainage waters
- to summarize the use of constructed wetlands to treat agricultural drainage
- to describe the wetland systems at Velký Rybník designed to treat agricultural drainage
- to evaluate treatment efficiency of mentioned constructed wetlands with respect to nitrogen and phosphorus.

The objective is to access the efficiency of the horizontal subsurface flow constructed wetlands in the treatment of agricultural wastewater over a period of 2.5 years.

3. LITERATURE REVIEW

3.1. Natural Wetlands

Wetlands are an ecotone – an edge habitat, a transition zone between dry land and deep water as shown in Fig.1, an environment that is neither clearly terrestrial nor clearly aquatic. (Hammer and Bastian, 1989). A more recent definition of wetlands by NRC (1995) and presented by Mitsch and Gosselink (2000) is A wetland is an ecosystem that depends on constant or recurrent, shallow inundation or saturation at or near the surface of the substrate." Wetlands are land areas that are wet during part or all of the year because of their location in the landscape. Wetlands can be found at topographic depressions or in areas with high slopes and low permeable soils. In other cases, wetlands may be found at topographic highs or between stream drainages when land is poorly drained. Historically, wetlands were called swamps, marshes, bogs, fens, or sloughs, depending on existing plant and water conditions, and on geographic setting (Kadlec and Wallace 2008).

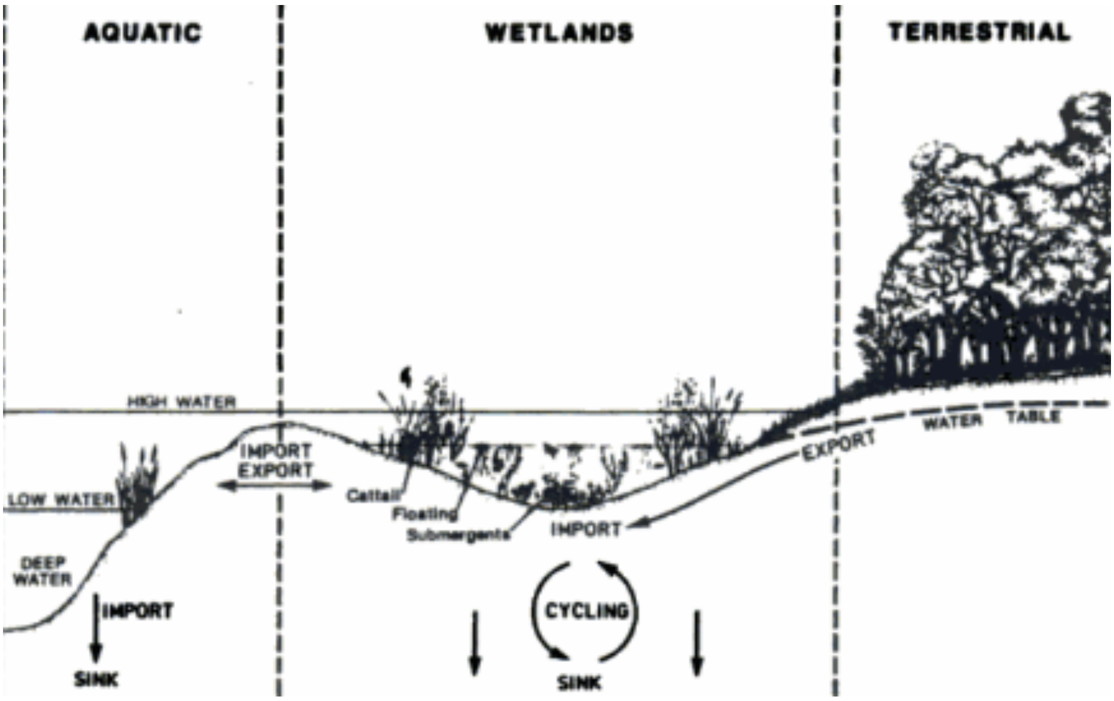


Figure 1: Wetlands as a transition between terrestrial and aquatic environment (Hammer and Bastian, 1989).

Wetlands are the major principal ecosystems on the planet that function in recycling the essential elements like oxygen, carbon, nitrogen, hydrogen, and phosphorus (and also metallic micronutrients).

Natural wetlands have been known to treat and purify water for a very long time. They have been acting as the planets' kidneys by purifying the waters and have been reported to perform this function for probably 250million years (Campbell and Ogden, 1999, Zedler, 2003). History times it as when sewage collection started in wastewater discharge sites in which early research efforts to replicate this in the form of constructed wetlands began. The ability of natural wetlands to retain nutrients from freshwaters was recognized long time ago and has been reported since the 1970s (e.g., Mitsch *et al*., 1979; Verry and Timmons, 1982; Richardson, 1990). Eutrophication of aquatic environments is a major environmental problem in large parts of the world. In Europe, EU legislation (the Water Framework Directive and the Marine Strategy Framework Directive), international conventions (OSPAR, HELCOM) and national environmental objectives emphasize the need to reduce the input of plant nutrients to freshwater and marine environments. A widely used method to achieve this is to let water pass through a constructed or restored wetland (CW) (Smith, 2003). The most important function of wetlands is water quality improvement, they provide effective, free treatment for many types of pollutants from point sources (municipal and certain industrial wastewater effluents) and non-point sources (mine, agricultural, and urban run-off) including organic matter, suspended solids, metals, and excess nutrients. (Hammer and Bastian, 1989).

Wetlands supersede other ecosystems with its high rate of biological activities, transforming many commonly found pollutants in wastewaters to harmless products or re-useable nutrients for the soil (Hammer and Bastian, 1989). For the fact that wetlands have a wide range of beneficial features that help in nutrient removal, from the type of vegetation to environmental conditions, climate and hydrology, which are prone to change over a long period of time, it can be difficult to deduce their values from one site to another (Kadlec and Knight, 1996).

Over the years, advancements in techniques and strategies have been made as a result of the research done to better understand the biological cycles and biochemical processes involved in Wetlands. This has helped tremendously in constructing treatment specific wetlands with the goal to perform a particular function (Kadlec and

Wallace, 2008). Also, research has attempted to evaluate constructed wetlands and their potential for treating water discharged from agricultural drainage tiles (Crumpton *et al*., 1993, Gersberg *et al*., 1983, Higgins *et al*., 1993). A large number of books have also been published on technical and scientific aspects of constructed wetlands, including USEPA (1988,1993,2000), WPCF (1990), Reed *et al*. (1995), Wissing and Hofmann (1995), Cooper *et al*. (1996), Kadlec and Knight (1996), Vymazal *et al*. (1998), Campbell and Ogden (1999), Kadlec *et al* (2000), Dias and Vymazal (2003), WERF (2006), Kadlec and Wallace (2008), and Vymazal and Kropfelova (2008).

3.2. History of Constructed Wetlands

The worldwide spread of this technology originated from research conducted at the Max Planck Institute in West Germany, starting in 1952 (Bastian and Hammer, 1993) and in the western hemisphere during the 1970s. The use of constructed wetlands has spread since 1985 because of its mechanical simplicity and biological complex systems that perform efficiently high levels of treatment.

Constructed wetlands were proposed as a suitable tool for removal of nitrogen from agricultural drainage in the early 1990s. Since then constructed wetlands with free water surface have been successfully used in Europe, North America, Asia, and Australia (Vymazal, 2017). The protection of water resources focused mainly on point source pollution such as municipal and industrial wastewaters that was easily identifiable and treatable based on their focused source, in the past. Non-point source pollution (NPSP) mainly associated with storm water runoff from agricultural, mining or urban lands used to be difficult to treat and had more detrimental effects on water resource quality than point source pollutants. Pierzynski et al. (1994) defined non-point source pollution as “pollution without an obvious, single point of discharge.”

In 1979, the U.S. EPA’s Report to Congress stated that non-point pollution caused 76% of the pollution to the lakes of which over half was caused by agricultural waste of which nitrate is the major pollutant (Shuckrow *et al*., 1980; Beutel *et* al., 2009). And according to the 2000 National Water Quality Inventory, agriculturally derived NPSP is the leading cause of water-quality degradation in surface waters (US EPA, 2002).

In 1973, the Mt. View Sanitary District in Martinez, California, constructed about 8.5 ha of FWS wetland marshes for wildlife habitat and wastewater discharge (James and

Bogart, 1989). Also in 1973, the first intentionally engineered, constructed wetland treatment pilot systems in North America were constructed at Brookhaven National Laboratory near Brookhaven, New York. These pilot treatment systems combined a marsh wetland with a pond and a meadow in series and were designated as the meadow/marsh/pond treatment system (Small, 1978). Also, the first HF system was built in 1972 Seymour, Wisconsin and researched through 1975 (Spangler *et al*., 1976b) the emergent vegetation was used to treat wastewater biologically to a degree of purity confirming its proficiency.

The first full-scale RZM also known as HSSF wetland went into operation for the treatment of municipal wastewater in 1974 in Liehenburg-Othfresen, Germany (Kickuth, 1977). The area of about 22 ha was used to dump waste material derived from mining iron ore. The concept of using heavy cohesive soils with low hydraulic conductivity was related to the traditional knowledge of soil treatment of sewage, based on the "sewage farming" in the United Kingdom (Cooper and Boon, 1987; Hiley, 1994). The first full-scale constructed wetland (CW) for water treatment was built in 1989, and in 1999 about 100 of them were in operation, majority of which were horizontal subsurface CWs, which were designed for secondary treatment of municipal and domestic wastewater. Recently HF CW has become used throughout the world, mostly in Germany where the number of these systems may exceed 50,000. Due to the low cost sand convenience of CW systems, they are starting to be employed in developing country as well.

Constructed wetlands are mainly composed of vegetation, a supporting medium or substrate which can be soils or porous media, microorganisms, and water. They remove water pollutants (such as organic load, fecal indicators, nutrients, suspended solids, heavy metals, organic compounds and nanomaterials) thereby improving water quality. (Kadlec and Wallace 2009; Vymazal 2010). Constructed wetlands are mostly designed to treat domestic wastewater focusing on nutrient removal and biochemical oxygen demand (BOD), while CWs for mitigating agricultural non-point runoff are directed at sediments and nutrients. Designs are made according to the functions expected of the CW. CWs can be built in all continents except Antarctica (Vymazal, 2011) They are particularly ideal for tropical and subtropical regions where the climate supports plant growth and microbial activity all year round which enhances the remediation processes (Merkl *et al*., 2005). Advantages of a constructed wetland include: relatively low construction costs for grading, dikes construction, piping and installation and vegetation planning depending on the type of constructed wetland; and low operating costs for monitoring water levels and plant viability, dike maintenance and water sample collection. Furthermore, treatment wetlands can be constructed using local materials and local labor, which is a major advantage in developing countries (Higgins *et al*., 1993).

3.3 Types of Constructed Wetlands

Major types of constructed wetlands (Fig. 2) include Free Surface Flow Constructed Wetlands, Subsurface Horizontal Flow Constructed Wetlands, and Subsurface Vertical Flow Constructed Wetlands.

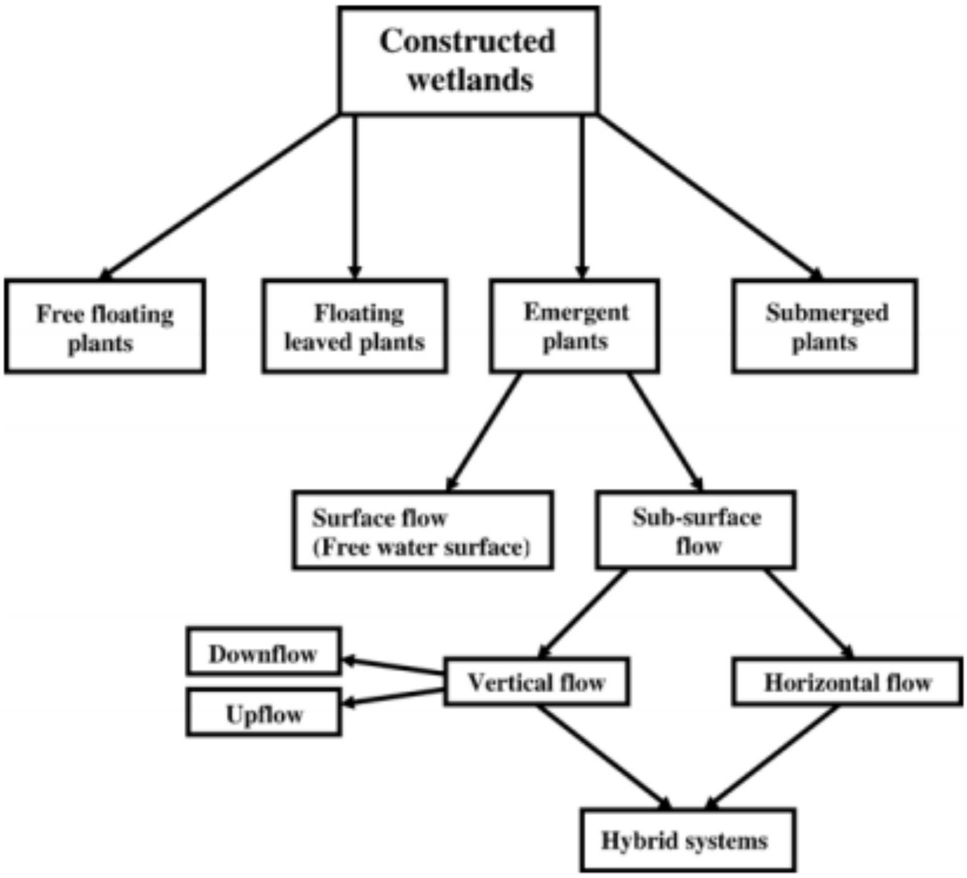


Figure 2: Types of Constructed wetlands (Vymazal, 2001)

3.3.1. Free Water Surface Constructed Wetlands (FWS CWs)

These are wetland systems with the water surface exposed or open to the atmosphere. This type of wetland is the closest to natural wetlands e.g. bogs, swamps and marshes. In this treatment, water flows from one inlet point to an outlet point over a vegetated soil surface. FWS consists mainly of one or more shallow basins with a barrier to prevent seepage to sensitive ground waters and also a submerged soil layer to support the roots of the emergent macrophyte vegetation. The most commonly used emergent vegetation includes cattail (*Typha spp*), bulrush (*Scirpus spp*) and reeds (*Phragmites spp*). FWS requires a large land area, especially if it is for Phosphorus and nitrogen removal. The plants form a canopy that covers the water surface as shown in Fig 3, thereby preventing algae growth and reducing wind-induced turbulence in the water flowing through the system. The submerged portions of the living plants, the standing dead plants, and the litter accumulated from previous growth are the most important parts as they provide the physical substrate for the periphytic-attached growth organisms (micro-organisms) responsible for the biological treatment in the system (EPA, 2000)

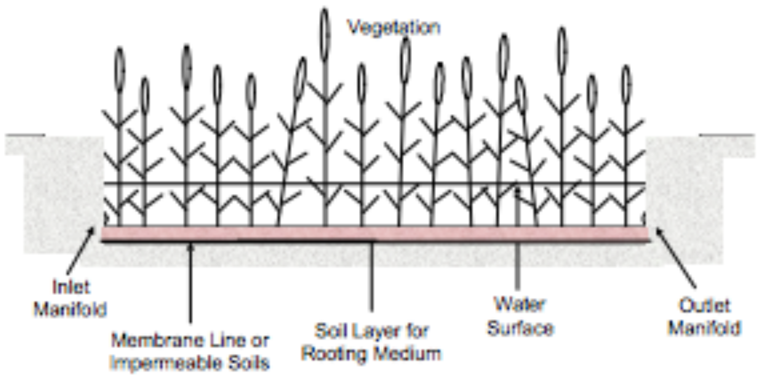


Figure 3: Free water Surface Wetland FWS (Reed, 2000)

3.3.2. Subsurface Flow Constructed Wetlands

Subsurface flow (SSF) constructed wetlands consist of beds in the ground, lined and filled with a granular medium, planted with emergent macrophytes. The wastewater flows through the granular medium and in the process comes in contact with biofilms and plant roots and rhizomes. A wide range of processes removes pollutants. SSF Constructed wetlands is constructed to replicate natural processes but in a more controlled environment. They are one of the most common types of CWs used all over the world (Garcia *et al*, 2010). Subsurface flow constructed reed beds generally have a greater potential to remove nitrogen than phosphorus (Vymazal *et al*., 1998). Subsurface flow wetlands are not common in agricultural settings because of the high cost of maintenance associated with the clogging of porous media (O'Geen *et al* 2010). Although, Kladivko *et al*., 2004 said subsurface drainages or tile drainage were a common way of reducing floods and keeping water level low during growing season in agricultural areas.

They are mainly designed to treat primary settled wastewater, although they are also commonly used to improve the quality of secondary effluents (Garcia *et al* 2010).

SSF CWs are classified into Vertical flow or Horizontal flow systems.

3.3.2.1. Horizontal subsurface flow wetland (HF)

In horizontal flow system, wastewater is maintained at a constant depth and flows horizontally below the surface of the granular medium. It is designed to treat primary effluent before soil dispersal or surface water discharge. And because plants cover the water during the treatment process, external influences of wildlife, pathogens or humans are diminished (Kradlec and Wallace, 2009).

It is the most widely used wetland system in Europe (Vymazal, 2005a). The fact that HF constructed wetlands for municipal sewage treatment usually exhibit higher treatment efficiency was demonstrated by Puigagut *et al*. (2007). The system is able to operate in both warm and colder conditions unlike the FWS system because of its insulation ability on the surface. HSSF is structured in form of a rectangular bed planted with an emergent plant (like reed canary grass (*Phalaris arundinacea*) and sweet manna grass *(Glyceria maxima*) and lined with an impermeable membrane as shown in Fig 4. It offers suitable conditions for nitrate reduction due to anoxic/anaerobic conditions in the filtration bed. The necessary organic are released from the decaying plant biomass and room and rhizomes (Zhai *et al*., 2013). HSSF wetlands have a limited capacity to oxidize ammonia, because of limited oxygen transfer. The system requires primary removal of big suspended solids; this is why it is usually applied as a secondary treatment system (Vymazal, 2008).

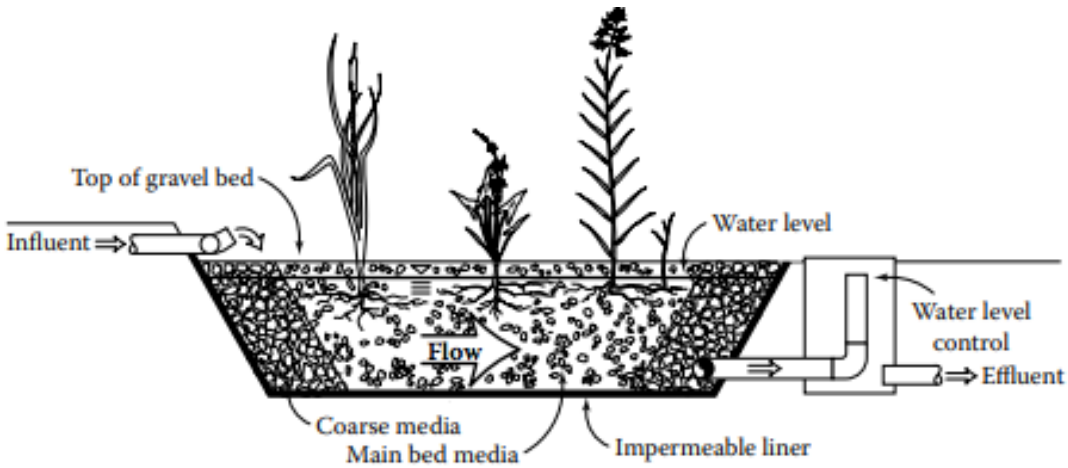


Figure 4: HF Constructed wetland (Wallace and Knight, 2006)

The distribution zone filled with stones, the impermeable liner, filtration medium, (filled with gravel and crushed rocks), the vegetation, water level in the bed, collection zone filled with large stones, collection drainage pipe, outlet structure for maintaining of water level in the bed (Vymazal, 2005).

Before 1995, gravel bed HSSF wetlands in the United States were frequently observed to be flooded (Kadlec and Knight, 1996). The two main causes were clogging of the media and improper hydraulic design. The same appeared to be true for other countries as well (Brix, 1994a), especially HF wetlands that used soil for the bed medium. The shallow soil bed is easily affected by the condition of low temperatures to reduce the removal of nutrients like as phosphorous and nitrogen (Zhai *et al*., 2016).

The HF CW can be insulated, by adding dry gravel and mulch layers; this balances the energy fluxes and prevents formation of ice (Henneck *et al*., 2001; Wallace *et al*., 2001; Kadlec, 2001b; Wallace and Knight, 2006). The layers add heat from naturally occurring insulators like standing dead, litter and snow trapped in the vegetation. These natural insulators perform important thermal functions especially in the winter months.

Anaerobic conditions present in horizontal SSF CWs permit the development of many groups of bacteria, several intermediate steps, and alternative biochemical pathways (Marahatta, 2004). In this research, HF CW was used.

3.3.2.2. Vertical subsurface flow system (VF)

Vertical flow (VF) wetlands (Fig. 5) distribute water across the surface of a sand or gravel bed planted with wetland vegetation. (Brix and Arias, 2005). The water is treated as it percolates through the plant root zone. The VF system provides more oxygen that helps nitrification than any other constructed wetland. Bio-solids dewatering wetlands can be thought of as a type of VF wetland system as there are a four types of VF designs depending on the hydraulic regimes: unsaturated flow (like conventional trickling filters), permanently saturated flow, intermittent unsaturated flow, and flood and drain wetlands (Garcia *et al*., 2010). The most common type of VF, which is often used in Europe, uses surface flooding (pulse loading) of the bed in a single-pass configuration (ONORM B 2505, 1997). VF CWs in Europe are developed to provide high levels of oxygen transfer thereby producing a nitrified effluent. This technology was initiated by the Max Planck Institute Process (MPIP) (Brix, 1994d). VF in North America is designed as vegetated recirculating gravel filters (Lemon *et al*., 1996). Upflow systems have been suggested to reduce oxygen transfer and promote reductive dehalogenation (Kassenga *et al*., 2004) and fill-and-drain systems have been implemented in North America to treat high strength waste and also oxidize ammonia (Behrends *et al*., 1996b;Austin and Lohan, 2005) The reed stems, rhizomes and roots provide drainage channels for water to pass through and sludges to dry quickly, by creating holes on the sludge surface and through evapotranspiration and microbiologically forced mineralization. These systems may be combined with HF or FWS wetlands to create nitrification-denitrification treatment trains (Cooper *et al*., 1999). The ability of VF wetlands to oxidize ammonia has resulted in their use in applications with higher ammonia than municipal or domestic wastewater. Concentrated wastewater can be treated in VF systems. Sludge from activated sludge plants may be dried in VF systems (Nielsen, 2004) and unsettled sewage can be treated in VF systems (Molle *et al*., 2005a). Another variation of VF wetlands relies on the use of overlying water to block oxygen creating anaerobic conditions in the bottom bed sediments. The surface water pools on top of the organics creating a downward flow into a zone with reducing conditions that foster appropriate sulfur chemistry to immobilize metals (Younger *et al*., 2002). Landfill leachates and food processing wastewaters can have high quantities of ammonia, and the key to reduction is the ability to nitrify. VF wetlands form part of the treatment process for those wastes (Burgoon *et al*., 1999;Kadlec, 2003c).

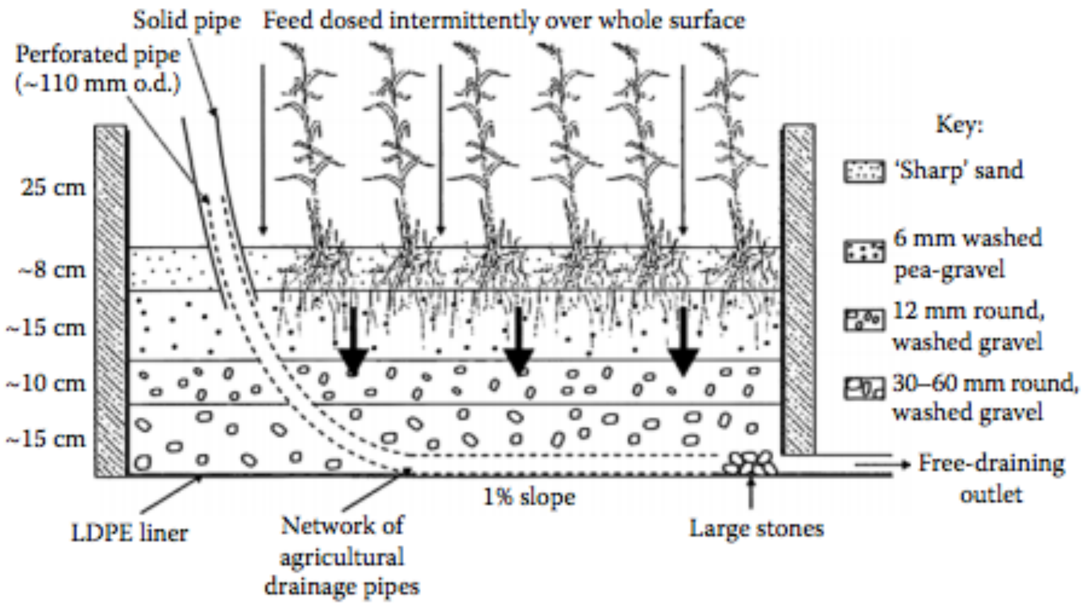


Figure 5: Arrangement of a Vertical flow constructed wetland (Cooper et al., 1996)

3.3.3 Hybrid constructed wetlands

This is a combination of two or more types of constructed wetlands in order to achieve higher removal efficiency when applied to municipal wastewater. This happens because one type of constructed wetland would not remove completely all nutrients or pollutants all depending on its design. Common combinations include HSSF-VF or VF-HSSF wetlands shown in studies as aerobic and anaerobic treatments done at the same time.

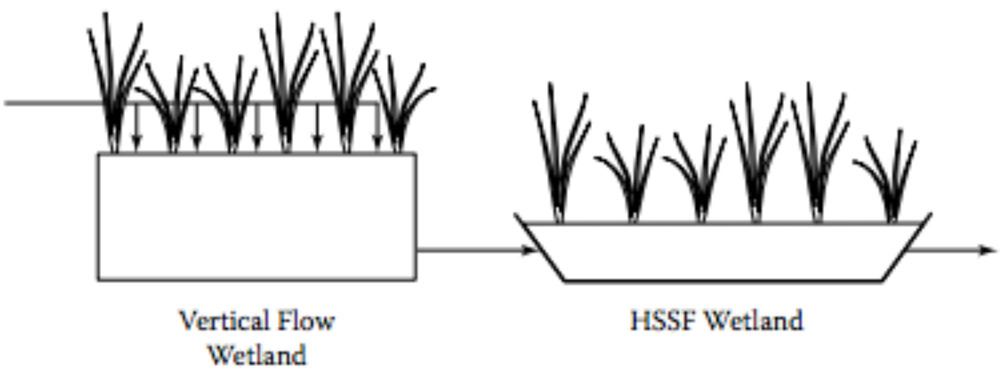


Figure 6: VF-HF wetland design (Source: Cooper et al., 1999)

An example is a three-stage constructed wetland design applied for the wastewater treatment of food processing, sewage and farming. It showed a level of nutrient removal especially for total nitrogen and phosphates (Serrano *et al*., 2011;Seres *et al*., 2017;Vymazal, 2008)

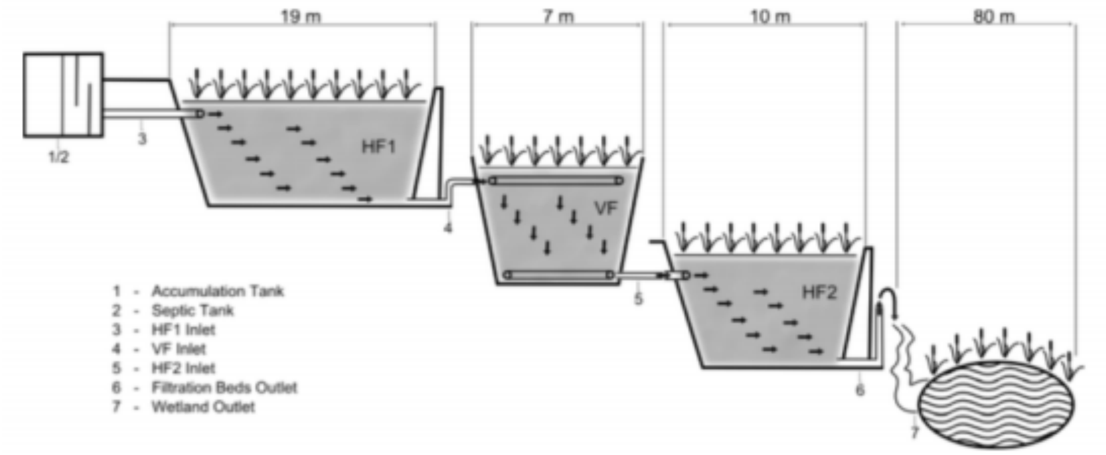


Figure 7: A three-stage hybrid constructed wetland in Czech Republic (Seres et al., 2017)

Removal rates for wetlands treating wastewater are generally higher. Vymazal (2007) reviewed nitrogen retention in constructed wetlands (Free-floating Plants (FFP), Free Water Surface systems (FWS), horizontal sub-surface flow systems (HF CWs) and vertical sub-surface flow systems (VF CWs)) and concluded that removal of total nitrogen varied between 40 and 50% with removed load ranging between 250 and 630 g N m−2 yr−1 depending on CW type and inflow loading.
The composition of nitrogen (N) and phosphorus (P) differs between wastewater effluent and agricultural cropland runoff (O'Geen et al 2010).
There are several mechanisms acting in CWs that contribute to the removal of contaminants, including: (1) sedimentation and burial (phosphorus, pesticides, particulate organic carbon, pathogens); (2) biogeochemical transformations (denitrification, methanogenesis, dimethylselinide production); (3) biotic uptake of nutrients and salts; (4) microbial degradation of pesticides and organic matter; (5) redox transformations affecting
solubility, sorption, and toxicity (e.g., As, Se, methyl-Hg); (6) predation of pathogens; and (7) photodegradation of pesticides and organic matter. As a result of these

processes, it is commonly considered that wetlands have a predominantly beneficial effect on water quality (Jordan et al., 2003; Zedler, 2003).

3.4 Factors that affect constructed wetlands

3.4.1.Vegetation (Macrophytes)

A constructed wetland begins its existence with the placement of the vegetation by the constructor in any type of subsurface flow Constructed wetland, and the seed bank associated with the selected soils in Free flow CWs.

The active reaction zone of constructed wetlands is the root zone (or rhizophere). As shown in Fig 8. Physiochemical and biological processes induced by interactions of plants, microorganisms, soil and pollutants take place here.

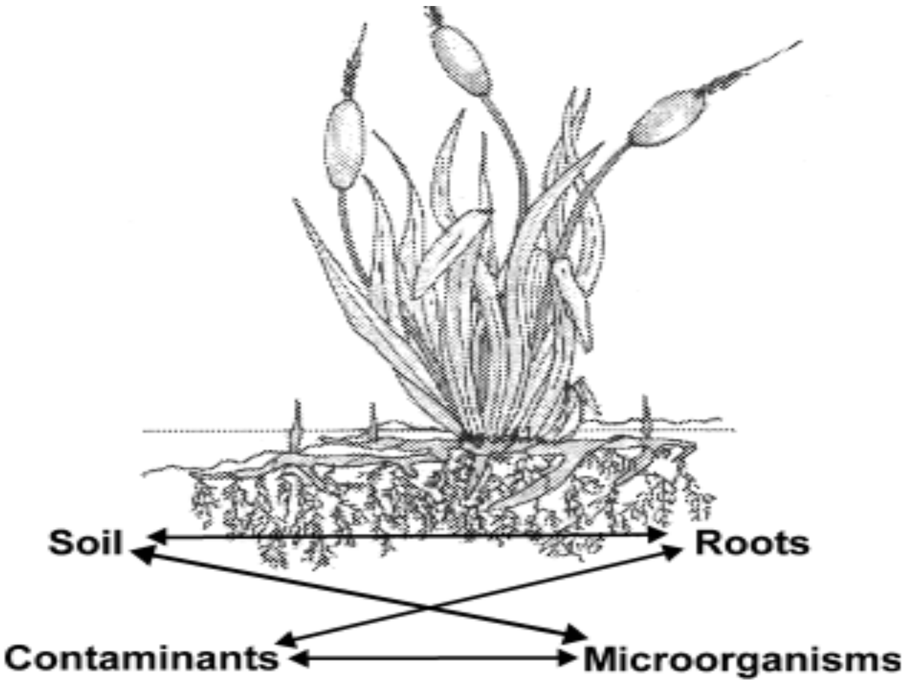


Figure 8: Possible interactions in the root zone of wetlands for wastewater treatment (Stottmeister et al., 2003).

Macrophytic plants provide much of the structure of the wetland treatment systems that fosters many removal processes. Wetland vegetation aids the transformation process by increasing the surface area of substrates for microbial attachments and biofilm communities responsible (Brix, 1997).

They have many peculiar properties that make them indispensable in constructed wetlands. The most important functions are the physical effects brought by the presence of the plants, they stabilize the bed surface, provide good conditions for filtration, prevent vertical flow systems from clogging, provide a large surface area for attached microbial growth, reduce the current velocity, increase contact time between the water and the plant surface area and insulate against frost during winter (Pettecrew and Kalff,1992 ; Somes et al 1996; O'Geen et al 2010). The importance of macrophytic plants for wetland treatment systems cannot be overemphasized as numerous studies have been carried out with and without plants and it has been proven that for high quality water treatment performance, plants are essential (Kadlec and Wallace, 2009). These new studies have reported that the choice of plant species, plant density and cropping systems affect the performance and efficiency of CWs in the removal of pollutants, greenhouse gas emissions and energy outputs. It is also known the plant root size can affect the hydraulic characteristics of the substrate and increase the retention time of wastewater in the substrate. The chosen technology directly affects the contaminants' biological degradation pathways and removal mechanism. While the anaerobic processes are mostly found in subsurface flow systems, aerobic processes prevail in surface flow systems. The hydraulic retention time, which includes the contact period between the water and the plant roots affects the extent in which the plant carries out its function in the removal or breakdown of pollutants. While plants are more effective in horizontal subsurface systems with long hydraulic retention times used to clean municipal wastewater, they have less effect in pollutant removal in periodically loaded vertical filters, which usually have short hydraulic retention times (Wissing, 1995).

Table 1: selection of plant species used in constructed wetlands. (Stottmeister et al 2003)

Scientific name	English name
Phragmites australis (Cav.) Trin. ex Steud.	common reed
Juncus spp.	rushes
Scirpus spp.	bulrushes
Typha angustifolia L.	narrow-leaved cattail
Typha latifolia L.	broad-leaved cattail
Iris pseudacorus L.	yellow flag
Acorus calamus L.	sweet flag
Glyceria maxima (Hartm.) Holmb.	reed grass
Carex spp.	sedges

In an experiment carried out in a temperature-controlled unit, when compared with the controls it showed that plants enhanced COD removal overall, they either attenuated (for *T. latifolia*) or eliminated (for *Carex rostrata* and *Scirpus acutus*) the seasonal decrease in performance expected at low temperatures. Hook *et al* (2003) showed that the presence of plants in HSSF constructed wetlands strongly affects the seasonal patterns of wastewater treatment. Also, wetlands vegetations have been shown to perform considerably well in nutrient uptake. For example, Phragmites, Scirpus and Typha were able to assimilate 90-130gN/m2 /yr (Debusk et al., 1995; Kadlec, 1999). Also, the effects of absorption by plants can be counteracted by mineralization and litter deposition, and so the plants must be harvested and disposed annually to ensure long-term nutrient removal (O'Geen et al., 2010). Since wetlands can take a number of years to achieve a fully developed vegetation community and root zone, the manner in which these systems are allowed to mature may be critical to their long-term performance (Stottmeister et al., 2003).

3.4.2. Rhizospheric microorganism

Microorganisms play an important role in Constructed wetlands by decomposition and remediation through their metabolism, especially the rhizopheric microorganisms. They also play a vital role in the transformation and mineralization of nutrients and organic pollutants (Stottmeister *et al*, 2003). They interact with the root exudation provided by the host plants, like the ion secretions, water, free oxygen, enzymes, mucilage and carbon contained metabolites (primary and secondary)(Bais *et al*., 2006). Microbial activities involved in Nitrogen removal are volatilization, denitrification, plant uptake, microbial uptake, ammonia absorption, ANAMMOX (anaerobic ammonium oxidation) and organic nitrogen burial. In a study carried out by Du *et al*., 2017 wetlands with *Canna generalis* showed high polyphosphate kinase activity, digesting phosphorus from soil and pseudomonas dominant microbial community gave good results. Introducing plant growth promoting bacteria in rhizophere could convert heavy metals and phosphorus into bioavailable and soluble forms as heavy metals affect nutrient uptake in plant tissues.

3.4.3.Temperature and pH

Climate also affects the constructed wetlands waste treatment efficiency. There have been many studies on the effects of temperature in CWs. Although most experiments on CWs have been as successful in the temperate regions of Europe and America as in tropical regions like Africa, each region has its advantage. For example, many CWs are placed in farmscapes or agricultural lands to intercept the storm water runoff, in such cases the systems receive the highest inflows during winter rains or spring snowmelts especially during the coldest time of the year (Werker et al., 2002). Temperature is a controlling variable for biochemical reaction rates and a seasonal factor in contaminant removal (O'Geen et al 2010).

Solar energy directly affects primary productivity, temperature and evapotranspiration (Kadlec, 1999) and contributes to photodegradation of organic compounds. Sunlight and wind control evapotranspiration and water loss affecting efficiencies calculated on a concentration basis (O'Geen *et al*., 2010).

3.5 Processes involved in Constructed wetlands

Processes that occur in CW are chemical, complex physical and biological. These include: microbial degradation or transformation of pollutants in the biofilms; sedimentation of solids; adsorption mediated by the supporting media; evaporation; direct oxidation and photooxidation of the pollutants; bacterial die-off and predation; phytoremediation processes and particularly the interactions between both processes (Wetzel, 2000); Kadlec and Wallace, 2008).

Jordan et al., 2003 said wetlands serve as sinks, filters and transformers of water-quality constituents. O'Geen et al., 2010 explained that additions, transformations and translocation were the three general processes that governed retention and/or removal of water-quality contaminants. Additions/ input loading affects the rate and pathway of removal processes, transformations lead to a change n form/phase and reactivity of constituents, while translocation processes render contaminants inactive, often through burial (O'Geen *et al*., 2010).

Evapotranspiration Water loss to the atmosphere occurs from open or subsurface water surfaces (evaporation), and through emergent plants (transpiration). A combination of the two processes is called Evapotranspiration. This water loss is closely tied to wetland water temperature (Kadlec and Wallace, 2009). Evapotranspiration is the primary energy loss mechanism for the wetland and serves to reduce majority of the energy. Water vapor loss is mainly caused by solar radiation in large wetlands and maybe heat transfer from the atmosphere for small wetlands. It is controlled by the same wetland energy balance, which describes wetlands water temperature (Garcia *et al*., 2010). Evapotranspiration rate varies because it depends on numerous factors influencing the ecosystem's prevailing microclimate as shown by Kadlec and Knight (1996). In Europe, constructed wetlands for wastewater treatments experience water loss of about 5 – 15mm/day in the summer because of evapotranspiration (Schutte and Fehr, 1992).

3.5.1 Nitrogen removal

Various forms of nitrogen are continually involved in reversible chemical processes from inorganic to organic compounds. While some of these processes require energy to proceed, others release energy, which is then used by organisms to grow and survive. The form of nitrogen found in the inflow is important to know beforehand as wetlands do not remove organic N and ammonium as effectively as they would remove nitrate (Phipps and Crumpton, 1994). The main form of nitrogen found in tile drainage is nitrate-N and this is caused by the nitrification of fertilizers found in agricultural fields (Polson and Addiscott, 2005). Also Fučik et al, 2015; Gramlich et al 2018 blamed it on the accelerated mineralization of organic matter in the soil, in aerobic conditions of tiled arable soils. High total nitrogen loading usually correlates with high flows in spring and fall runoff, while low TN loads occur in the summer at low stream flow where nitrogen removal processes have more time (Hill, 1996). All of these transformation processes are required for wetland ecosystems to function (Vymazal, 2007). Large quantities of N are removed by Denitrification (Faulwetter *et al*., 2009). It is the process whereby a biofilm bacterium converts bioavailable N in the form of nitrate (NO_3^-) to atmospheric N_2. This also means that denitrification reverses the large-scale anthropogenic N fixation and counters eutrophication. In order to promote this effect, wetlands should be constructed to enable denitrification. The removal of nitrogen by nitrification and denitrification is highly dependent on water temperature (USEPA, 1988; Reed *et al*., 1995; Kadlec and Knight, 1996). This was also confirmed by Reilly *et al* 2000, Bachand and Horne (2000a, 2000b), who studied the dentification of wastewater in FWS CWs and concluded that nitrate removal depends on both water temperature and organic carbon activity.

Nitrogen removal efficiency (as shown in Table 2) is highly variable and the major factors that affect removal efficiency are inflow load and the ratio between the drained catchment and constructed wetlands surface areas (Fisher and Acreman, 2004; Koskiaho and Putinen, 2005; O'Geen et al., 2010).

Table 2: Removal of total nitrogen in various types of constructed wetlands (mean values) (Vymazal, 2007)

CW Type	Unit	TN in	TN out	Efficiency	N
Concentrations					
FFP	mg l^{-1}	14.6	6.6	54.8	14
FWS	mg l^{-1}	14.3	8.4	41.2	85
HSSF	mg l^{-1}	46.6	26.9	42.3	137
VSSF	mg l^{-1}	68.4	37.9	44.6	51
Loadings				Removed load	
FFP	g m^{-2} yr^{-1}	838	431	407	14
FWS	g m^{-2} yr^{-1}	466	219	247	85
HSSF	g m^{-2} yr^{-1}	644	394	250	113
VSSF	g m^{-2} yr^{-1}	1222	592	630	42

3.5.2. Phosphorus removal

CWs have become a popular management practice for phosphorus removal from agricultural runoff (Jordan et al., 2003; Raisin and Mitchel, 1995; Reinelt and Horner, 1995). P has a conservative nature in wetlands, the vegetation increases sedimentation and trapping of PP by slowing water velocity thereby providing a substrate for particles to adhere to and prevent resuspension (Braskerud, 2001).

Phosphorus has been known in nature as a limiting nutrient for algae and cyanobacteria in most freshwaters (Vollenweider, 1968; Miller et al 1974)

In wetlands, phosphorus occurs as phosphate in both organic and inorganic compounds, they occur predominantly as phosphate in natural waters and wastewater. They are good examples of elements that move in both directions between water and sediments. Phosphates are classified as organically bound phosphates, ortho-phosphate or condensed (pyro-, meta- and poly-) phosphates (US EPA, 2000).

The main sustainable removal mechanism for phosphorus is plant uptake and subsequent harvesting (Lantzke *et al*., 1998). However, the extent to which particular mechanism employed depends on the type of constructed wetland (Kadlec and Knight, 1996). Kadlec and Knight (1996) also reported that mechanisms that remove

phosphorus in CWs include only sorption on antecedent substrates, storage in biomass and the formation and accretion of new sediments and soils. The first two processes mentioned are saturable which means that they have finite capacity and so cannot contribute to long-term phosphorus removal (Dunne and Reddy, 2005). Although, the amount of phosphorus that can be removed by harvesting the plant biomass usually constitutes only an insignificant fraction of the amount of phosphorus loaded into the system with the sewage (Brix, 1997). Phosphorus uptake by macrophytes is highest during the start of growing season, and most regions that time is in early spring before maximum growth rate is reached (Boyd, 1969, Vymazal, 1995). When Phosphorus enters the wetland inflow column, it is quickly absorbed by bacteria, periphyton and aquatic plants (SM2). Research by Davis, 1982 and Richardson and Marshall, 1986 using radioisotopes has shown that the wetland biota provide a small short-term sink for phosphorus while the wetland soils serve as a long-term sink.
The removal of phosphorus in HF CWs is limited because media used for HF wetlands (e.g. crushed stones) do not contain high quantities of Fe, Al or Ca to aid precipitation and/or sorption of phosphorus (Vymazal, 2008).

Table 3: Removal of total phosphorus (TP) in various types of constructed wetlands (mean values) (Vymazal, 2007)

CW Type	Unit	TP in	TP out	Efficiency	N
Concentrations					
FFP	mg l^{-1}	3.8	2.2	42.1	14
FWS	mg l^{-1}	4.2	2.15	48.8	85
HSSF	mg l^{-1}	8.75	5.15	41.1	149
VSSF	mg l^{-1}	10.5	4.25	59.5	78
Loadings				Removed load	
FFP	g m^{-2} yr^{-1}	200	127	73	
FWS	g m^{-2} yr^{-1}	138	68	70	85
HSSF	g m^{-2} yr^{-1}	141	96	45	104
VSSF	g m^{-2} yr^{-1}	126	54	72	62

4. Materials and Methods

4.1 Description of the site

An ongoing experiment that started in 2018 used three experimental HF CWs to treat tile drainage from a watershed of about 15.73 ha in Czech Republic. The site located about 100km southeast from Prague in the watershed of a drinking water reservoir called Švihov, the major drinking water supply for Prague.

Picture 1: Detail of the drainage (bottom) and sedimentation pond with a distribution box from where the water is distributed to the constructed wetlands (top). Photo Jan Vymazal

It has an average altitude of 510m a.s.l. and area of drained fields within the watershed as 9.85 ha (Pic 1). The three CWs have surface areas of 79m2 (CW1), 90m2 (CW2) and 98m2 (CW3), which are planted with a combination of two hydrophytes: *Phalaris arundinacea* (Reed canarygrass) and *Glyceria maxima* (Sweet mannagrass) planted in parallel bands.

Fig 9 shows the substrates arrangements and water level adjustment. In the first two CWs, crushed rock (4-8mm) is mixed with air-dried birth woodchips in the substrate in a volume ratio of 10:1. The third CW has a 20cm layer of birch woodchips on top of gravel (4-8mm). The first CW has a water level of 10cm kept above the surface, the second CW has the water level kept 5cm below the surface, and the third CW has its water level kept 10cm above the surface to ensure the woodchips are flooded. All wetlands are 1.0 m deep and lined with 1 mm plastic liner. The experiment site is still being monitored and maintained with regular sampling being carried out.

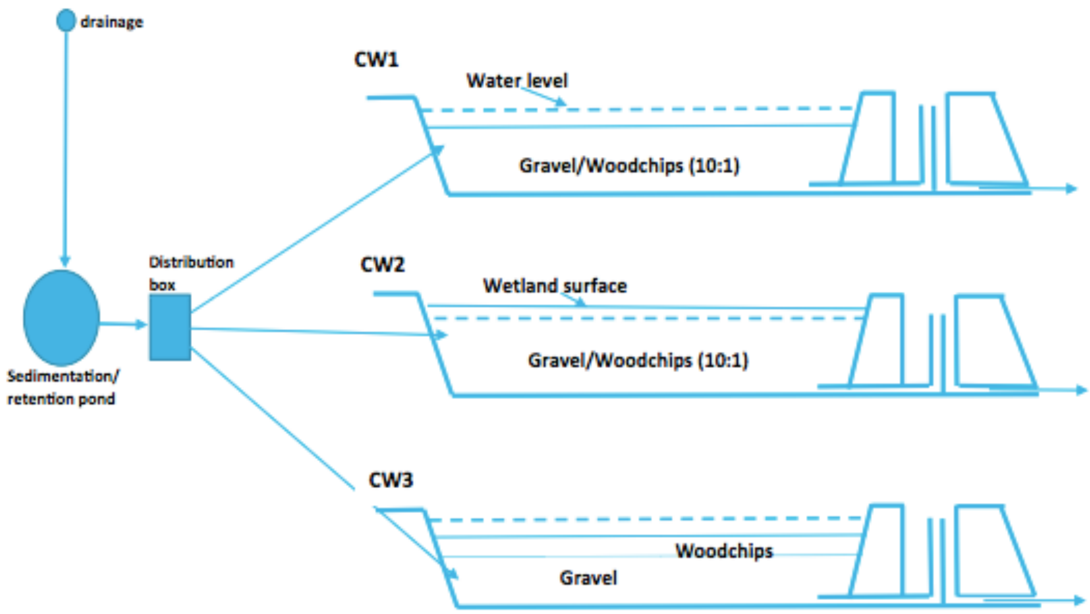
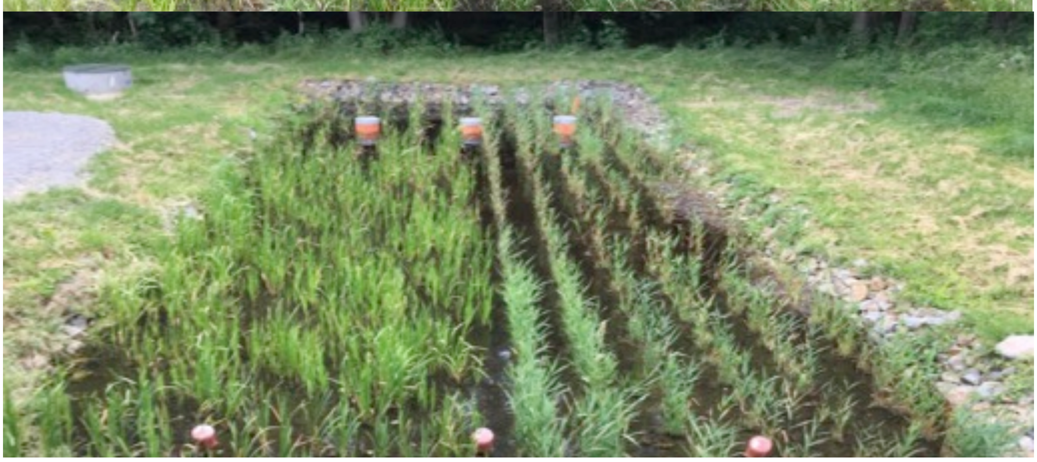


Figure 9: showing the schematic layout of three Constructed Wetlands: CW1, CW2 and CW3 (Vymazal et al 2020)

In 2020, water samples were taken at four locations: inflow to the wetlands and outflows from CW1, CW2 and CW3. The automatic samplers (ISCO 6712) were set up to sample the drainage water whenever they reached higher flows, which usually

meant higher Nitrogen concentrations, especially on the rising limb (Fucik *et al* 2017). Although most of the sampling had to be done manually due to unexpected low flows during the monitoring period. This was caused by a drop in the shallow groundwater level, which is the main source of water for the monitored drainage system, which is a consequence of drought in Central Europe as experienced in the last two years.
The outflows are equipped with continuous measurements of flow, dissolved oxygen and water temperature while the inflow is equipped with continuous measurements of flow and dissolved oxygen with 10mins reading. Water inflow was measured during each sampling by a HQ30d Portable Multi-Parameter Meter (Hach Lange, Loveland, USA). Nitrite, nitrate and sulphate were determined by ion chromatography on a Metrohm 883 Basic IC Plus analyzer (Herisau, Switzerland). Total nitrogen (TN) and total organic carbon (TOC) in water samples were analyzed using the Formacs HT TOC/TN analyzer (Skalar, Breda, The Netherlands). Ammonium was measured colorimetrically according to a Czech/European standard method ČSN EN ISO 7150-1 (1994) using Cary 60 UV-VIS spectrophotometer from Agilent.



Picture 2: Pictures of the three CWs planted with Glyceria maxima (left) and Phalaris arundinacea (right)

Statistical analysis was performed with the SPSS data analysis software. Differences between inflow and outflow were calculated using t –test. The results of the sampling done from August 2018 to December 2020, was analyzed to get an overview on performance throughout the duration of the experiment.

Statistical analyses were performed with the software Statistica12 (StatSoft, Tulsa, OK, USA). Differences in plant biomass, nutrient concentrations in the biomass and differences between treatments were analyzed after checking the normality using the Shapiro-Wilk, but because plant growth was affected by drought and was not enough to be harvested, plant biomass results was not used in this report. Nutrient concentrations were tested by parametric ANOVA and a post hoc Tukey HSD tests. The comparison of treatment wetlands was tested by non-parametric Kruskall-Wallis and Dunn tests. All statistical analyses were evaluated at $\alpha = 0.05$ level.

5. RESULTS

5.1 Flow

The tile drainage flow average for the entire monitoring period from Aug 2018 to Dec 2020 in Fig 10 was 0.10l/s while the average flow into the three wetlands was 0.128l/s, 0.083l/s, and 0.089l/s with a hydraulic load of 0.139l/m2d, 0.080l/m2d and 0.078l/m2d for CW1, CW2 and CW3 respectively.

Average flow

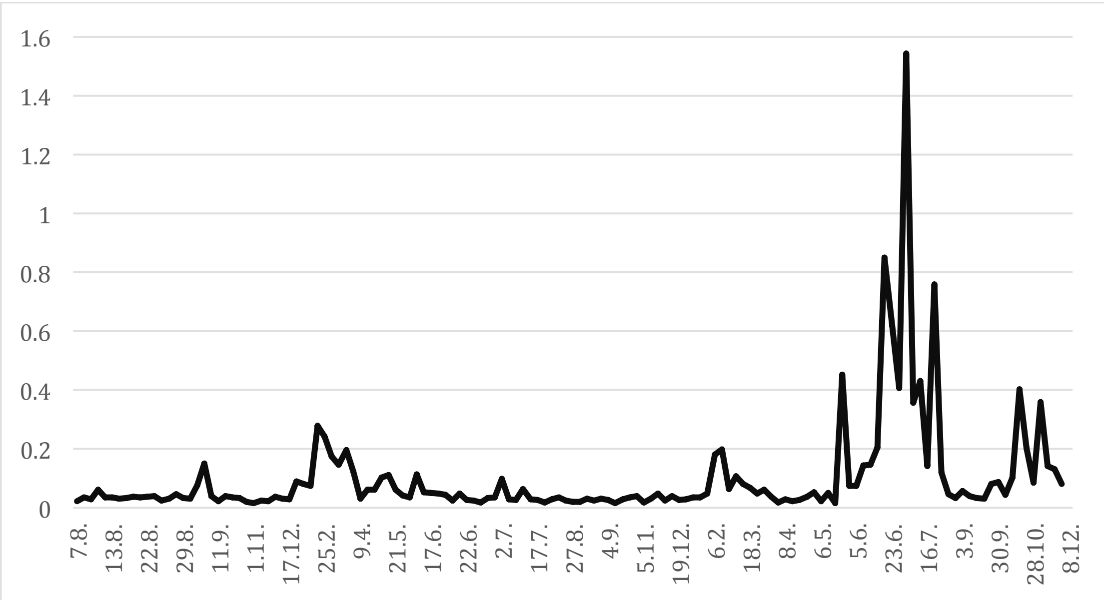


Figure 10: Average daily flow of the days sampling was done from August 2018 – December 2020.

Temperature

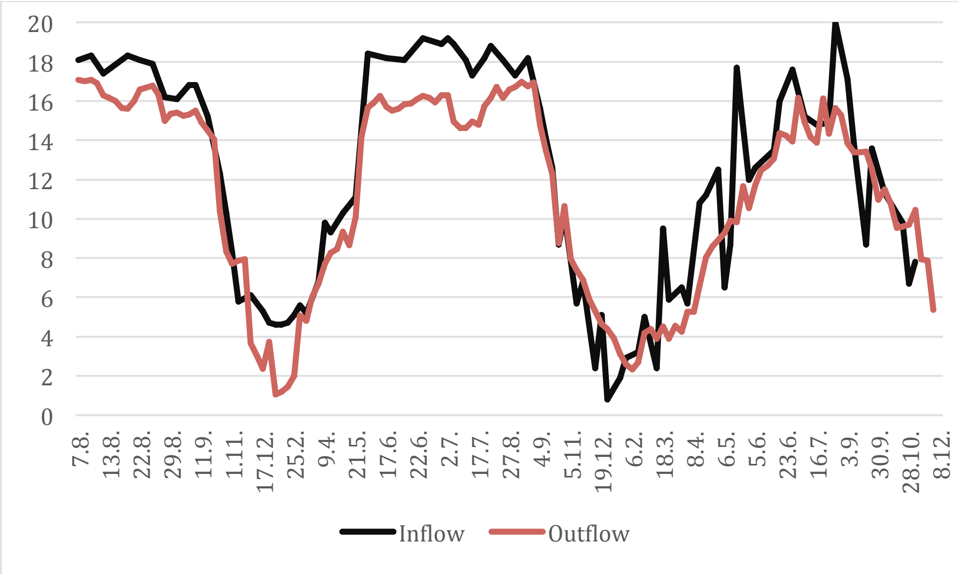


Figure 11: Water temperature concentrations of inflow and outflow during the period August 2018 to August 2020.

5.2 Water Temperature, pH, and suspended solids

The average inflow water temperature during the samplings was 11.69°C (range from 0.8- 20°C) while average outflow water temperature from the constructed wetlands was 11.26°C (ranging from 1.05 – 17.09°C) (Fig 11). However as reported by Vymazal et al., 2003 there were no significant statistical differences between the average water temperatures for outflow and inflow. Temperature of the wastewater followed seasonal patterns

The Average pH value at the inflow was 6.81. While the outflow pH values from CW1, CW2 and CW3 were 6.9, 6.94 and 6.9 respectively (Fig.12). The pH shows that the agricultural wastewater was almost neutral (7). There was no significant difference in outflow and inflow pH except for a slight increase in the outflow for CW3 in a particular period, which could not be explained. However, a significant increase in pH was reported by Hoffmann et al., 2019.

The drainage water contains only very low concentrations of suspended solids (<2 mg/l), which is because the drainage is fed from only one source, by water percolating to the drainage system at the depth of 1.2m. The drainage discharge is about 100m below the agricultural field in between there is grass meadow. Also, drainage water is collected in a small pond before it is discharged into constructed wetlands. This means that there is no danger of clogging caused by suspended solids in the drainage water. Although as reported by Vymazal et al., (2020) other processes contributes to the clogging such as precipitation of insoluble sulfides under anaerobic conditions of a subsurface flow wetland.

pH

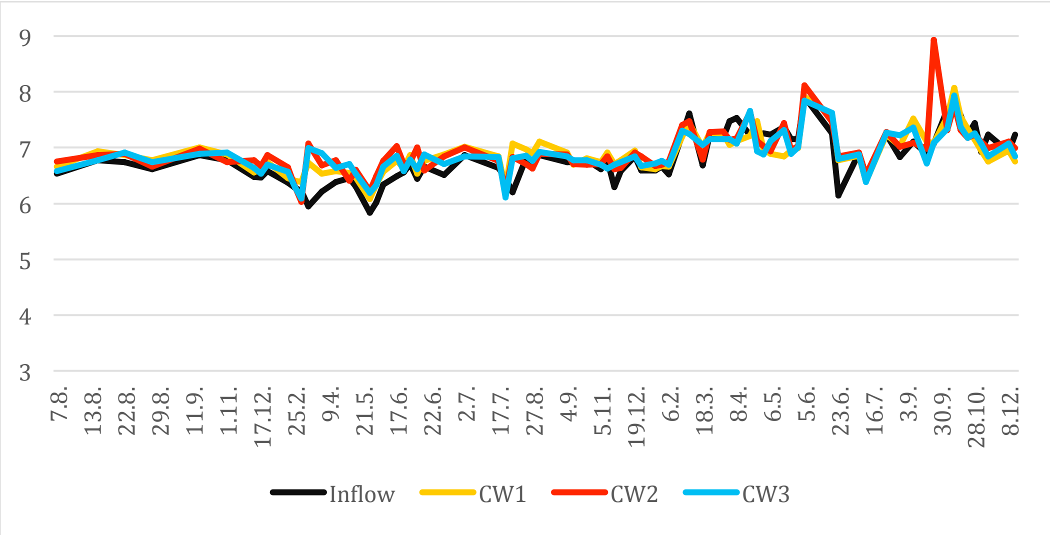


Figure 12: pH levels of the inflow and outflows from CW1, CW2 and CW3 during the period of August 2018 to December 2020.

5.3 Nitrogen

Nitrogen inflow concentrations exhibited a common pattern occurring in tile drainage in Czech Republic as reported by Vymazal et al., 2020 and Fucik et al., 2015, which is steady values during the summer and fall and increased values during winter and early spring (Fig 13). The average inflow concentration of total nitrogen was 16.41mg/l (range from 9.58 – 41.23) mg/l while the average outflow total nitrogen concentrations were 7.1863 (0.14 – 52.89), 6.629(0.19 - 55.01), 6.321 (0.02 - 120.18) mg/l (Fig 13) resulting in average removal efficiencies of 56.20%, 59.60% and 61.48% for CW1, CW2 and CW3 respectively. As it was reported earlier by Vymazal et al., (2020), the average outflow concentrations from all the three constructed wetlands were significantly lower than the inflow ($p<0.05$) but there was no significant difference between the average outflow concentrations from CW1, CW2 and CW3 ($p > 0.05$). Although removal efficiencies were higher in the first year of the experiment Vymazal et al (2020) reported removal efficiency percentage of 61.2%, 62.6% and 70.9% for CW1, CW2, and CW3 respectively. The slight difference can be caused by some factors like the weather, possible errors during sampling. The outflow concentrations varied widely during the monitored period (Fig13) Outflow concentrations were as low as 0.02mg/l in the beginning of the monitoring for about a month after which the concentrations started to increase with higher concentrations especially in winter, mostly around January 2019 after which concentrations began to decrease again. There was a huge spike in concentrations in February 2020, which drops to normal levels after the spike to moderate amounts afterwards (Fig 13).

TN

mg/l

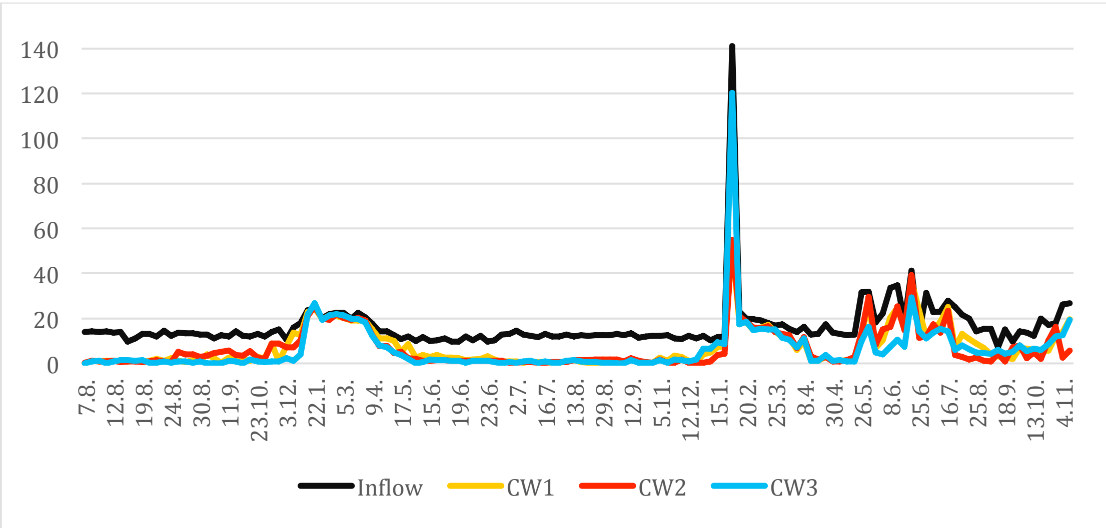


Figure 13: Concentrations of total nitrogen at inflow and outflows from CW1, CW2 and CW3 during the period August 2018 to December 2020.

The highest TN loads removed occurred in periods with high inflow, which was in February and June as shown in Fig 13. Nitrogen removal efficiency depends on the nitrogen loads (Knowles 1992; Seitzinger et al 2006). The average total nitrogen removal loads for the whole period is 1.92g/m^2d, 0.962g/m^2d and 0.839g/m^2d for

CW1, CW2 and CW3 respectively. As reported previously by Vymazal *et al*, the removed loads were also affected by inflow loadings.

The average nitrate inflow concentration is high as constructed wetlands remove nitrate more effectively than they would remove any other form of nitrogen (Phipps and Crumpton, 1994). With average inflow concentration of 67.854mg/l (range of 29.80 – 606.00mg/l and average outflow concentrations of 27.776mg/l (0.04 – 229.50) mg/l, 24.123mg/l (0 – 237.6) mg/l and 24.021mg/l (0 – 526) mg/l with removal efficiencies of 59.07%, 64.45% and 64.6% for CW1, CW2 and CW3 respectively.

As discussed by Vymazal et al 2020 nitrate was the largest volume of total nitrogen removed which is because all other forms of nitrogen increased their concentration due to the release of organic nitrogen from woodchips and consequent mineralization of the organic fraction. The removal efficiency percentage for both TN and nitrate are similar because the major removal process for total nitrogen was denitrification. High proportion of nitrate-N in tile drainage is reported in numerous papers (e.g Steidl et al., 2019;Tanner and Sukias, 2011;Carstensen et al., 2019; Hoffmann et al., 2019). Horizontal subsurface flow constructed wetland provided suitable conditions for nitrate reduction due to the anaerobic conditions in the filtration bed (Zhai et al., 2013)

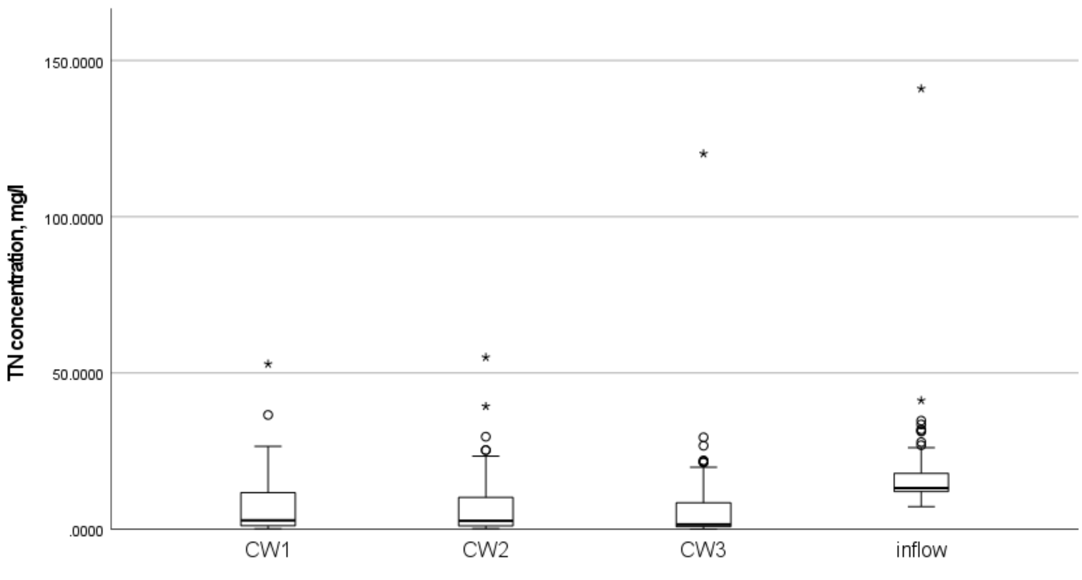


Figure 14: Box and whiskers of TN concentrations in influents and effluents from the CWs during the period August 2018 to December 2020

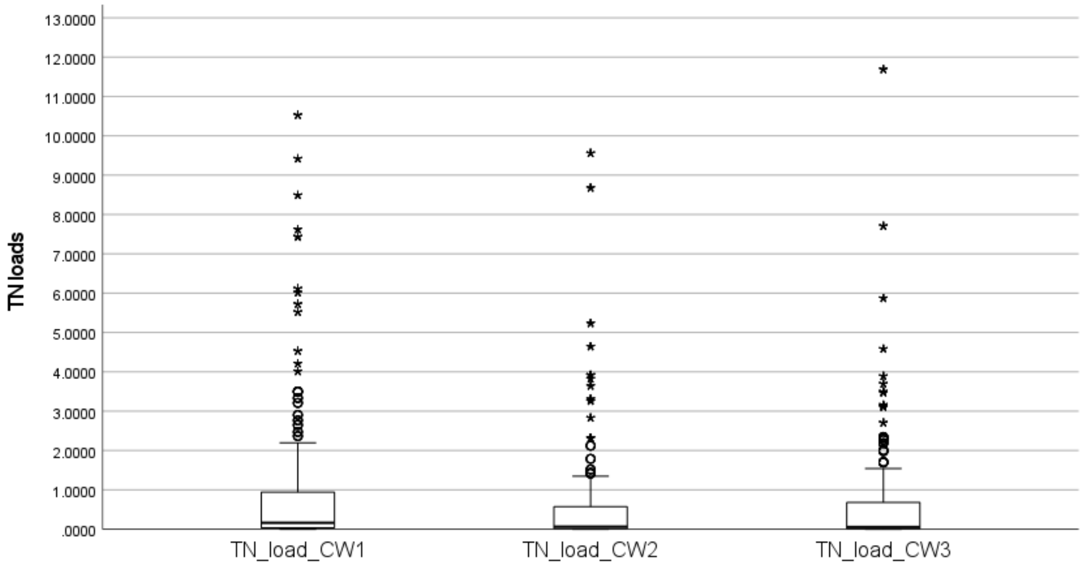


Figure 15: Box and whiskers of removed TN loads in the CWs during the period August 2018 to December 2020.

Nitrate

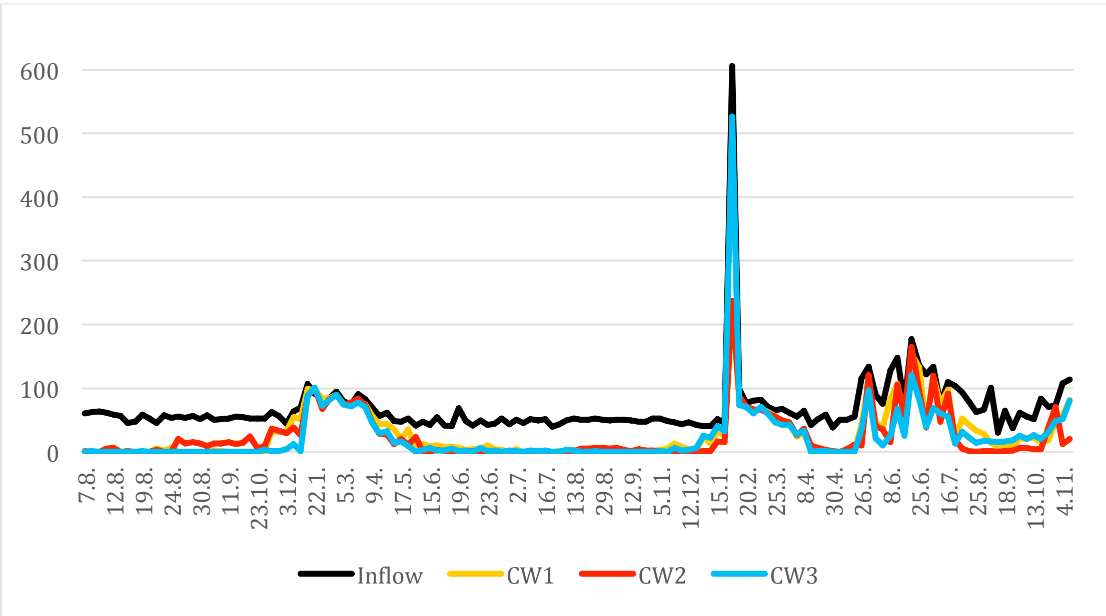


Figure 16: Concentrations of nitrate-N at inflow and the outflows from CW1, CW2 and CW3 during the period August 2018 – December 2020.

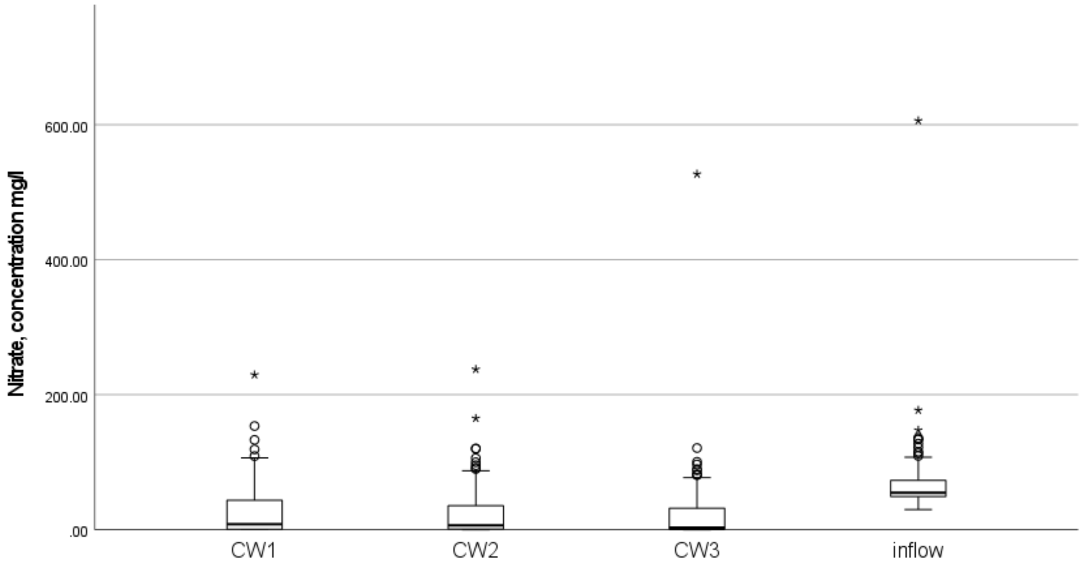


Figure 17: Box and whiskers of Nitrate concentrations in influents and effluents from the CWs during the period August 2018 to December 2020

TP

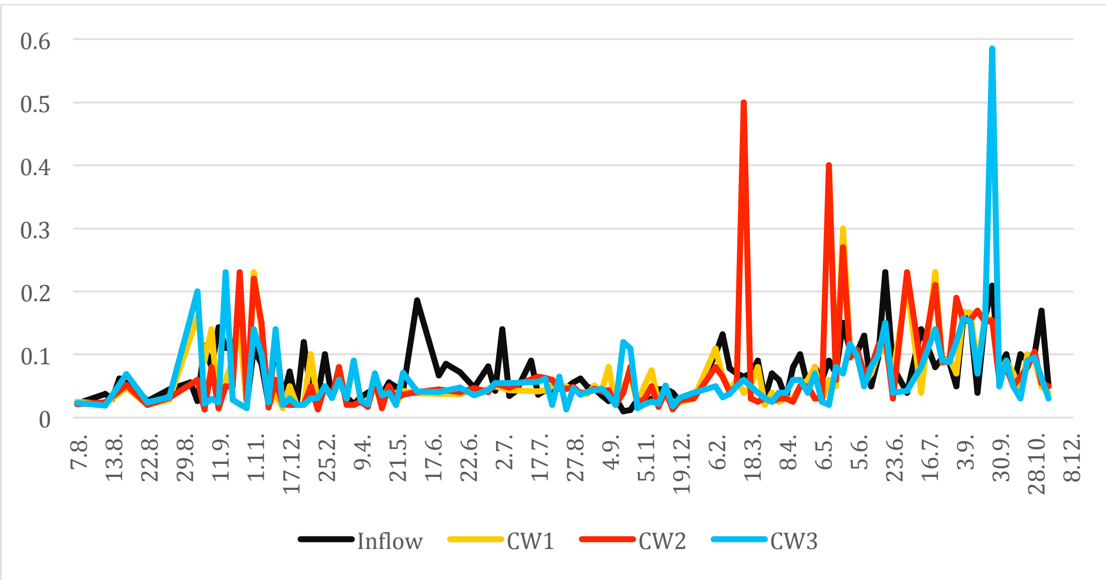


Figure 18: Total phosphorus concentration in inflows and outflows of CW1, CW2 and CW3 during the period August 2018 to December 2020

5.4. PHOSPHORUS

Phosphorus inflow concentrations were lower than expected (Fig 18). The average inflow of total phosphorus was 0.07097mg/l (range from 0.01 – 0.186mg/l) while the average outflow total phosphorus concentrations were 0.0651mg/l (0.014-0.21), 0.0696mg/l (0.013-0.5) mg/l and 0.0623mg/l (0.013-0.585) mg/l (Fig 16) resulting in average removal efficiencies of 8.27%, 1.930 % and 12.21% for CW1, CW2 and CW3 respectively. Also the average load concentration of phosphorus was 0.0517g/m2d, 0.0106g/m2d and 0.00595g/m2d for CW1, CW2 and CW3. The removal efficiencies were different in this case with CW3 having the highest removal efficiency and can be attributed to the CW contents (Fig 9). The concentration level of phosphorus remained low and steady in the first year and continued at that pace with a few spikes in CW2 in March and May 2020, and in September 2020 for CW3.

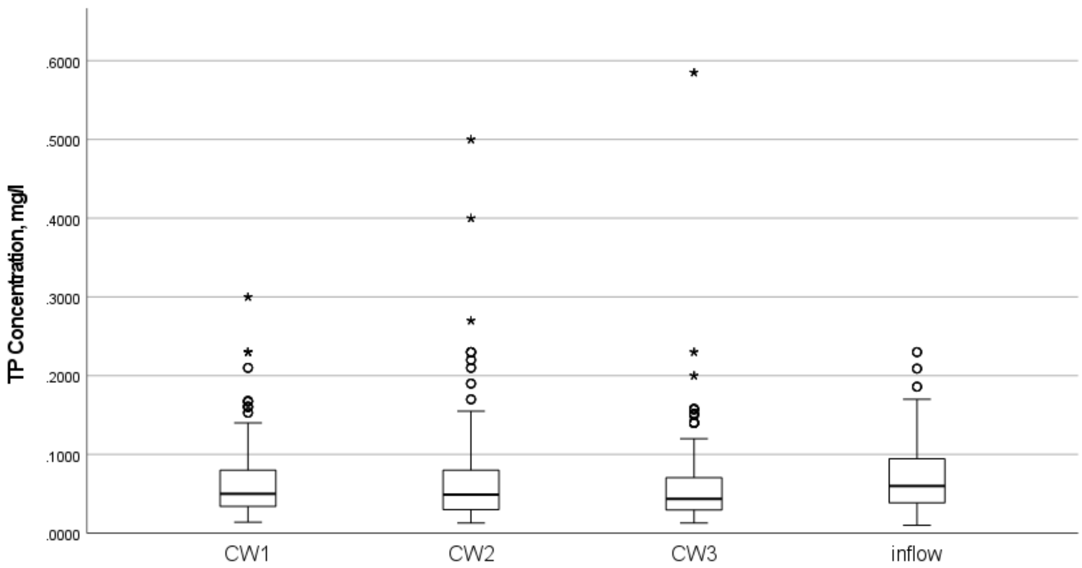


Figure 19: Box and whiskers of TP concentrations in influents and effluents from the CWs during the period August 2018 to December 2020

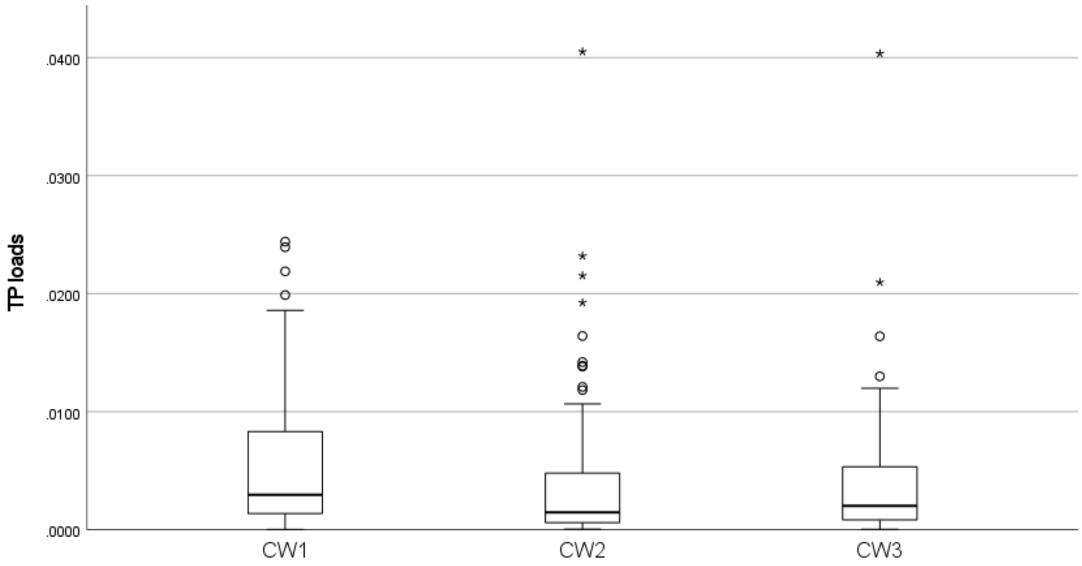


Figure 20: Box and whiskers of removed TP loads in the CWs during the period August 2018 to December 2020.

6. Discussion

Bruun *et al.* (2016b) reported removal of 57%, which is in line with results obtained from efficiencies for a longer period. The average TN removal efficiencies of 56.20%, 59.60% and 61.48% for CW1, CW2 and CW3 respectively, was higher compared to Vymazal and Brezinova, 2018 average TN 52.6% removal efficiency of a naturally vegetated ditch and lower when compared to the 92% removal efficiency of simulated runoff in agricultural ditches in Mississippi (Moore et al 2010). The results tallied with Lu et al (2009b), who reported a removal efficiency of TN to be 61% and was slightly higher than Kadlec et al (2010) who reported 50% TN removal efficiency. The TN removal efficiency of a tile drainage system in a Farmfield in Illinois was 37%. This particular wetland treatment was from 1995- 1997 (Kovacic *et al* 2000), which was lower than our result.

The efficiency of a surface flow CW in reducing N pollution from croplands in NE Italy that began in 1998 showed 90% removed nitrogen, which was in the form of nitric nitrogen, this is of course higher in efficiency than the result (Borin and Tocchetto, 2007).

A lot of papers have been written on nitrogen removal in agricultural drainage using free water surface constructed wetland, and as is expected the removal efficiencies varied per country. In USA, Larson et al, 2000 reported an efficiency of 84.5%, Diaz et al, 2012 reported 72%, Beutel et al., 2009 68% and Hunt et al 1999 had 57.3% while Tanner et al 2005 in New Zealand reported 78.3% efficiency and Kim et al 2010 reported 61.4%. While authors like Hunt et al, 1999 in USA and Kim et al., 2010 in Korea are on par with our results Reports by Larson et al., 2000 and Diaz et al, 2012 in the USA and Tanner et al, 2005 in New Zealand showed higher results than our results probably because of the type and size of the free water surface wetland which have been known to have higher overall performances than horizontal surface wetlands.

Other papers that had lower nitrogen removal efficiency than our paper are Tanner and Sukias, 2011 with 35.4%, Diaz *et al*., Braskerud, 2002 from Norway with17.4% and 4.0% and Kovacic et al 2006 with 22.0% (Supplementary material 5,Vymazal, 2017).

In the case of treatment wetlands in Yasima basins in Central Washington used to treat irrigation return flow in a nitrate dominated environment, the total Nitrogen removal efficiency was 57-63% while nitrate had a 90-93% removal efficiency (Beutel *et al*., 2009) which when compared to our result of nitrate removal efficiency of 61.4%

(59.07-64.5%) could be as a result of the high temperatures experienced in that region which aids denitrification. It also affects the process by controlling the rates of diffusion at the sediment- water interface in constructed wetlands (Crumpton and Phipps, 1992). In a similar case a wetland constructed for Kaoping River basin in Taiwan, to treat all kinds of non point source pollutants (untreated domestic, agricultural, industrial wastewaters) had a TN removal efficiency of 61% and also a removal efficiency of 66% for TP, which can be explained by the temperature in the region (Wu et al 2010).
Koracic et al., 2000 reported a nitrate removal efficiency of 38% out of the 37% TN (because nitrate was 99% out of total Nitrogen removed) in a 3-year study of tile drainage treatment near agricultural fields which was low when compared with our results nitrate efficiency of 59.07%, 64.5% and 64.6% for CW1, CW2 and CW3 respectively for a a study period of August 2018 – December 2020.

The average TP efficiency removal percentage for the three CWs were 7.47% (8.27%, 1.930 % and 12.21%) which is higher when compared to 2% for the overall average of 6 out of 9 CWs for the treatment wetland in the farmfield in Illinois (Moore *et al* 2010). Also Kovacic et al 2000, reported removal efficiency of TP to be 2%, which was a contrast to the 22% efficiency removal of dissolved phosphorus. Many studies confirm vegetated wetland beds as a very effective method for phosphorus removal as it has been shown to remove larger amounts of phosphorus than any other constructed wetland. (Maddox and Kingsley, 1989; Debusk *et al* 1990; Mitchell et al 1990;Van Oostrum and Cooper, 1990). When compared to the results from Flora and Kröger (2014a) at a drainage ditch at the aquaculture farm facility in Mississippi which had a removal efficiency of 47% and Vymazal and Brezinova 2018 who had TP removal efficiencies of 52.6% in 2015 and 51.3% in 2016 from a naturally vegetated ditch in Czech Republic, it is very low. Also compared to higher results from Hodai et al (2017) in Indiana, which was 65% also in a 2-stage ditch like Vymazal and Brezinova (2018) and Lu et al (2009a) who reported 57% removal efficiency of TP our results are low, although Braskerud (2002) reported a removal efficiency of 21-24% which is more in line with our result.

Steiner and Freeman, (1989) and Watson et al (1989) reported phosphorus removal is unpredictable and depends on the type of constructed wetland, substrate, chemical composition, removal efficiencies vary and range from 0-98% as seen in the reports above (Watson et al 1989; Steiner et al). Also phosphorus removal in constructed wetlands is usually low unless specific substrates are added to aid sorption (O'Geen *et al* 2010).

Although, Kadlec and Knight (1996) found little or no effects of temperature on overall constructed wetland performance. Kadlec and Reddy, 2001 concluded that if it existed it would be that increased temperature will cause a decrease in wetland treatment performance. Temperature affects several biogeochemical processes that regulate nutrient removal in the soils of wetlands thereby influencing the efficiency of the wetland treatment (Kadlec and Reddy, 2001). While overall performance might be affected, high temperatures increase evapotranspiration, cooler temperatures reduce evapotranspiration (O'Geen et al 2010). Sirivedhin and Gray (2006) reported a two-order magnitude increase in denitrification rates in wetland sediments, when the temperature was increased from 4-25°C in the field experiment, there was a slight increase in nitrate concentration of 0.001 – 0.606 kg N_2O/kg sediment per day compare to 0.003–1.014 kg N_2O/kg sediment per day.

In the case of efficiencies, Debusk *et al*., 2005 said that newly constructed wetlands are thought to be more effective at removing P than older wetlands due to rapid vegetation growth associated with P uptake. This is not the same at this particular site because the vegetation is not fast growing and the removed concentrations of Phosphorus were very low. This is contrary to the report by O'Geen *et al* 2007 who in an experiment on subsurface agricultural drainage for the San Joaquin basin saw that TN removal was 45% for the older CW (10 year old CW2) while the new CW removed 18% (CW1 new) It was also the same for TP where removal efficiency for the new CW was 18% while the old wetland was 72%.

In addition to initial wetland design features, a number of management techniques have been evaluated for improving long-term P removal performance by CWs, including routine vegetation harvesting, removal of accumulated sediment, and chemical immobilization of P in sediment using amendments (Debusk *et al*., 2005). This can only be done in areas where the wetlands plants grow fast enough to be harvested more than three times a year.

7. Conclusion

Wetland treatments have evolved over the five decades with innovations, new concepts and changes to the constructed wetlands according to its required needs especially with the changing climates, and environmental conditions. The results obtained reveal possible consistence in the efficiency of horizontal subsurface flow in agricultural tile drainage treatment. The average total nitrogen removal loads for the whole period is 1.92g/m^{2}d, 0.962g/m^{2}d and 0.839g/m^{2}d resulting in average removal efficiencies of 56.20%, 59.60% and 61.48% for CW1, CW2 and CW3 respectively

Average concentration loads of total phosphorus removed, was 0.0517g/m2d, 0.0106g/m2d and 0.00595g/m2d resulting in removal efficiencies of 8.27%, 1.930 % and 12.21% for CW1, CW2 and CW3 respectively. Also, the substrates used in the third constructed wetlands site (a mixture of birth woodchips on top of crushed rocks and water level 10 cm above the surface), has proved to be very effective and should be researched on more. The CW3 was noted to be more efficient than the other two wetlands in removing phosphorus as well as total nitrogen (TN). Non-point source pollutants can be removed successfully if all the parameters are put into consideration during designing and management of the constructed wetland.

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Supplementary materials

SM1: Phosphorus removal process (Reddy and D Ángelo, 1996)

SM2 : TN loads g/m^2d and TN statistical table

SM3:TP loads g/m^2d and statistical table

SM4:Nitrate loads g/m^2d and statistical table

SM5: Supplementary Material 5. The amount of nitrogen removed in constructed wetlands with free water surface treating agricultural drainage waters. Systems are ranked according to the removed load. *Based on concentration. Updated and modified from Vymazal (2017).