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REPORT

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THE UTILITY OF AGRICULTURAL CONSTRUCTED WETLANDS

Project acronym: AQUISAFE 1

by

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The Utility of Agricultural Constructed Wetlands

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Abstract

The Aquisafe project aims at mitigation of diffuse pollution from agricultural sources to protect surface water resources. The first project phase (2007-2009) focused on the review of available information and preliminary tests regarding

(i) most relevant contaminants,
(ii) system-analytical tools to assess sources and pathways of diffuse agricultural pollution,
(iii) the potential of mitigation zones, such as wetlands or riparian buffers, to reduce diffuse agricultural pollution of surface waters and
(iv) experimental setups to simulate mitigation zones under controlled conditions.

The present report deals with (iii), providing a review of the potential of constructed wetlands to protect surface waters from diffuse agricultural pollution.

Population growth and industrialization have lead to the demise of large majorities of natural wetland systems. Recent research continues to suggest the importance of these often saturated areas in the natural remediation of pollutants in water, as well as being aesthetically pleasing and acting as potential habitat for declining species. The drastic losses in wetland areas, combined with the realization of their importance, have stimulated recent attempts at wetland restoration and even construction of wetlands where they would not have naturally occurred.

In terms of substance remediation, constructed wetlands were traditionally used for the treatment of point sources, such as urban or industrial waste water. Recently they have also become increasingly popular for the treatment of diffuse pollution from agriculture and urban storm runoff.

Constructed wetlands have been shown to be efficient in the treatment of nutrients, organic matter and heavy metals. Few studies also show their potential against trace organics, such as pesticides and pharmaceutical residues and against pathogens.

Retention efficiencies vary significantly among case studies. In agricultural settings the following design criteria should be considered:

- Water residence time in wetlands is critical. Some studies concerning nutrient removal suggest that a constructed wetland should be about 5 % of the watershed area and assure water residence time of 7 days.
- Vegetation is important to slow down flow and increase sedimentation. Regular cutting and removal of plants is controversially discussed, since it may reduce their beneficial effect on wetland hydrology.
- Constant redox conditions are important to avoid release of sedimented or adsorbed pollutants.
- A combination of constructed wetlands with buffer strips showed very positive results.

Table of Contents

Chapter 1

Introduction

As populations continue to grow, and more nations become progressively industrialized, the task of maintaining the availability of even the most basic of natural resources has become increasingly challenging. The most fundamental example of this may be the degradation of natural bodies of water as a result of human impacts. As progress continues, the reserves of other natural resources (i.e. metals, petroleum, etc.) become increasingly strained and less easily accessible, the need for land in which to store refuse and other waste products intensifies, and the amounts of pollutants deriving from agricultural, commercial, and industrial activities increase. As a result, the bodies of water surrounding and supplying these activities continue to degrade increasingly more rapidly. In addition to the influx of pollutants into these systems, human modification of natural landscapes has also altered the hydrology of these water bodies. Higher percentages of impervious surfaces, decreased vegetation, and the rerouting of the natural flow of water via sewers, ditching, and agricultural drainage tile has made the natural hydrology of some areas nearly nonexistent. Perhaps one of the most crucial mistakes leading towards the current status of global water quality has been the removal of natural wetland areas via artificial drainage.

Wetlands are naturally occurring areas possessing three general characteristics relating to the hydrology, soils, and vegetation present in the area. In the United States, wetland ecosystems are legally defined because they are regulated. A comprehensive, and still widely accepted, definition of wetlands was adopted by the U.S. Fish and Wildlife Service in 1979 and presented in a paper entitled *Classification of Wetlands and Deepwater Habitats of the United States* (Cowardin et al., 1979):

> Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water…Wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes (water loving plants); (2) the substrate is predominantly undrained hydric soil; and (3) the substrate is nonsoil (organic) and is saturated with water or covered by shallow water at some time during the growing season of each year.

While the general definition of these systems is simple in principle, numerous, more complex definitions continue to be developed. These descriptions vary according to the objectives of the interested party, and tend to fall into two distinct groups; scientific and legal (Mitsch and Gosselink, 2000).

The increased interest in wetland environments, and the source of these varying definitions and interest groups, lies in the ongoing research that continues to uncover the importance of these areas to several aspects of global biology, hydrology, climatology, etc. Functions of wetland environments have been shown to include the storage and cleansing of natural and human wastewater as well as acting as carbon dioxide sinks and, in turn, climate stabilizers (EPA, 1994; Mitsch and Gosselink, 2000; IDEM, 2006). They also recharge groundwater aquifers, reduce the frequency and intensity of floods and droughts, and protect shorelines (EPA, 1994; Mitsch and Gosselink, 2000; IDEM, 2006). Wetlands are also known to host a myriad of flora and fauna unique to those environments, maintain extremely high biodiversity, and act as necessary habitats for migratory birds (EPA, 1994; Mitsch and Gosselink, 2000; IDEM, 2006).

Unfortunately, many of the Earth's wetland areas have been artificially drained or filled to accommodate the growth of cities and agriculture, or to ameliorate human health issues such as malaria (Figures 1 and 2) (Mitsch and Gosselink, 2000; IDEM, 2006). For example, it is estimated that, since the 1600's, the United States has drained more than

50% of the wetlands that once naturally occurred in the lower 48 states (i.e. excluding Alaska and Hawaii) (IDEM, 2006). By far the largest portion of this land alteration, and perhaps the largest land reclamation effort in history, lies within the Mississippi River Basin in the agriculturally dominated region of the Midwest. In Indiana alone, 4.7 million acres of the original 5.6 million that covered the state in the early 1700's were lost by the 1980's (IDEM, 2006). This corresponds to a statewide reduction from 25% coverage to less than 4% coverage by area (IDEM, 2006).

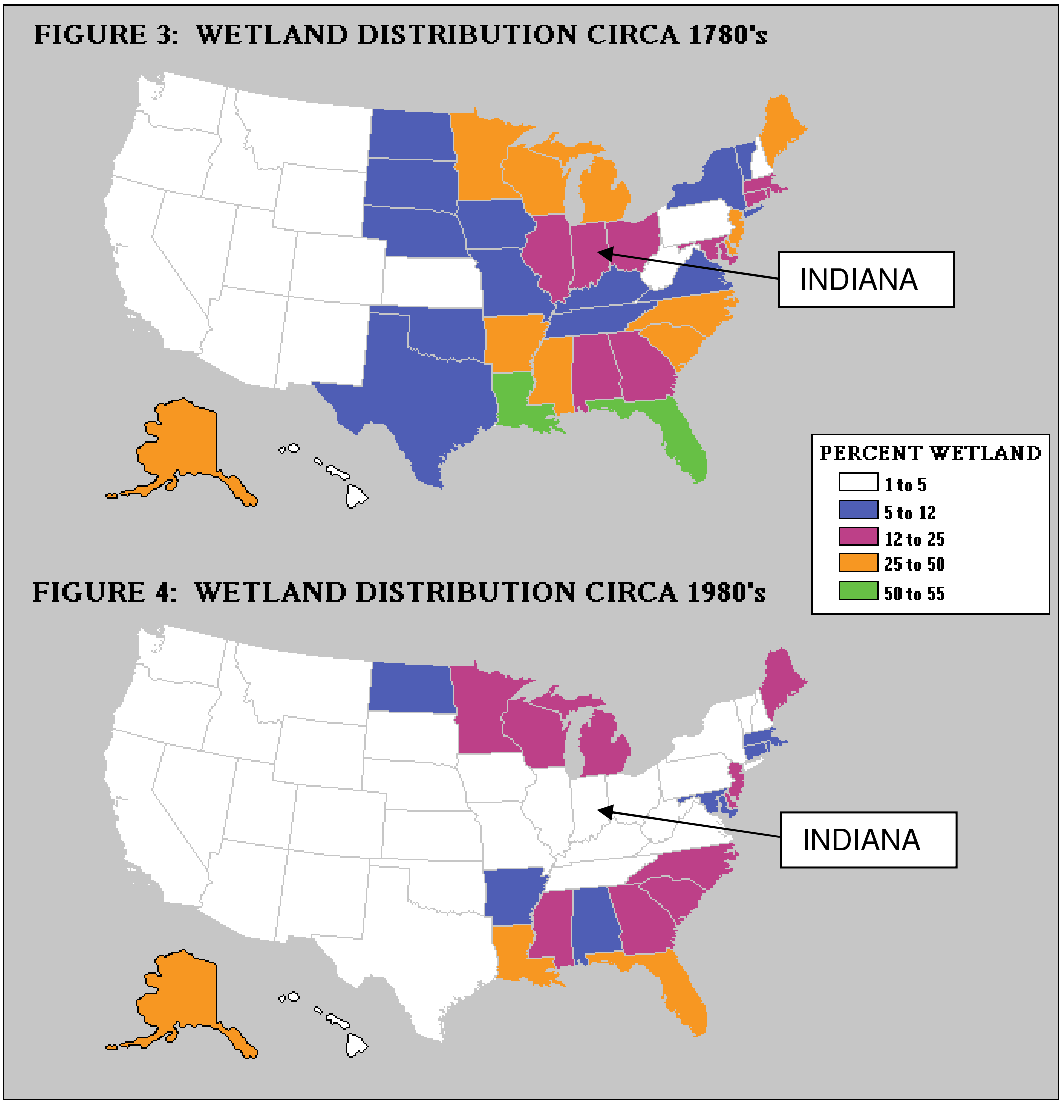

Figure 1 Maps comparing percentages of wetland areas in the U.S. in 1780 and 1980.
(http://www.geology.iupui.edu/academics/CLASSES/G690/Wetland_Ecosystems/2006/Lecture_01.ppt)

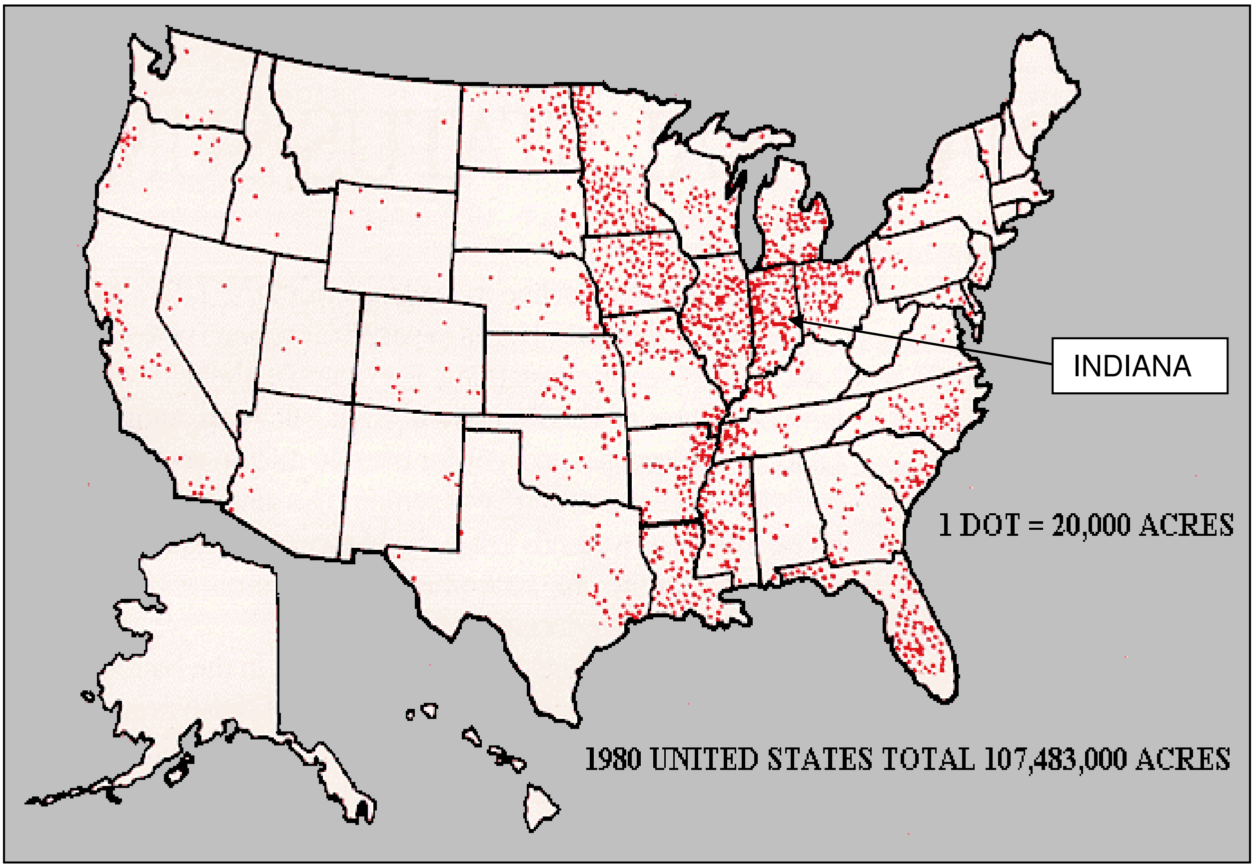

Figure 2 Map indicating areas of artificial drainage in the U.S. as of 1980.
(http://www.geology.iupui.edu/academics/CLASSES/G690/Wetland_Ecosystems/2006/Lecture_01.ppt)

As science has continued its efforts in understanding the potential function and value of wetlands in regards to water quality, so too has the concern for and knowledge of water quality issues within the general public grown. Even in the Midwestern United States, where agricultural ties run deep through history and change seems to have been slowed by disputes between science and tradition, the popularity of these unique landscapes for their aesthetic value and potential sources of naturally cleaner water seems to be growing. The drastic losses in wetland areas, combined with the realization of their importance, have stimulated recent attempts at restoration and even construction of wetland areas where they would not have naturally occurred. Constructed wetlands generally have less construction, operation, and energy costs than other forms of wastewater treatment (Hunt et al., 1995). In addition, they are more flexible to pollutant loading and soil specificity in that they are able to be built on aerated upland soils and, with subsequent flooding, hydric soil conditions will be able to develop and hydric vegetation able to establish itself (Hunt et al., 1995).

The following is a literature review exploring the aspects of constructed wetland design and utility as well as a description of and success in sequestering a wide range of common water pollutants as a basis for the implementation of constructed wetlands against diffuse agricultural sources. Chapter 2 gives an overview of the significance of a range of common water pollutants and their possible removal by constructed wetlands. Since similar pollutants are tackled by constructed wetlands under very different settings (e.g., point versus diffuse sources), chapter 3 reviews different types of implementation. Finally chapter 4 highlights important design criteria for the implementation of constructed wetlands in agricultural settings.

Chapter 2
Constructed Wetlands

There exist two primary types of constructed wetlands used in the treatment of wastewater. In this report, wastewater includes all types of polluted water. Our goal is to try to understand the applicability of wetland processes on contaminant removal regardless of whether the sources occur as point or diffuse pollution. These include surface flow systems and subsurface flow systems (Figure 3) (EPA, 1994; Mitsch and Gosselink, 2000). Surface flow systems are those in which wastewater input is brought into a basin lined with low permeability soils (EPA, 1994; Mitsch and Gosselink, 2000). These are generally the most popular type as they more closely mimic natural wetlands (EPA, 1994; Mitsch and Gosselink, 2000). They are more often subject to saturation or standing water and act as a better habitat for a variety of plant species including emergent, submerged, and floating vegetation (EPA, 1994; Mitsch and Gosselink, 2000). Advantages to implementing a surface flow system generally include a lower capital and operating cost as well as straightforward construction, operation, and maintenance (EPA, 1994). However, they do generally require a larger area than subsurface flow systems as there is a reduced surface area for the filtration and adsorptions of contaminants as well as the accommodation of biofilms (EPA, 1994).

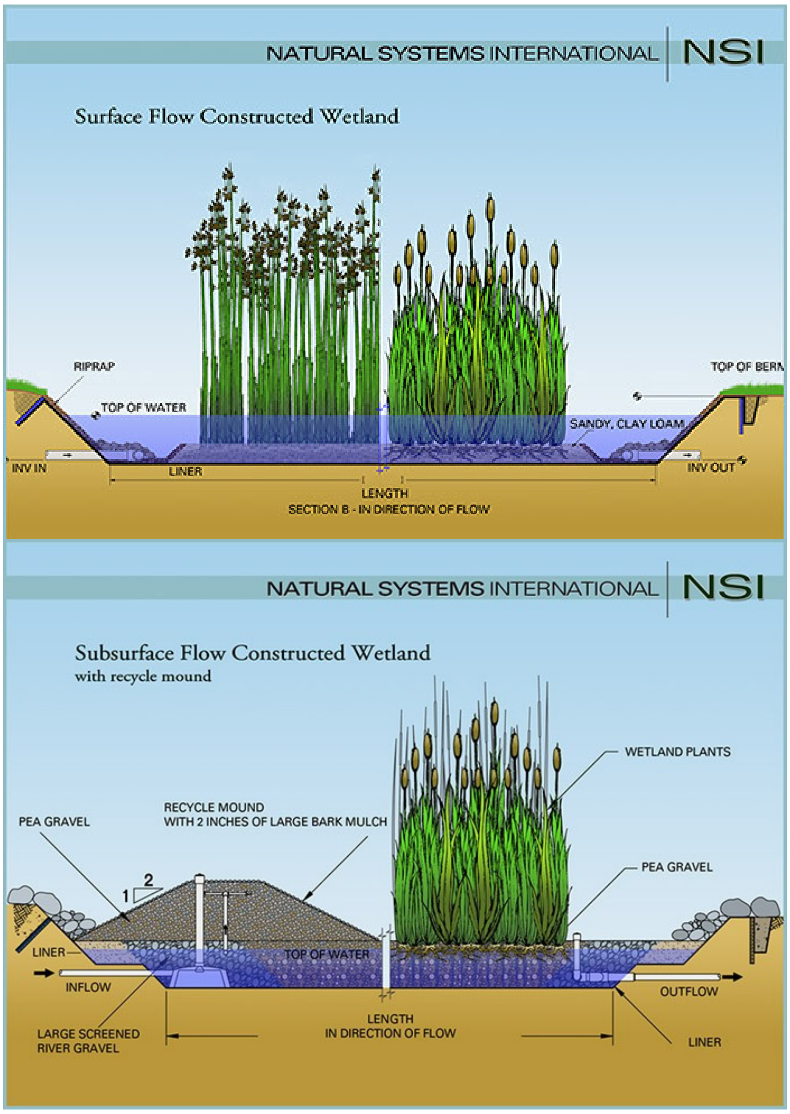

Figure 3 Diagrams comparing the design of surface flow constructed wetlands to subsurface flow constructed wetlands. (www.natsys-inc.com)

Subsurface wetlands more closely mimic a wastewater treatment plant in that water is brought into a basin lined in high permeability substrate (i.e. gravel or sand), is not subject to saturation or standing water, and is usually only associated with one or two emergent species such as cattails (*Typha sp.)* and bulrush (*Scirpus sp.)* (EPA, 1994; Mitsch and Gosselink, 2000). These systems generally function well at cold temperatures (have a greater cold tolerance), minimized pest and odor problems, and greater assimilation potential per unit of land area than surface flow systems (EPA, 1994). However, subsurface flow systems are generally more expensive to construct per unit and are therefore most commonly used to treat smaller flows (EPA, 1994). They are also generally more costly to repair and maintain, and have been associated with clogging issues, limited oxygen diffusion, and unintentional instances of surface flow (EPA, 1994; Hunt et al., 1995).

Early in the development of these systems, when natural wetland regulation was non-existent, natural wetlands too were used to treat wastewater (Mitsch and Gosselink, 2000). However, as the government and general public came to view wetland environments as increasingly rare and of particular value, these actions were halted (Mitsch and Gosselink, 2000). Only now, after extensive studies, have these practices been allowed by the Environmental Protection Agency (EPA), the enforcers of the Clean Water Act, under very controlled conditions (Mitsch and Gosselink, 2000).

Chapter 3
Pollutants

Constructed wetlands are designed to treat a wide variety of water pollutants. In order to fully understand the utility of constructed wetlands, it is important to understand the potential detriment that these target substances may cause if allowed to enter aquatic systems in excess. The following chapter is a description of many of the pollutant categories, for which these systems are designed to control, including their effects on humans and ecology and a report on the effectiveness of these constructed wetland systems in sequestering them in a variety of settings. An abbreviated summary is given in Table 1.

Pollutant type	**Main source of pollutants**	**Detriment to surface waters/wetlands**	**Potential remediation by wetlands**	**Observed range of efficiencies***
Nutrients, organic matter	-Mostly diffuse: Fertilizers and liquid manure -Point sources: Combined Sewer Overflows (CSO)	-Eutrophication of surface waters and decreased dissolved oxygen levels	-Denitrification (for NO_3), -Adsorption and plant uptake (for P), -Degradation (for organic matter)	-Nitrate: 29 - 75% -Total P: 1 - 93%
Low pH (from sulfuric or nitric acid)	-Diffuse: Acid rain -Point source: Lignite mining	-Limits biodiversity and microbiological processes -Problem of release of adsorbed substances	-Neutralization	-Situations requiring 90% or more removal of pH at or below 4 not economically feasible
Heavy metals	-Diffuse: Runoff from roads and houses, atmospheric deposition -Point source: Industry, mining	-Reduced microbiological processes due to toxicity	-Retention via adsorption	-Cu, Zn, Pb: 60-80% -Cr: up to 70 % -Fe, Mn, Co: > 90 % -Ni: 63 %
Pesticides	-Diffuse: Agriculture, gardens and parks, building materials	-Various unknown affects to flora and fauna (i.e. DDT and predatory birds)	-Retention possible via degradation and adsorption	-Few studies show high potential (e.g., 90 % for metolachlor, 83 % for simazine)
Pharmaceuticals	-Diffuse: Combined sewer overflows; manure applications -Point source: Wastewater treatment facilities	-Various unknown affects to flora and fauna (i.e. estrogenic compounds and the feminization of male fish)	-Biological degradation	-Highly substance-dependent
Pathogens	-Diffuse: Combined sewer overflows, animal waste	-Cause disease -Degrade taste and odor	-Retention/ degradation	-Few studies show high potential (e.g. > 99 % for fecal coliforms)

*Results from a wide variety of wetland designs, environments, loading rates, etc…

Table 1: Summarization of sources and effects of several categories of surface water pollutants, the potential implications wetland areas may have in remediating these pollutants, and a succinct collection of a few of the retention capabilities as reported by the reviewed literature.

3.1 Fertilizers and Liquid Manure

The application, and subsequent loss, of fertilizers and liquid manure in excess of plant demands and soil sorption capacities augments the nutrient transfer to aquatic ecosystems worldwide and has caused concerns regarding the eutrophication of both fresh and coastal waters (Uusi-Kämppä et al., 2000; Liikanen et al., 2004; Reinhardt et al., 2005; 2006). When such compounds enter into these bodies of water, they increase biological production just as they would in a terrestrial setting. When high amounts of artificial and inorganic fertilizers enter these systems, eutrophication of the water may lead to an overabundance of organic matter, in the form of algal blooms, which are associated with elevated rates of bacterial composition, deoxygenation in the water, and ultimately death of intolerant organisms and the further accumulation of organic matter. The potential for the development of taste and odor and algal toxins is an important consideration for drinking water sources. Allochthonous input of organic wastes into these systems can carry similar and potentially more detrimental effects.

Before the implementation of legislation such as the Clean Water Act in the United States, several portions of major river systems were subjected to anoxic conditions and large scale fish kills. In fact, the application of these substances in the Midwestern United States continues to fuel a large anoxic zone in the Gulf of Mexico as a result of accumulation in, and transport by, the waters of the Mississippi River Basin. The implementation of treatment systems, such as constructed wetlands, may help relieve the burden on many of these affected systems through diversion and treatment of the polluted waters at the source before they are able to enter the natural waterways. The removal efficiencies of several wetlands treating both point and diffuse sources of these pollutants are reported in Chapter 4.

3.2 Acid Mine Drainage and Acid Rain

Acidification of water bodies is generally most closely associated with two sources; coal mining and acid rain (Ricklefs, 2001). In areas of heavy coal mining, such as the Eastern United States, the introduction of sulfur and reduced sulfur compounds, found in the coal, to atmospheric oxygen results in the oxidation of these compounds by sulfur bacteria to sulfates. When these sulfates are introduced to water draining the mines, this ultimately leads to the creation of sulfuric acid which is then transported to surrounding water bodies. It is from this process that the term acid mine drainage originated, and in some instances may cause pH levels so low that very few or no species are able to survive (Mitsch and Gosselink, 2000; Ricklefs, 2001; Ye et al., 2001).

Acid rain is also associated with combustion processes. During combustion nitrogenous and sulfurous oxides are formed. When combined with rain producing clouds, these gases dissolve to form acidic precipitation whose pH may be as low as 3 or 4, while the average pH for normal precipitation is approximately 5.6 (Ricklefs, 2001). In addition to being one of the primary sources of coal, the northeastern United States and bordering portions of Canada are appropriately some of the regions that are most affected by acid rain (Ricklefs, 2001). Due to the fact that there is a tendency for lakes and rivers in this region to be oligotropic, and therefore deficient in dissolved bases, acidic inputs into the system are not as readily buffered (Ricklefs, 2001). The pH of some water bodies in this region have been found to fall as low as 4.0 at which point the growth and reproduction of fish and other organisms are affected and may even result in death (Ricklefs, 2001). Soils may also be affected in that a lowering in soil pH may increase the rate of leaching of soil nutrients and precipitation of phosphorus compounds thereby making them unavailable for uptake by plants. Moreover, such low pH values may also destroy clay minerals. These results are all in addition to the obvious detriment to plants and property that acidified rain may be associated with including the destruction of plant leaves, paint on homes and automobiles, and limestone structures. Indiana adversely contributes to acid rain in the northeastern United States and Canada because Indiana fuels most of its electric power plants with low-grade (high sulfur and mercury) coals mined locally. Prevailing winds transport these atmospheric pollutants from the Midwest to the northeast and Canada.

While acidified rain may be a more difficult issue to approach and require the removal of the gases that aid in this process from power plants and automobiles, finding alternatives for energy, and reducing the overall demand for these energy sources, the use of treatment options including constructed wetland may aid in instances associated with acid mine drainage. While the elements of constructed wetland design aiding in the neutralization of elevated pH levels are not well understood, some success has been achieved. This topic is described in more detail in Chapter 4.2.

3.3 Heavy Metals

Heavy metals are another toxin becoming more prevalent in waterways and are associated with some of the same anthropogenic processes as acidification. Sources of these metals include wastes produced though mining, mineral smelting, and manufacturing processes, fungicides, and burning of leaded gasoline in countries where they have not yet been outlawed. In addition to direct input into soil and bodies of water, these metals may also accumulate if released into the air in the form of particulate matter or if carried by runoff (e.g., from building materials or road runoff). While the effects of these metals are not yet fully understood, even low concentrations of mercury, arsenic, lead, copper, nickel, zinc, cadmium, manganese, and other heavy metals are toxic to most life forms. For example, if copper concentrations are found to be around 30 parts per million (ppm) in temperate soils, the area is considered unpolluted (Ricklefs, 2001). If this concentration increases to 100 ppm, mosses, lichens, and large fungi are adversely affected (Ricklefs, 2001). At 1000 ppm, earthworm populations drop significantly, and at 5000 ppm, most vascular plants are unable to survive (Ricklefs, 2001). Without many of these organisms, the decomposition of organic matter and nitrification of organic nitrogen also decreases. In vertebrates, these metals tend to be stored in the fatty tissues and passed onto larger prey. Therefore, if one progresses up the food chain, they find that levels in organisms increase exponentially and have been linked to gastric problems as well as interferences in neurological function and birth defects in more serious cases. Levels of over 1,000 ppm can be found in a 10-20 kilometers radius around smelting sites (Ricklefs, 2001). While these can be spread out with the use of taller smoke stacks, continued accumulation is inevitable without on site based systems of remediation.

Success has been found in several studies researching the sequestration of metals in constructed wetland systems. In one study, copper concentrations were reduced by over 80% and chromium concentrations by 69.5% in surface flow, subsurface flow, and floating aquatic vegetation type systems (Jin et al., 2003). In another, copper, zinc, and lead were reduced by 60 to 80% in a surface flow constructed wetland with minimal remobilization in surface flow (Knox et al., 2006). In addition, sediment was found to be the primary sink for these toxins and removed even more with the presence of organic matter (Knox et al., 2006). Finally, Ye et al. published results in 2001 regarding a surface flow constructed wetland's success in reducing metals from coal combustion byproduct. Iron, manganese, and cobalt were able to be removed by greater than 94% and nickel by 63% in the second year of the study (Ye et al., 2001). This study also found similar results to Knox et al. (2006) in that sediment acted as the primary sink for these pollutants.

It is important to note that constructed wetlands can also act also as a source for heavy metals when mineralisation of organic matter takes place (e.g., waste water irrigations fields can release heavy metals after a stop of irrigation).

3.4 Organic Compounds

3.4.1 Pesticides

The wide range of organic compounds accumulating in waterways may be the most diverse and readily intensifying. The primary sources of these compounds derive from their usage as pesticides including herbicides, fungicides, and insecticides. While some naturally occurring pesticides such as nicotine and pyrethrins are used as pesticides, most are laboratory created to ensure that the targets have had no previous exposure and therefore no opportunity to evolve resistance. These include organomercurials such as methylmercury, chlorinated hydrocarbons such as DDT, lindane, chlordane, and dieldrin, organophosphorus compounds such as parathion, malathion and glyphosate, carbamate insecticides, triazine herbicides, such as atrazine and simazine, and anilide herbicides, such as metolachlor (Ricklefs, 2001). While they are successful in their usage, they may accumulate in other areas of the ecosystem and negatively effect native populations of flora and fauna. While, in more recent years, steps have been taken towards methods of application that reduce these negative outcomes, e.g., by banning the most problematic substances, such as DDT, the increased application of these compounds in order to ensure their effectiveness often outweighs the effort.

While alternatives to the current intensive use of chemical pesticides are still in development, the use of mitigation systems such as constructed wetlands and microorganisms that are able to metabolize these compounds can be used to reduce the input into natural systems. A 2003 study of herbicide removal from wastewater produced by an enclosed nursery was conducted by Stearman et al. with the use of a subsurface flow system. They found that in cells with plants and lower rates of flow, metolachlor was able to be removed by as much as 90.2% and simazine by 83% (Stearman et al., 2003). In cells that were not vegetated or flow was higher, these percentages, though still significant, were reduced (Strearman et al., 2003). In another study, constructed wetlands were built for the purpose of monitoring the transport and fate of atrazine and were subjected to simulated storm and runoff event flow equal to three volume additions (Moore et al., 2000). During the study, three target concentrations of 0, 73, and 147 micrograms per liter were reached and water, sediment, and plant samples collected (Moore et al., 2000). Results of the study found that 17 to 42% of the measured atrazine mass was retained within the first 30 to 36 meters of the wetlands and below detection in all of the collected sediment and plant samples (Moore et al., 2000). Aqueous half lives of the atrazine ranged from 16 to 48 days and a conservative buffer travel distances of 100 to 280 meters was proposed to be necessary for effective runoff mitigation (Moore et al., 2000).

3.4.2 Pharmaceuticals

A more recent water quality concern associated with organic compound contamination is the presence of pharmaceutical compounds in surface, ground and drinking water. These enter the waterways through a combination of pathways and include a) incomplete absorption by the body and subsequent passage in human waste products; b) the disposal of unwanted medications through sanitary sewers; and c) runoff from pasture lands and/or areas with land application of manure containing veterinary pharmaceuticals. Major cities throughout the United States, as well as Asia, Australia, Europe, and Canada, have discovered medications and their byproducts in local drinking water supplies. These compounds include antibiotics, such as sulfomethoxazole, anelgesics, such as acetaminophen, ibuprofen and diclofenac, betablockers, such as propanolol, the sex hormone ethinylestradiol, anti-epileptics, such as carbamazepine, lipid-lowering agents, such as clofibric acid, antidepressants including fluoxetine, and a wide range of others including those for veterinary use. Given the very low concentrations in finished drinking water (typically one needs to drink several liters per day for 70 years to get one daily dose of the original medication), a possible impact on humans of long term exposure is medically unlikely but not understood (Dieter and Mückter 2007, Dorne et al. 2007) and it is likely that the timing of exposure is important as is the understanding that exposure is not to individual compounds but rather complex mixtures making daily dose calculations but one method of considering the potential impacts. One also needs to consider that pharmaceuticals are designed for human intake, in contrast to other trace organics, which occur in even higher concentrations, such as pesticides. However, negative impacts of pharmaceutical residues have been shown in in non-target aquatic organisms (e.g., Isidori et al. 2005; Bencic et al., 2009).

While the effect of long term exposure to these compounds is not fully understood, the impact is likely to be negative to both humans as well as aquatic organisms. One commonly documented impact is the feminization of fish as the result of exposure to estrogenic compounds found in birth control as well as other medications and trace organics with hormone-like structure, such as bisphenol-A as a plastic ingredient or certain pesticides. Some populations of male fish have begun creating egg yolk proteins, which is a process usually exclusively carried out by the females (Associated Press, 2008). Other organisms, such as earthworms and zooplankton, residing at the base of the food chain, have also begun showing effects from these compounds (Associated Press, 2008).

Naturally based systems of remediation such as constructed wetlands have been found to reduce some pharmaceutical compounds at a high rate. In a study conducted in Spain, several pharmaceutical compounds were able to be removed, with varying efficiency, through the implementation of a subsurface flow constructed wetland (Matamoros and Bayona, 2006). Categorized according to removal efficiency, more than 80% of caffeine, salicylic acid, methyl dihydrojasmonate, and carboxy-ibuprofen and 50-80% of ibuprofren, hydroxyl-ibuprophen, and naproxen were able to be removed (Matamoros and Bayona, 2006). Some compounds were found to be resistant to removal including ketoprofen and diclofenac (Matamoros and Bayona, 2006). Others were removed by hydrophobic interactions, including mostly perfume and musk compounds, and concentrated in the gravel substrate as the result of sorption onto organic matter (Matamoros and Bayona, 2006). Similar tendencies of substance removal are found in WWTP (e.g. Miege et al. 2008).

3.4.3 Pathogens

Pathogens are microbes that cause disease and include a few types of bacteria, viruses, protozoa, and other organisms. These too have become an increasing concern in many water bodies as they are often associated with sewage discharges, leaking septic systems, stormwater runoff and runoff from animal feedlots. In addition to causing disease, these organisms may produce compounds associated with foul odor and taste of the water and therefore demand immediate action as the result of public outcry. Pathogens may include coliforms (i.e. fecal, *E. coli*), cryptosporidium, *Giardia lamblia*, hepatitis A, rotaviruses, and caliciviruses and are known as the cause of several illnesses of varying severity (EPA, 2008). Healy and Cawly (2002) reported a 99.77% removal efficiency of fecal coliforms by a constructed wetland's treatment of municipal wastewater.

Chapter 4
Settings of Constructed Wetlands

Constructed wetlands have been implemented in a number of settings for the treatment of wastewater. These settings include the treatment of point source pollution such as municipal and industrial wastewaters and landfill leachate as well as nonpoint sources such as mine drainage, stormwater, and agricultural wastewater. Since removal efficiencies are expected to be very different for different types of wastewater (e.g., high versus low concentrations or presence of toxic compounds) and flow (regular flow versus storm flow), settings are discussed separately below.

4.1 Point Source Pollution

The treatment of municipal wastewater with the use of wetlands was one of the initial studies of this nature, and its beginnings date from the implementation of natural sites which were eventually halted in the United States by the EPA (Mitsch and Gosselink, 2000). Today, this form of treatment is primarily conducted with the use of subsurface flow wetlands in order to remove suspended sediments, biological oxygen demand (BOD), and nutrients such as nitrogen and phosphorus. (EPA, 1994; Mitsch and Gosselink, 2000). While wetlands are capable of storing other pollutants, including trace metals, controversy still exists as pollutants may concentrate in the substrate and fauna over time (Mitsch and Gosselink, 2000). Such systems have been constructed and implemented in many areas of the United States, Europe, Australia, and New Zealand (EPA, 1994; Mitsch and Gosselink, 2000).

Healy and Cawly (2002) published results from a surface flow constructed wetland and showed a 51% reduction in total nitrogen from a municipal wastewater source in Ireland. These results have been found to increase with the presence of organic matter in the substrate of these systems (Bastviken, 2003). In the same study by Healy and Cawly (2002), a 13% reduction in total phosphorus was reported. In addition, biological oxygen demand was reduced overall by 49% (Healy and Cawly, 2002). Several other studies have reported even more significant reductions in phosphorous. Also in a surface water system, Jing et al. (2002) reported 72, 80, and 46% removals of chemical oxygen demand, available nitrogen, and organic phosphorus, respectively from municipal and industrial wastewater sources.

Subsurface flow systems are also more commonly implemented for the treatment of landfill leachate (EPA, 1994; Mitsch and Gosselink, 2000; Bulc, Ferfila, and Vrhovšek, 2004). Groundwater that has percolated through the landfill is collected by impermeable liners and diverted through these systems (Mitsch and Gosselink, 2000; Bulc, Ferfila, and Vrhovšek, 2004). While of widely variable quality, the water originating from these systems tends to be regularly high in ammonium nitrogen and chemical oxygen demand, but may also include biological oxygen demand, organic carbon, chlorides, iron, manganese, and phenols in addition to being anoxic (Mitsch and Gosselink, 2000; Bulc, Ferfila, and Vrhovšek, 2004). Applied constructed wetland systems act as one of several options in the treatment of these areas and may be incorporated with spray irrigation, physical/chemical treatment, biological treatment, and piping to a wastewater treatment facility (Mitsch and Gosselink, 2000).

4.2 Nonpoint Source Pollution

The use of constructed wetlands as downstream treatment systems for mineral mines has been regularly implemented since the 1980's (Mitsch and Gosselink, 2000). Surface flow wetlands tend to be more commonly used in these instances (EPA, 1994).

Acid mine drainage waters are subject to very low pH and high concentrations of metals including iron, sulfate, aluminum, and trace metals, and is a serious water quality issue in many coal mining regions of the world (Mitsch and Gosselink, 2000; Ricklefs, 2001). In fact, studies of wetlands for this purpose may have been initiated when wetland environments dominated by cattails (*Typha sp.)* and bulrush (*Scirpus sp.)* were observed near acid seeps in a harsh environment where no other vegetation was able to survive (Mitsch and Gosselink, 2000). Concrete elements of design for these sites are not readily available, though some have been implemented and shown to be successful, as situations requiring upwards of 90% removal rates or the treatment of pH levels below 4 are not always economically feasible (Mitsch and Gosselink, 2000). However, in situations where few other alternatives apply, these may serve as low cost alternatives to chemical treatment or downstream water pollution.

While municipal and industrial wastewater treatment has improved due to more stringent regulations in recent years, agricultural and urban nonpoint pollution is perhaps the most difficult to regulate and control (Kronvang et al., 2005; Reinhardt et al., 2006). Studies involving the treatment of nonpoint source pollution, including stormwater and agricultural runoff, do not exist in as great numbers and are generally not understood to the degree of other pollutant sources (Mitsch and Gosselink, 2000). Unlike others, nonpoint sources are highly variable in pollutant type and concentration and are dependent on seasonal variation and land usage. Nutrients such as nitrogen (N) and phosphorus (P), in the form of artificial and inorganic fertilizers, are commonly applied to agricultural land and urban landscapes in order to increase productivity within these systems. Organic wastes, such as manure or sewage sludge, are also sources of these nutrients and are commonly applied for similar purposes or may derive from confined animal feeding operations. However, in areas such as the Midwestern United States, where agricultural land usage is commonly dominant, runoff potential is high, and streams may have little to no buffer zone, substantial portions of these nutrients are able to make their way into groundwater as well as surrounding lakes, rivers, and streams.

Another factor that makes agricultural runoff so difficult to control is the extensive drainage networks implemented in many of the agricultural areas worldwide. For example, 37% of the cropland in the Corn Belt and Great Lakes Regions of the United States is artificially drained by surface channels, subterranean tiles, or a combination of both (Kovacic et al., 2000). In the Eagle Creek Watershed, CIWRP research has shown that up to 80% of the cropland is artificially drained. These networks act as a major source of nitrate and phosphorus as they rapidly redirect nutrient loads to surface waters (Kovacic et al., 2000). Nutrient-rich, pedogenic water from preferential flow paths is collected by extended, artificial, subsurface drainage systems promoting lateral transport and preventing further sorption through contact with the subsurface matrix (Reinhardt et al., 2005). In the Upper Embarras River of central Illinois, subterranean tiles drain 70 to 85% of cropland and contributed an estimated 75 to 91% of the total N load to the river in 1995 and 68 to 82% in 1996 (Kovacic et al., 2000). Additionally, from 1995 to 1996, an estimated 46 to 59% of the load of dissolved P was exported by tile drains (Kovacic et al., 2000).

The contributions of pollutants from these areas to surrounding aquatic systems have been found to be a significant portion of the total input to these watersheds. In most northwestern and southern European countries, a net input of P to agricultural land is applied and therefore the total P content of agricultural soils is increasing thereby making the soil more vulnerable to loss via erosion and leaching (Kronvang et al., 2005). As a result, P loss to streams from agricultural areas in smaller catchments has been shown to be significantly higher than the P loss from similar non-agricultural catchments (Kronvang et al., 2005). While P loads have been reduced from point sources in the U.S. and Europe, P losses from agricultural areas are the main P source to many rivers, lakes, reservoirs, and coastal waters (Kronvang et al., 2005). Increased agricultural production and livestock densities has caused diffuse pollution to increase and, in some areas, more than 80% of the P load has been estimated to originate from intensively used agricultural lands (Reinhardt et al., 2005). In Nordic countries, contribution of agricultural P to surface waters is estimated to be 15, 20, 17, and 41% of total P loading in Denmark, Norway, Sweden, and Finland, respectively (Uusi-Kämppä et al., 2000).

In addition to the contributions of crop lands, livestock wastewater is another major factor in the degradation of waterways adjacent to agricultural areas. One-sixth of the $175 billion dollar agricultural industry in the United States comes from animal production, many of which are large-scale operations taking place in confinement, or relatively small areas, where large per-unit-area quantities of waste are produced (Hunt et al., 1995). Types of livestock water treated by constructed wetlands include dairy manure and milkhouse wash water, runoff from concentrated cattle-feeding operations, poultry manure, swine manure, and catfish pond water (Knight et al., 2000). Potential treatment techniques need to be capable of processing both liquid and solid waste for dairy and swine operations (Hunt et al., 1995). Commonly, many simply apply waste to available land for treatment but this is often hindered due to issues such as odor, high solids content, high nutrient concentrations, and limited pumping distance (Hunt et al., 1995). Additionally, factors such as new regulations, residential development, and increased animal numbers, and available land for treatment sites may be reduced (Hunt et al., 1995).

While fewer studies worldwide have been conducted, constructed wetlands for the treatment of livestock wastewater tend to be surface flow and have been found to drastically reduce many of these pollutants (EPA, 1994; Mitsch and Gosselink, 2000). For example, livestock producers in at least 26 states are currently using constructed wetlands for livestock wastewaters (Hunt et al., 1995) and a database of 1300 operational data records, for 135 wetlands at 68 sites, state that removal rates for 5-day BOD, TSS, ammonium, total nitrogen, total phosphorus, chemical oxygen demand (COD), and fecal coliforms are potentially very high in constructed wetlands receiving such pollution (Knight et al., 2000). While such successes are possible, and even documented, constructed wetlands treating livestock wastewater are also subject to problems and failure. One such concern lies in the fact that storm events have been found to be highly detrimental in the efficiency of some constructed wetland systems as high flows are known to reduce nutrient retention and, in some instances, cause a net release of these pollutants (Mitsch and Gosselink, 2000; Reinhardt et al., 2005). While problems may exist, understanding how these systems work, and implementing the knowledge into their design, may greatly increase one's chances of constructing an efficient and sustainable wetland.

Chapter 5

Design Criteria for Agricultural Constructed Wetlands

A number of factors should be taken into consideration when designing a constructed wetland for use in an agricultural setting. Such considerations should include

- the types of wastewater the site will be treating and the unique problems associated with them,
- the total land area required for all components of the treatment system,
- hydraulic flow decreasing residence time and affecting
    - BOD reduction,
    - pollutant transport rate into and through the system,
    - odor,
- losses to evapotranspiration and seepage,
- climate, and
- potential pre- and post-treatment stages (Hunt et al., 1995; Cronk, 1996).

Constructed wetlands may be useful when implemented as a component of a farm-wide waste management plan and, while cost is low, sites must be maintained in order for the initial investment to be successful and several design modifications may be necessary before effective treatment is obtained (Cronk, 1996). Best Management Practices for erosion control and waste handling in addition to onsite constructed wetlands, nutrient-sediment control systems in small watersheds, and natural wetlands along streams and at strategic locations in large watersheds may provide low-cost, efficient control (Hammer, 1992).

While other concerns certainly exist, the main objectives of the constructed wetland established in agricultural areas are commonly to intercept, retain, and reduce the loads of suspended solids and the various forms of phosphorus and nitrogen travelling from the agricultural area to the surrounding bodies of water (Liikanen et al., 2004). As mentioned earlier, in these settings, surface flow wetlands are generally implemented. These systems are generally made up of a combination of up to four components including

- sedimentation ponds,
- wetland filters,
- overflow zones generally covered in vegetation or stones, and
- outlet basins (Braskerud, 2005).

As a result of the high erosion potential for sediment, particulate bound nutrients, such as phosphorus, and solid waste from livestock, sedimentation ponds may be an integral component of any surface flow wetland. Especially in the case of livestock operations, treatment for solids removal is needed ahead of the wetland in the form of mechanical solids separation, stack pads with leachate collection, or collection in lagoons for settling and treatment thereby ensuring the wetlands do not receive an overabundance of carbon that affects the balance between aerobic and anaerobic processes supporting nitrification and denitrification (Hunt et al., 1995). In addition, sediment burial is considered to be the major long-term P storage in wetlands (Liikanen et al., 2004) and, while denitrification is commonly a primary component in the removal of nitrogen in these systems (Kovacic et al., 2000; Reinhardt et al., 2006), in some studies it was determined to be minor removal mechanism (< 1%) while settling and increased storage was the largest removal mechanism (Newman et al., 1999).

5.1 Vegetation

In addition to encouraging the settling of particulate matter, other remediation techniques such as vegetative uptake and adsorption by sediment may be planned into the construction of an agricultural wetland. The results obtained through various techniques of planting and managing vegetation in these systems are varied and, therefore, the topic remains one of debate amongst researchers.

For example, the types of plants used in constructed wetlands are often chosen based strongly on the type of waste being treated. Free-floating plants have been found to be effective for treatment of raw sewage and primary and secondary effluents while submersed plants have been found useful in *polishing* or treating tertiary municipal effluent (Hunt et al., 1995). Emergent plants are most commonly used in constructed wetlands for agricultural wastewater treatment as the roots and rhizomes provide surfaces for bacterial growth, the filtration of solids, nutrient uptake, and as source of oxygen to anoxic soil environments in turn promoting both nitrification and denitrification (Hunt et al., 1995). However, the implementation of plants according to effluent type is debatable as one study reported that while macrophytes within the system were found to improve reductions in treating municipal and industrial wastewater, the type of vegetation did not affect the results (Jing et al., 2002). Regardless, the most commonly used emergent macrophytes in the treatment of agricultural wastewater are generally *Scirpus, Typha, and Juncus* or bulrushes, cattails, and rushes respectively (Hunt et al., 1995).

Vegetation aids in the retention of pollutants in several direct and indirect ways. Relating back to sedimentation, some studies have shown that dense vegetation increases the hydraulic roughness. This, in turn, decreases overland flow velocity and sediment transport capacity thereby aiding in the sedimentation of particles and the removal of P from cropland runoff (Uusi-Kämppä et al., 2000). Sediment and P loading and retention generally increase with runoff (Braskerud, 2005) and increased retention time is important for settling (Uusi-Kämppä et al., 2000; Braskerud, 2005). Vegetation also increases sediment retention by decreasing resuspension of sediments in shallow waters (Uusi-Kämppä et al., 2000; Braskerud, 2001; Liikanen et al., 2004; Reinhardt et al., 2005; Braskerud, 2005). Vegetation produces a local reduction of turbulence and water velocity which increases retention time under high water velocity compared to those without (Uusi-Kämppä et al., 2000). Finally, the leaves and stems of macrophytes may additionally create local deposition surfaces for sediment (Uusi-Kämppä et al., 2000). However, as vegetation cover increases, a level is reached where hydraulic load and sediment load may have greater influence on retention performance (Braskerud, 2001).

Vegetation may also be significant in taking up dissolved inorganic P at high DRP loads (Liikanen et al., 2004). The large surface area and short reproduction time associated with finely dispersed phytoplankton is able to eliminate P efficiently and respond quickly to changing DRP concentration (Reinhardt et al., 2005). Finally, planted wetlands showed significantly improved removal rates for CBOD at higher loadings when compared to non-planted wetlands and a 1.3 to 2.6 fold higher mass removal for total BOD (Tanner, Clayton, and Upsdell, 1995 I)

The use of vegetation in wetlands does not always significantly reduce the nutrient removal or retention capacities within these systems. In many studies, the macrophyte component of constructed wetlands is thought to be an insignificant sink for major contaminants such as phosphorus (Lu, Zhang, and Xiang, 2006). This may be the result of a number of factors. In addition to a portion of P being immobilized in biomass, some wetlands store large amounts of organic matter and therefore also sequester a large quantity of P bound to organic molecules (Liikanen et al., 2004). However, in both cases, a portion of the retained P can be converted into dissolved inorganic P which can subsequently be partially released back to water (Liikanen et al., 2004). Similarly, duckweed (*Lemna sp.)* production provided only a short term N sink and fed internal N recycling as the settling matter, most of which was duckweed, had a higher N content than the sediment suggesting 80% of the settled N was released during benthic mineralization (Reinhardt et al., 2006).

Another factor affecting the retention of nutrients in these systems lies in the fact that nutrient assimilation by plants follows their seasonal growth pattern which is maximal during the growing season in the summer (Liikanen et al., 2004). However, in many areas, such as the boreal region, the maximal load of P and growth rate of plants do not match as the highest nutrient load occurs in spring when the growth of plants is negligible and, during summer months, when the biomass production is highest, the flow of water and nutrients is lowest (Uusi-Kämppä et al., 2000; Koskiaho et al., 2003; Liikanen et al., 2004). As a result, biomass only represents short-term P storage and the turnover time for P in aboveground parts of macrophytes varies from months to the annual growing season (Liikanen et al., 2004). Seasonal variations, such as temperature and flooding, may also affect retention capacities. Low water temp and short residence time limited microbial activity and plankton assimilation and explains low or negative rates in the winter months (Reinhardt et al., 2006). A constructed wetland affected by cold climate became a net source of ammonia following plant die back in fall, and while concentration declined over time, the mass retention was significantly greater during the summer than during the winter for all variables except fecal coliform (Newman, Clausen, Neafsey, 1999). In another case study, macrophytes were found to play only a minor role in short term P retention as P loading was mainly driven by short term flood events, but may have been governed by settling phytoplankton and floating microalgae (Reinhardt et al., 2005).

As a result, plant efficiency to remove P has been reported to be greatest at low P concentrations (Liikanen et al., 2004). Reported high rates of plant uptake of phosphorus where P was present at relatively low concentrations may suggest macrophytes to be especially useful in obtaining low residual P concentrations in final effluents (Lu, Zhang, and Xiang, 2006). Emergent macrophytes take up P already in sediments with root networks but their reactivity to incoming short-term P loads is limited (Reinhardt et al., 2005). However, they may contribute to building up long-term P storage and provide the substrate for periphyton growth (Reinhardt et al., 2005). Regardless, planted wetland sediments generally retain P more efficiently than unplanted ones (Tanner et al., 1995 II). Plant biomass and tissue nutrient levels sampled to evaluate plant nutrient uptake showed that in planted wetlands, the mean annual removal rates increased gradually with mass loading rates while the unplanted showed a marked decline in removal at high loadings (Tanner et al., 1995 II). Net storage by plants in the first year of monitoring accounted for between 3 and 20% of the greater N removal and between 3 and 60% of the greater P removal in the planted wetlands (Tanner, Clayton, and Upsdell, 1995 II). Despite this fact, P retention by vegetation is bound to the seasonal growth pattern may not be as singlehandedly effective as P retention by chemical sorption reactions which are fast and occur year round (Liikanen et al., 2004).

Additional controversy lies in the debate as to whether or not one should intermittently harvest wetland vegetation and/or dredge sediments as to permanently remove the immobilized nutrients from the system. In addition to the problem of disposing of these materials, harvesting may reduce the potential for the increased settling of sediments (Uusi-Kämppä et al., 2000). However, positive results from doing so have also been documented. The cutting and removal of plant cover can remove P and, in contrast, if the plant is left decaying in place, some P may later be released in a dissolved form (Uusi-Kämppä et al., 2000). In one study, 58% of TP was removed by plant uptake and by biannual harvesting and seed transport (Lu, Zhang, and Xiang, 2006). Harvesting of floating leaved plants stimulates phytoplankton growth and oxygen production while further adding to the removal of the recyclable biomass N pool (Reinhardt et al., 2006). However, complete biomass and sediment removal should be avoided to provide sufficient organic carbon substrate for denitrification (Reinhardt et al., 2006).

5.2 Sorption to Sediments

P bound to inorganic sorption components may be a more long-term storage for P (Liikanen et al., 2004). Inorganic P is retained by Ca in calcareous soils and calcite, clay minerals, organometallic complexes, Fe and Al oxides and hydroxides, and oxalate-extractable iron (Hunt et al., 1995; Uusi-Kämppä et al., 2000; Liikanen et al., 2004). Sorption/desorption of P is an equilibrium reaction governed by the amount of these materials, their P saturations, and soil pH measurements (Uusi-Kämppä et al., 2000). Low salt concentration in the runoff water and high water to soil ratio favors desorption of P from soil to water and low temperatures may also influence P sorption/desorption (Uusi-Kämppä et al., 2000). Wetlands constructed on mineral soils may be more efficient in retaining P than natural wetlands where a high content of organic matter often diminishes the amount of active sorption components (Liikanen et al., 2004). Mineral wetland soils containing Al and Fe have been reported to have optimal P retention capacity in wetlands without vegetation (Liikanen et al., 2004). Even at high input concentrations, the high P sorption capacity of constructed wetland soils appeared to promote retention (Koskiaho et al., 2003). Additionally, at high hydraulic loads, the redox potential in the outlet water was kept as aerobic and therefore the redox-sensitive P in the wetland sediment is conserved as long as sufficient amounts of water flowed through the wetland (Braskerud, 2005).

Various factors may reduce a wetland's retention capacity in these situations. Iron-bound P is sensitive to changes in the redox potential and during anoxic conditions it can be released although part of the mobilized P can be resorbed by Al oxides (Liikanen et al., 2004). At elevated DRP loads, retention may diminish under anoxic flow because Fe^{3+} may at least partially be reduced to Fe^{2+} (Likannen et al., 2004). However, wetland sediments are also able to retain P under anoxic conditions indicating that high amounts of Al oxides may also play an important role as a sorption component due to their capacity not to dissolve at low redox values (Liikanen et al., 2004). P bound to Al oxides, however, can be partially replaced by OH- ions produced as a result of an increase in pH with lowering redox potential (Liikanen et al., 2004). Also, with time, the amorphous and poorly crystalline forms of Al, Fe, and Mn with which retention of P is reported to correlate within wetland soils may become more saturated, thus reducing retention capacity (Liikanen et al., 2004). A rapid decline of P removal efficiency in treating swine wastewater fell from 99 to 78% in one year as reported by one study which was likely due to high content of poorly crystalline iron oxi-hydroxides of the wetland soil, strong soil reduction, and high load of P (Hunt et al., 1995). The addition of iron or aluminum may have increased efficiency in this case (Hunt et al., 1995).

5.3 Other Design Considerations

Constructed wetland design must include adequate area to meet water quality goals (Knight et al., 2000). Retention of pollutants, such as TP, increases with increasing ratio of constructed wetland or pond area to watershed area (Uusi-Kämppä et al., 2000; Kovacic et al., 2000). Treatment wetlands with a watershed to wetland ratio of approximately 15 to 20, or approximately 5%, were commonly suggested throughout the literature review (Kovacic et al., 2000; Reinhardt et al., 2005). However, constructed wetlands of even 0.1% watershed area were reported to contribute to cleaner waterways (Braskerud, 2005). In addition, wetlands constructed on 50% of the tile systems, if applicable, would have the capacity to remove 18-23% inflow N (Kovacic et al., 2000).

The ratio of wetland size to watershed area is a major factor in part due to its implications on retention time. Retention time of incoming water was by far the most commonly reported influence on the retention of the entire spectrum of the reported pollutants in these studies. In one study, the longest residence time was associated with best performance while the shortest residence time functioned only occasionally and was a net source for DRP and the combination of nitrite and nitrate (Koskiaho et al., 2003).

Another reported that a retention time of 1.7 to 3.8 days had significant correlation with P removal efficiency while water temperature, inflow P load, inflow P concentration, and hydraulic load rate had little influence on P removal efficiency (Lu, Zhang, and Xiang, 2006). Retention of bioavailable P increased significantly with water residence time (Reinhardt et al., 2005). If residence time exceeded 10 days, removal ranged from 50 to 90% but decreased or was negative if residence time felt to any less than 5 days (Reinhardt et al., 2005). Finally, mean mass removal of CBOD increased from 60-75% to 85-90%, total BOD (CBOD + NBOD) from 50 to 80%, and faecal coliforms from 90-95% to >99% with increasing wetland retention time during first 12 months, though mean annual suspended solids removals of 75-85% were recorded irrespective of loading rate (Tanner et al., 1995 I). High levels of dissolved humic color in wastewaters were only slightly affected by passage through the wetland at short retention times, but were reduced by up to 40% at longer retentions (Tanner et al., 1995 I). Even in wetland systems treating municipal and industrial effluents that were much smaller in scale than normal surface water flow systems, high retention capabilities were achieved with increased residence time (Jing et al., 2002).

Higher hydraulic loading rate and subsequent pollutant mass loading rate may also decrease retention time and, in turn, the removal of pollutants (Knight et al., 2000; Liikanen et al., 2004; Reinhardt et al., 2005). Typical nutrient retention wetlands are able to retain more than 50% DRP load, if residence time exceeds 7 to 10 days. In contrast, a constructed wetland with retention times of less than 7 days for 60% of the time was studied by Reinhardt et al. (2005). In their case study, 43% of the DRP load came during only 65 days of the year during high discharge events causing residence times of less than 3 days and average retention efficiencies of -34% (Reinhardt et al., 2005). As a result, annual retention of bioavailable P was only 2% while overall retention of TP was only 23% as the result of particulate settling (Reinhardt et al., 2005). Outflow levels of CBOD, suspended solids, and fecal coliforms rapidly mirrored changes in influent loadings and mass removals of each showed monotonic relationships to mass loading rate with little variation between planted and unplanted sites except for CBOD at high loading (Tanner et al., 1995 I).

Reinhardt et al. (2005) suggest that in order to ensure a minimum residence time of 7 days, the storage volume should exceed expected maximum daily water discharge approximately 7 fold. Additionally, a wetland able to retain about half of its agricultural DRP load requires a surface area that equals about 4% of its catchment area though any device able to lower export of phytoplankton would increase P retention and reduce the necessary surface area (Reinhardt et al., 2005).

Various other suggestions associated with constructed wetland design are proposed in this collection of studies. Shallow wetlands, less than or equal to 0.5 m in depth, were found more effective than deeper ponds in removing phosphorus due to the reduction in time required to settle to the wetland bottom (Uusi-Kämppä et al., 2000; Braskerud, 2005). The removal of topsoil on former arable land was found to be essential to achieve a good P removal efficiency since it contained high amounts of P (Liikanen et al., 2004). Flooding of this soil in the constructed wetland may have lead to release of P into flowing water and the site to become a potential source of P rather than a sink (Liikanen et al., 2004). Pre- and post-treatment steps were also suggested as important in total pollutant removal. Wastewater with very high BOD commonly exceeds the capability of the wetland to supply oxygen and will subsequently limit treatment (Hunt et al., 1995). As a result, these wastewaters may require dilution, anaerobic pre-treatment, or very low loading rates (Hunt et al., 1995; Cronk, 1996). Oxygen also varies with season and time of day due to differences in temperature as well as light and dark periods (Hunt et al., 1995). Successful design must also include adequate pretreatment as to protect the health of wetland biota (Knight et al., 2000). Issues with ammonia concentration and toxicity to plants can be managed by controlling rainwater and lot runoff in addition to dilution with fresh-water-recycled effluent (Hunt et al., 1995).

Since no direct stream discharge of animal wastewater is allowable by law due to the high loading rates needed for substantial mass removal, effluent must be reused or applied to land through irrigation, directly to surrounding land, or further treated in receiving ponds or lagoons before being discharged, reused as fresh water, or recycled to wetlands (Hunt et al., 1995). Application to crop lands and vegetative strips or woodlands do not require discharge permits or monitoring for water quality and therefore fit very well within the capacities of constructed wetlands (Hunt et al., 1995). A study described by Uusi-Kämppä et al. (2002) investigated the use of overland flow in vegetative strips composed of varying lengths of grass and trees in which the soil permeability caused overland flow and lateral subsurface flow. These overland flow strips were found to be good potential pre- or post- wetland components of functional and sustainable treatment systems due to the substantial nutrient assimilative capacities of grass filter strips and wooded riparian zones. (Uusi-Kämppä et al., 2000). The creation of wetlands formed an adjacent riparian buffer that was effective in removing an additional 9% of nitrate, through denitrification, from the 15% berm seepage water component thereby raising the total nitrogen removal to 46% of input in the study by Kovacic et al. (2000).

Chapter 6
Conclusions

As is evident in the results described above, constructed wetland systems are a valuable option in the treatment of wastewater. Able to reduce concentrations of a variety of aquatic pollutants, including nutrients, toxins, and pathogens, derived from a variety of sources, constructed wetland systems may also be less expensive to build and maintain, create habitat for flora and fauna that have been forced out of many areas as the result of land use change, and create a more aesthetically pleasing option of wastewater treatment. Continued research lies in the study of effective design and implementation of these systems. Mass balances may also be needed as different pollutants tended to collect in different portions of the wetland system (i.e. substrate, organic matter, vegetation). In addition, research of holding capacities of these systems may also be useful in their future management.

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