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Application Of Artificial Treatment Wetland Systems In The English Coulee

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APPLICATION OF ARTIFICIAL TREATMENT WETLAND SYSTEMS IN THE ENGLISH COULEE

by

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ABSTRACT

Artificial treatment wetlands are commonly used for treating urban stormwater runoff and surface waters affected by agricultural runoff. While this method is well-established, its effectiveness can vary based on environmental, design, and hydraulic factors. In Grand Forks, ND, the English Coulee, a hydraulically altered class III stream, is currently experiencing elevated nutrient pollution, specifically nitrates and phosphates. Implementing a floating treatment wetland (FTW) system in the English Coulee is a potential solution. This paper analyzes the feasibility and requirements for a 4.64 ha FTW to achieve a minimum of 50% nutrient removal, utilizing comprehensive modeling approaches. Through HEC-HMS and HEC-RAS simulations, along with nutrient uptake and adsorption models, the study provides insights into the design specifications needed to meet nutrient removal objectives. However, due to the English Coulee's unique cold region hydrology and flood patterns, implementing and maintaining a wetland of this size would be impractical when compared to alternative methods for addressing its water quality issues.

Chapter 1: Introduction

The English Coulee, a small stream in Grand Forks County, North Dakota, originates from a low-head dam just outside the city of Grand Forks. It meanders through farmland, passes through the University of North Dakota campus, and eventually empties into the Red River. Figure 1 shows the location of the English Coulee and its geographic extent.

The English Coulee is a tributary of the Red River which flows through the Red River Valley, which, despite being referred to as a 'valley,' exhibits a very flat topography. This flatness results from the area's history as the lakebed of the ancient glacial Lake Agassiz. The Red River Valley is prone to significant flooding due to this flat terrain and seasonal precipitation patterns (Wilson et al., 2005). Heavy precipitation, spring snowmelt, and ice jams collectively contribute to flooding within the Red River watershed which includes the English Coulee. Consequently, significant modifications have been made to the English Coulee's hydrology to mitigate flood risks to the city of Grand Forks, which will be discussed in more detail later in this chapter.

Before the flood control measures were constructed in the 1990's, several stock dams were constructed in the English Coulee during the 1930s. These dams were not initially intended for flood control but rather to create water reserves for agricultural purposes (AE2S, 2022). In its natural state, the English Coulee would have exhibited higher flow rates in the spring and lower flow rates in late summer and fall.

The English Coulee is classified as a Class III stream, signifying that it experiences seasonal flow patterns, lacks permanent fish populations, and has a notable impact on downstream water quality due to sediment (NDDEQ, 2016). Several structures and diversions are in place within the coulee, each serving specific hydraulic functions outlined in this section.

In 1990, the Soil Conservation Service erected a low-head dam designed primarily for flood control purposes (Benson, 1994). Additionally, in 1984 the North Dakota State Water Commission completed a diversion channel in the English Coulee which can be seen in Figure 1 (Benson, 1994). Further details regarding this dam will be provided later in this section. Additionally, in 1997, the Army Corps of Engineers constructed several structures as part of a comprehensive flood protection system (AE2S, 2022). These constructions encompass a diversion structure, diversion channel, and levee closure. Their function is to redirect excess water from the primary channel around the city of Grand Forks and discharge it into the Red River just north of the city during periods of high flow (Benson, 1994).

Apart from flood control infrastructure, multiple stock dams were established within the English Coulee. Unlike flood control measures, these structures were originally designed to maintain a reservoir of surface water and were initially installed in the 1930s to meet agricultural water requirements (AE2S, 2022). Figures 2 through 4 display some of the hydraulic structures mentioned. A more comprehensive examination of the stock dam, situated on the University of North Dakota's campus, will be undertaken in Chapter 3.

These structures play a crucial role in mitigating flood damage during the spring, when flow and water levels significantly increase due to snowmelt and precipitation. However, as the season progresses into summer and fall, the flow within the English Coulee becomes severely restricted, resulting in near-stagnant conditions. This issue can be attributed to several factors related to the hydraulic structures.

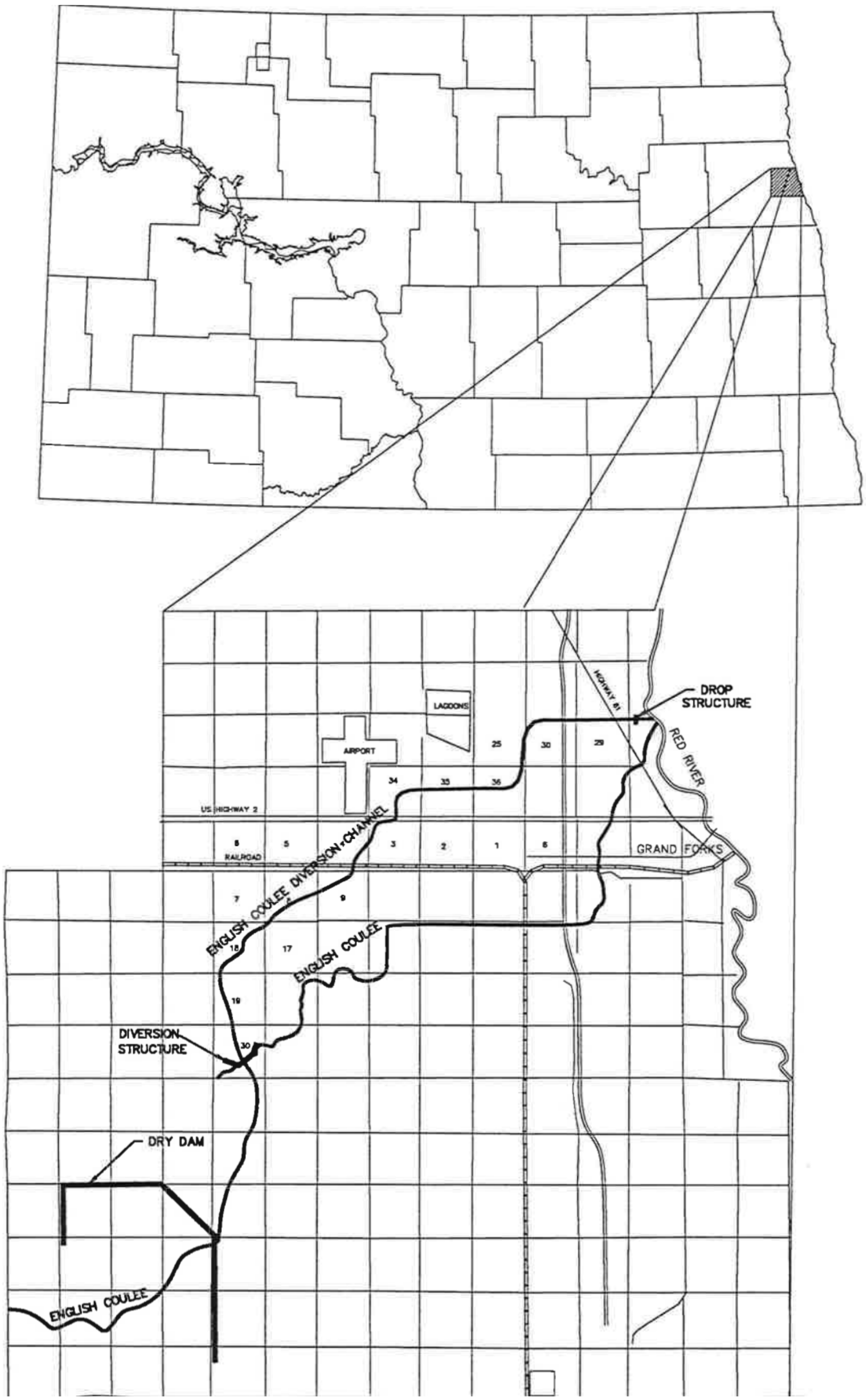

Figure 1. Location Map reprinted from "HYDRAULIC ANALYSIS OF THE ENGLISH COULEE DIVERSION" by the North Dakota State Water Commission, by Benson, B. (1994), p.2.

At the low-head dam, the intake structure may be positioned too high, which limits the intake from the reservoir that supplies the English Coulee. This dam, categorized as a dry dam and designed exclusively for floodwater retention, should have its principal spillway inlet set at the submerged sediment pool elevation. This adjustment should ensure that the reservoir can drain within a reasonable timeframe, following the Natural Resources Conservation Service (NRCS) TR-60 guidelines for earth dams and reservoirs. The volume allocated for sediment storage should be commensurate with the intended 100-year lifespan of the dam and the expected sediment loading from upstream. Based on data obtained from a United States Geological Survey (USGS) stream gauge located 2 miles upstream of the dam from 2008 to 2009 (USGS, 2023), the annual average sediment loading rate was calculated at 0.111 metric tons per day. Over the dam's 100 year intended service life, this amounts to 3,151.29 m^3 of sediment.

Original construction documents and a recent NRCS report reveal that the weir for the primary spillway crest is positioned at an elevation of 274.32 m in NAVD88, providing a storage capacity of 353,170.48 m^3, which exceeds the dam's requirements. The NRCS dam assessment also notes obstructions at the auxiliary spillway and primary spillway outlets due to dense cattail growth (NRCS, 2021).

Another hydraulic structure affecting flow in the English Coulee is a stock dam, maintaining a specific water level during drier periods of the year, as shown in Figure 4 below. Although initially installed to support agricultural development, an examination of active surface water diversion permits indicates that these stock reservoirs are no longer used for this purpose (NDDWR, 2023).

The English Coulee faces a challenge with elevated nutrient levels in its water. These excess nutrients enter the system primarily through runoff from the surrounding watershed. The

English Coulee, with a watershed covering approximately 347 km^2 (NDDEQ, 2016), experiences an annual loading rate of 2.12 metric tons per year of total phosphorus and 2.115 metric tons per year for total nitrogen (AE2S, 2020). The dominant land use in the English Coulee watershed is row crop agriculture (NDDEQ, 2016). In regions characterized by significant agricultural and urban development, nutrient loading—specifically nitrates and phosphates from fertilizers in agricultural runoff—tends to be substantially higher (Delkash et al., 2018).

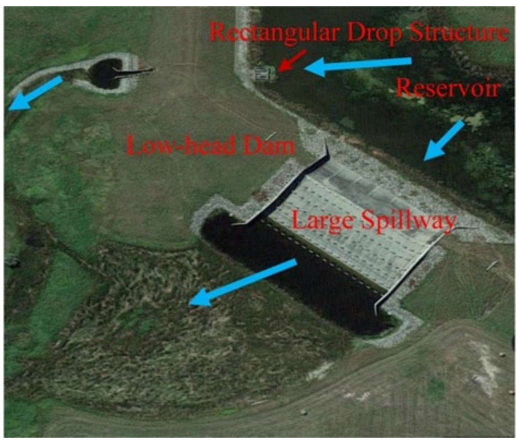

Figure 2. Low-head Dam. Adapted from "Revitalization of an Urban Stream in Terms of Water Quality and Recreational Usability: Assessment of Options through Hydrologic and Hydraulic Modeling" by Lim, H., et al. (2018), World Environmental and Water Resources Congress, p.130-139. Aerial imagery provided by Google Earth, 2018.

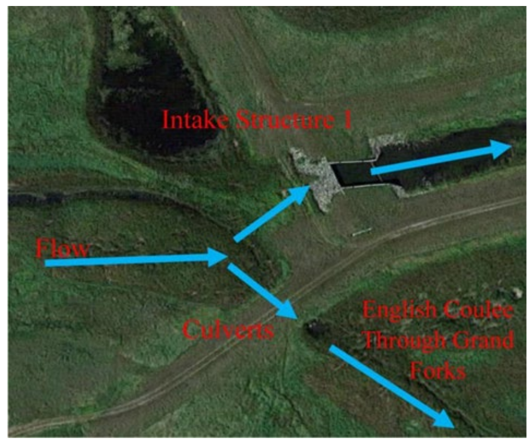

Figure 3. Diversion Channel. Adapted from "Revitalization of an Urban Stream in Terms of Water Quality and Recreational Usability: Assessment of Options through Hydrologic and Hydraulic Modeling" by Lim, H., et al. (2018), World Environmental and Water Resources Congress, p.130-139. Aerial imagery provided by Google Earth, 2018.

Figure 4. Stock Dam. Photograph by Michael Rosati. Taken at stock dam on the UND campus on July 17th, 2021.

Both agricultural and urban runoff can contribute to increased nitrate and phosphate loading in the surrounding water bodies (Delkash et al., 2018). One natural defense against this issue is the use of riparian buffer strips, which are strips of natural vegetation along the banks of rivers or streams and serve as a common method for controlling nonpoint source pollutants (Lee et al., 2003). These natural riparian areas act as buffers against incoming sediments, nitrates, and phosphates (Kaushal et al., 2014).

Over the past two decades, the riparian zone along the English Coulee has remained relatively stable, as demonstrated in Figure 5 and 6 below. These Figures illustrate the results of an NDVI analysis conducted over this period, highlighting changes in vegetation health and density. This analysis was performed using ArcGIS Pro, developed by Esri in Redlands, California. The results, as depicted in the Figures 5 through 7, show that there has been minimal change in the land around the English Coulee between 2003 and 2023. Despite the lack of significant change, a substantial portion of the land surrounding the English Coulee has been converted to cropland.

Within the riparian zones along the English Coulee, several common plant species can be found, as indicated by data from the National Wetlands Inventory and the US Fish and Wildlife Service. These species include *Hordeum jubatum*, *Rumex* spp., *Scolochloa festucacea, Polygonum* spp., *Alisma* spp., *Typha* spp., *Scripus acutus*, *Salix* spp., and *Populus deltoides* (USFWS, 1987). When present, these buffer systems along the banks can significantly reduce nutrient inputs from runoff, as well as the presence of total suspended solids (TSS) in the stream. Studies have demonstrated that such buffers can remove up to 62% of nitrates and up to 58% of phosphates when they contain mixed native vegetation (Lee et al., 2003).

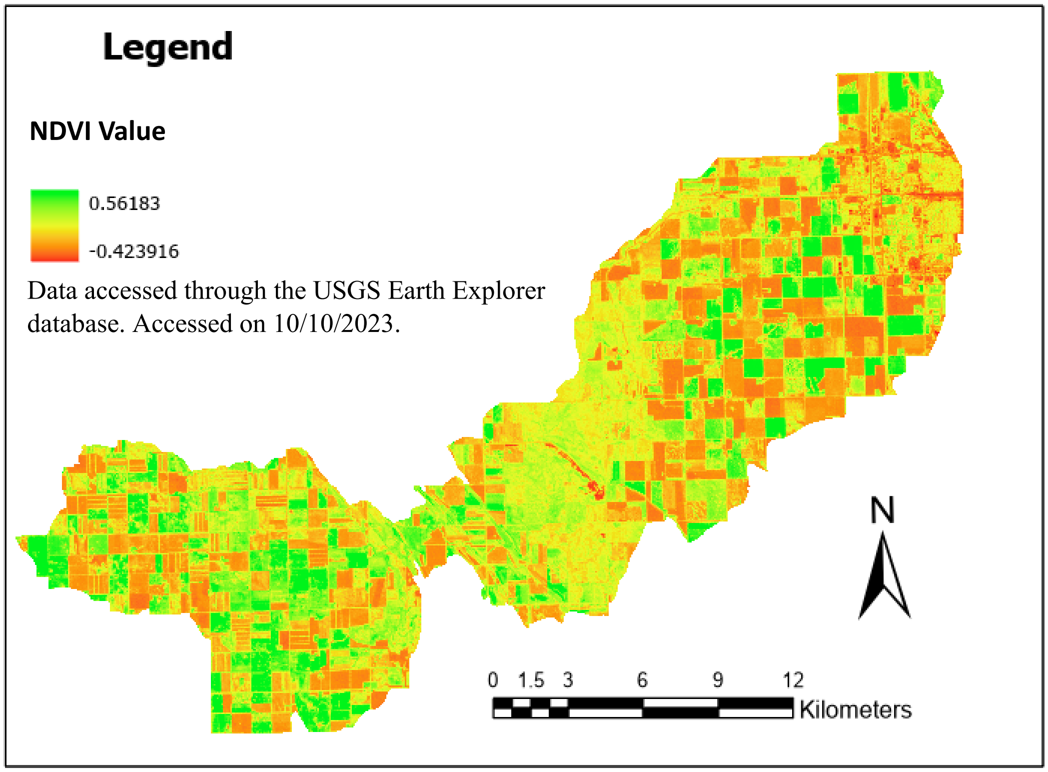

Figure 5. NDVI Analysis of the English Coulee Watershed in the Year 2000 (Fall).

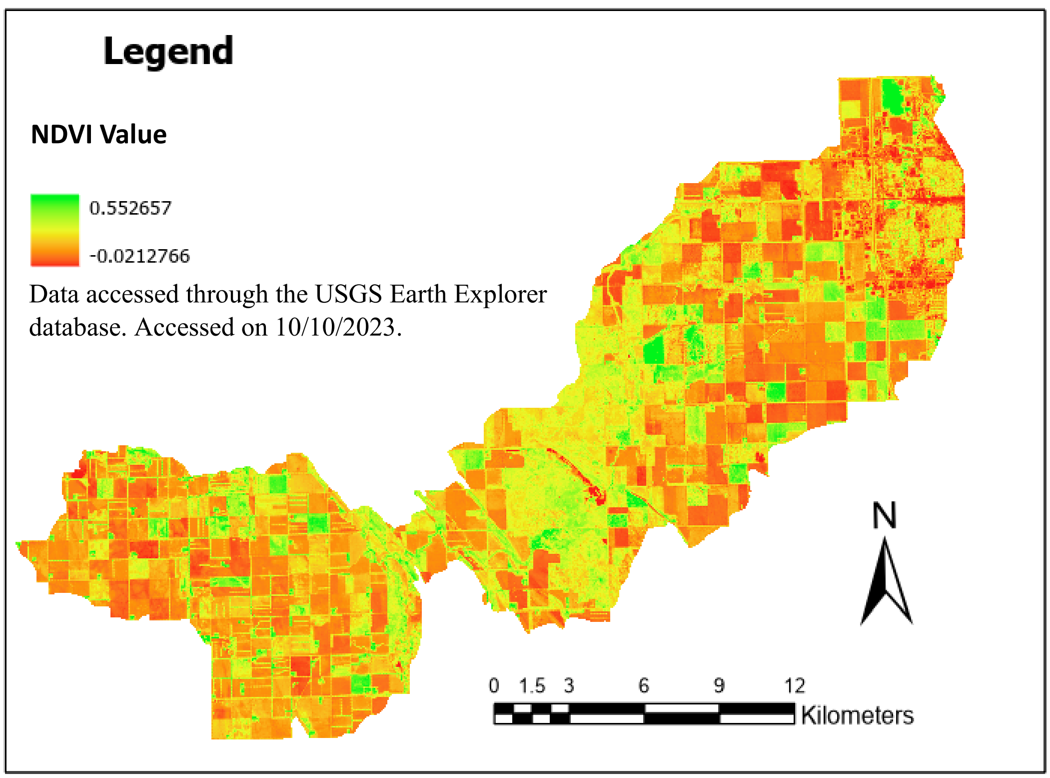

Figure 6. NDVI Analysis of the English Coulee Watershed in the Year 2023 (Fall).

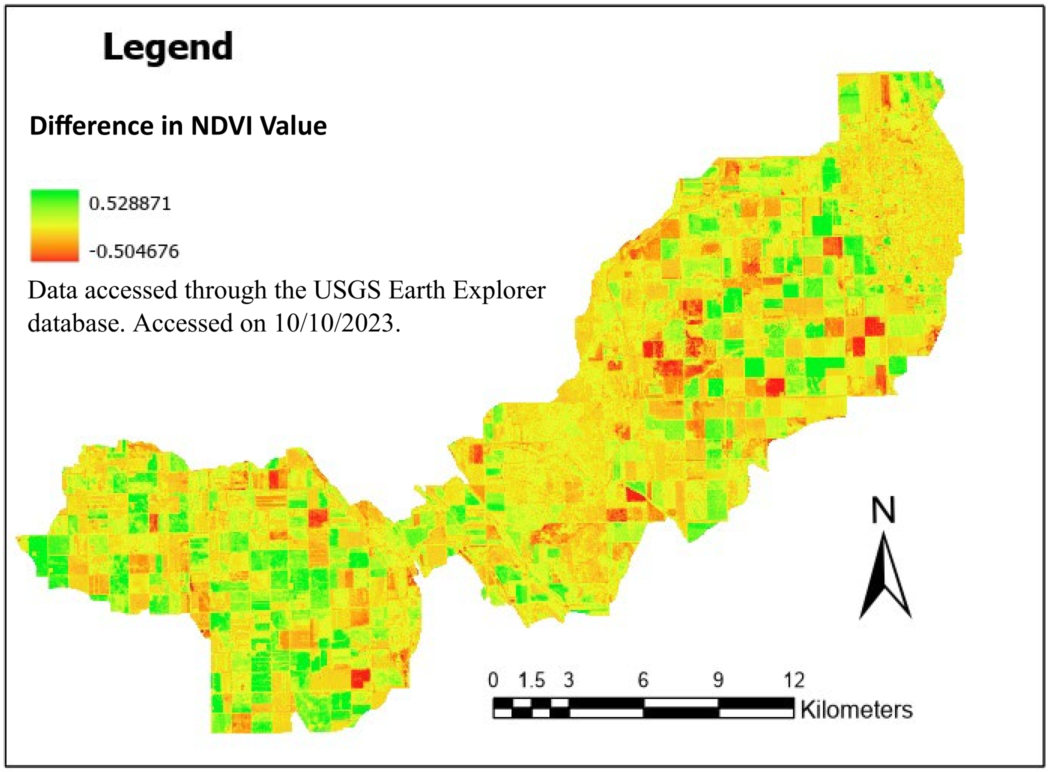

Figure 7. Difference in NDVI Analysis 2000 to 2023.

The influx of nutrients from runoff, coupled with low flow velocity, has created ideal conditions for algal blooms (Long et al., 2011; Wang et al., 2015). These blooms can lead to eutrophication, discoloration, and unpleasant odors (Anderson et al., 2012). Consequently, there have been frequent community complaints about the English Coulee's odor, and a study by the North Dakota Department of Health found low levels of dissolved oxygen (NDDEQ, 2019).

The city of Grand Forks is presently in the second phase of a three-phase plan aimed at addressing challenges within the English Coulee. The initial phase primarily focused on gauging public interest and determining the community's aspirations for the English Coulee. This phase was designed to achieve three key objectives. Firstly, it sought to gauge community interest and discern public opinion concerning enhancements to the English Coulee. Secondly, it aimed to comprehend the water quality issues within the English Coulee and assess their extent. Lastly, it attempted to assess flood protection considerations with any alterations proposed for the flow in the English Coulee.

The outcomes of these initial phase goals revealed a public consensus that the water quality in the English Coulee is poor and in need of improvement. Various factors were identified as contributing to the impaired water quality, with notably high concentrations of phosphorus in the sediment. Furthermore, it was determined that the FEMA flood models were deemed inaccurate, prompting a decision to exclude them from the project moving forward.

The subsequent phases will entail the development of a more detailed hydraulic model to gain a better understanding of potential modifications to the English Coulee. Additionally, there will be a focused effort on further evaluating sediment deposits within the English Coulee and formulating a comprehensive plan for sediment removal. For detailed information and updates regarding these phases, please refer to the City of Grand Forks' website (City of Grand Forks, 2021).

This paper explores the use of artificial floating treatment wetlands (FTWs) as a potential solution for the English Coulee's problems. FTWs are a type of man-made wetland characterized by a floating platform on the water's surface, allowing for the growth of aquatic plants on its surface. The roots of these plants extend into the water column. By emulating natural wetlands, artificial treatment wetlands have the potential to reduce nitrate and phosphate concentrations, mitigating algae growth and decay, which contribute to odor, discoloration, and low dissolved oxygen levels (Anderson et al., 2012).

Chapter 2: Literature Review

2.1 Treatment Mechanisms

The goal of FTWs is to emulate natural wetland processes to enhance water quality by removing excess nutrients and filtering out pollutants. This is achieved through physical and biological processes, including sedimentation, microbial and macrophyte action for biological removal, and chemical removal via adsorption (Greenway, 2007). See Figure 8 below for an overview of these processes.

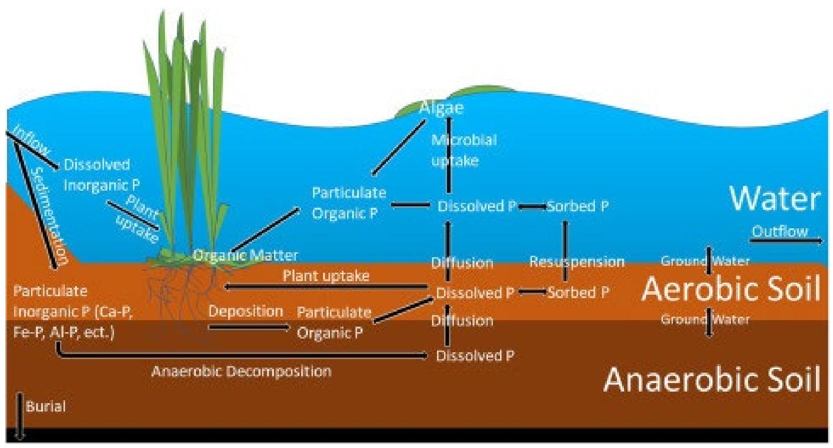

Figure 8. Nutrient Cycling in Wetlands. Adapted from "Fundamentals of Tropical Freshwater Wetlands" by Deemy, J., et al. (2021), Chapter 6, Nutrient Cycling, p.133-160

One of the primary mechanisms for nutrient removal in wetlands is nutrient uptake, where macrophytes directly use nutrients for growth by absorbing them through their root systems. For FTWs, the overall removal rate depends on the macrophytes used and the environmental conditions, with macrophyte uptake capacity ranging from 1000 to 2500 kg N/ha/yr for nitrogen and 50 to 150 kg P/ha/yr for phosphorus (Brix, 1994).

Sedimentation is another critical mechanism for removing solid particulates. Gravity causes sediments to settle as water enters the system. The removal efficiency depends on channel bed length, fluid velocity, fluid density, fluid viscosity, and particle size. Stokes' equation is commonly used for laminar flow at lower velocities, while for higher flow velocities with turbulent behavior, a different settling model, such as Newton's Regime, must be considered (Tchobanoglous, 2014; Crittenden, 2012). Wetlands can dissipate kinetic energy from the flow, enabling more particles to settle (Brix, 1997, 1994).

Adsorption is a related physical mechanism where pollutants are trapped on the surface of particles. This process is driven by the ionic and charge characteristics of molecules. Pollutants are attracted to the surface of sediments in the substrate or suspended in the water column, leading to their removal from the water. This attraction relies on ionic forces generated by polar ends of molecules or overall net charge (Kajjumba et al., 2019). Refer to Figure 9 for a visual representation.

In the case of FTWs, adsorption plays a crucial role in removing pollutants. Depending on the chosen medium and macrophytes, it can be responsible for high removal rates, such as up to 96.3% for total phosphorus and 97.8% for total nitrogen in natural water studies (Shen et al., 2021). This process extends to the removal of toxic metals like lead, arsenic, and mercury, among others (Shahid et al., 2020). Metals can directly sorb to the organic components in peat, while nitrates and phosphates attach themselves to trace amounts of iron and aluminum in the substrate. Iron oxides can be added to the substrate to enhance adsorption capacity, a common practice in wastewater treatment, or the natural iron and aluminum in the substrate can be relied upon (Borggaard et al., 2005).

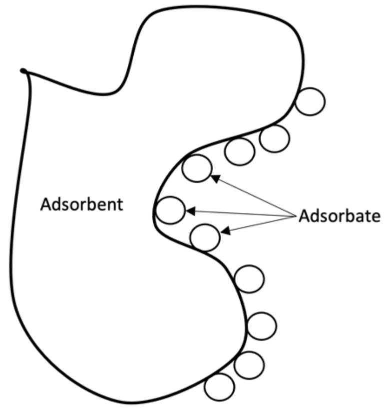

Figure 9. Adsorption Mechanics.

The primary biological mechanism for nutrient removal relies on the microbial community within the substrate and macrophyte rhizomes (Greenway, 2007; Shahid et al., 2020; Wang et al., 2014). Natural wetlands typically maintain soil saturation and limited oxygen, favoring anaerobic respiration. In this process, oxygen is replaced by other molecules as terminal electron acceptors in respiration. Nitrate is often the preferred molecule in such conditions. This eventually leads to denitrification, releasing N_2 gas as a byproduct (Araujo et al., 2015).

$$NO_3^- + 2\,H^+ + 2\,e^- \rightarrow NO_2^- + H_2O \quad (1)$$

$$NO_2^- + 2\,H^+ + e^- \rightarrow NO + H_2O \quad (2)$$

$$2\,NO + 2\,H^+ + 2\,e^- \rightarrow N_2O + H_2O \quad (3)$$

$$N_2O + 2\,H^+ + 2\,e^- \rightarrow N_2 + H_2O \quad (4)$$

The nitrogen removal reactions in FTWs involve enzymes provided by microbes. The initial step employs the enzyme nitrate reductase, which transfers electrons from NADH and NADPH to nitrate, reducing it to nitrite. Subsequently, nitrite reductase catalyzes the production of nitric oxide. In the next stage, nitric oxide is transformed into nitrous oxide through the enzyme nitric oxide reductase. Finally, the enzyme nitrous oxide reductase facilitates the reaction that generates nitrogen gas (Araujo et al., 2015).

This leads to the issue of the various nitrogen forms in the environment. Nitrogen can exist in several forms, with the most common being nitrate, nitrite, ammonia, and nitrogen gas. In this study, the primary focus is on nitrate due to its prevalence in farmland runoff, where nitrate uptake by plants and algae is the dominant method of assimilation (Ali, 2020). Plants can also indirectly use ammonia, either directly or by microbial conversion to nitrate, when sufficient oxygen is available (Greenway, 2007). The conversion of nitrite and ammonia to nitrate is governed by the equations below (EPA, 2002; Ford et al., 1980).

$$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 2\ H_2O + 4H^+ \tag{5}$$

$$2NO_2^- + O_2 \rightarrow 2NO_3^- \tag{6}$$

$$NH_3 + O_2 \rightarrow NO_2^- + 3H^+ + 2e^- \tag{7}$$

$$NO_2^- + H_20 \rightarrow NO_3^- + 2H^+ + 2e^- \tag{8}$$

Similar to denitrification, these reactions are microbially catalyzed. The microbes responsible for these reactions include *Nitrosomonas*, *Nitrobacter*, *Nitrosococcus*, and *Nitrospina* (EPA, 2002; Ford et al., 1980). FTWs have demonstrated effectiveness in removing ammonia from wastewater through this method, with removal efficiencies ranging from 96.8% to 21.6% (Shen et al., 2021). Like the physical processes described earlier, the use of microbes for nitrogen removal represents another natural process incorporated into standard water and wastewater treatment systems, which FTWs aim to replicate.

Microbes are highly efficient in nitrogen removal, and a wide array of other microbial species possess the capability to degrade, fix, or sequester various pollutants, while some even enhance the growth rates of specific macrophytes. This microbial diversity extends to pollutant treatment, spanning from plastics to heavy metals (Shahid et al., 2020). These microbes can either be intentionally introduced or naturally occur in waterways, colonizing the substrate and rhizomes of macrophytes by forming a biofilm (Shahid et al., 2020).

Phosphate removal can present more challenges compared to nitrogen, as microbes do not fix phosphorus as they do with nitrogen. Instead, wetlands remove phosphorus by acting as a reservoir and incorporating it into their growth and expansion processes. Phosphorus is retained by adhering to particles or surfaces within the substrate or through uptake by plants. While phosphorus utilized by plants eventually returns to the water through decomposition, it can become temporarily sequestered in less readily decomposable organic matter. Similar to nitrogen, phosphorus can exist in various forms in the environment. In this study, the primary concern pertains to orthophosphate (phosphate), an inorganic form of phosphorus that is bioavailable to plants and is commonly found in fertilizers (Dunne et al., 2005). In this inorganic form, plants have evolved specific mechanisms for absorbing inorganic phosphorus and transporting it across intracellular membranes (Schachtman et al., 1998). Due to its limited usability resulting from strong chemical bonds and poor mobility in water (Dunne et al., 2005), phosphorus often acts as the limiting nutrient for plant growth in natural, non-anthropogenically influenced systems (Schachtman et al., 1998). The availability and stability of inorganic phosphorus are influenced by a wide range of factors, encompassing water and soil chemistry parameters such as pH, dissolved oxygen concentration, mineral composition, temperature, retention time, and loading rate (Dunne et al., 2005).

In the context of FTWs, the removal of phosphate involves a surface complexation reaction with the wetland substrate, which, in this experiment, is peat. Phosphate ions, which carry a negative charge, interact with trace aluminum and iron oxides present in the peat (Bhattacharyya et al., 2018). The extent of phosphate adsorption is influenced by multiple factors, including the concentrations of the adsorbate in both aqueous and solid phases, flow dynamics, the quantity of adsorbent used, temperature, and pH (Kajjumba et al., 2019). These variables collectively determine the extent of adsorbate removal, driven by a combination of ionic and Van der Waals forces (Kajjumba et al., 2019).

Depending on the specific macrophytes employed, studies have demonstrated that FTWs can achieve impressive phosphate removal rates, reaching up to 90.7% (Shen et al., 2022). Phosphate removal in FTW systems primarily hinges on three major mechanisms: adsorption for phosphate, plant uptake for both nitrate and phosphate, and microbial activity for nitrate (Shen et al., 2022). In this study, these mechanisms will constitute the primary areas of focus.

2.2 Design Options for Recreating Natural Wetlands

The practice of creating artificial wetlands for water treatment has gained significant popularity in recent decades due to their remarkable effectiveness in removing excess nutrients, specifically nitrogen and phosphorus, from urban, agricultural runoff, and natural waterbodies, all without the extensive maintenance and infrastructure required by conventional water and wastewater treatment plants (Colares et al., 2020). The goal of these artificial wetlands is to replicate the natural processes described earlier. The design of these wetlands varies according to the intended outcomes, location, and the water being treated. The three most prevalent types of artificial wetlands are FTWs, surface flow, and subsurface flow. These methods have gained wide acceptance within the scientific community as viable treatment options for stormwater runoff and for reducing specific watershed nutrients, namely phosphorus and nitrogen (White, 2013).

However, each type comes with its own set of advantages and disadvantages. The design differences are visually represented in Figures 10-12.

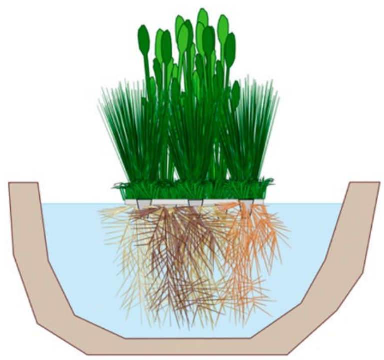

Figure 10. Floating Treatment Wetland (FTW). Adapted from "Wetland Technologies for Nursery and Greenhouse Compliance with Nutrient Regulations" by White, S. (2013), HortScience, 48, p. 1106-1107.

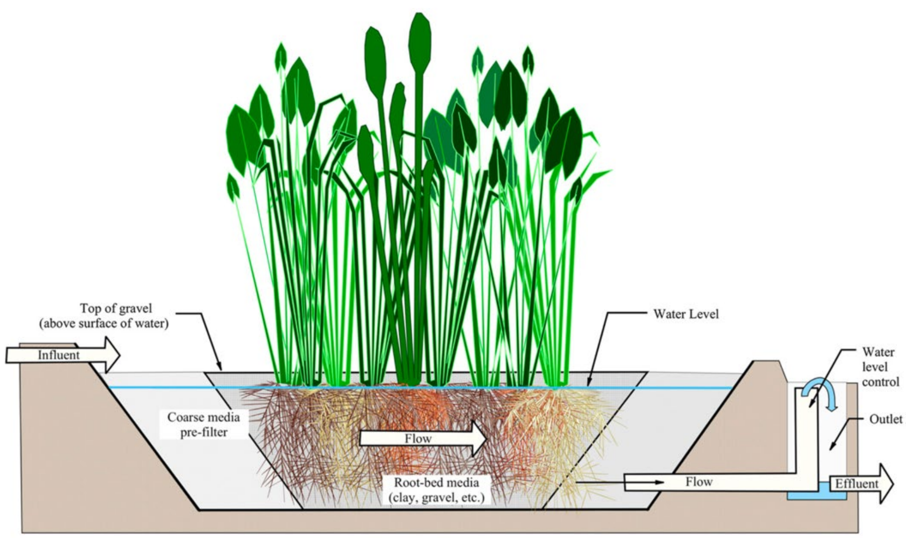

Figure 11. Subsurface Flow Wetland. Adapted from "Wetland Technologies for Nursery and Greenhouse Compliance with Nutrient Regulations" by White, S. (2013), HortScience, 48, p. 1106-1107.

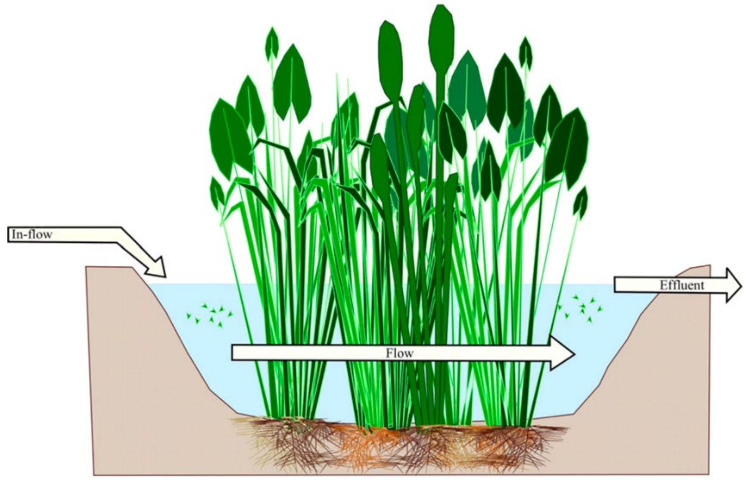

Figure 12. Surface Flow Wetland. Adapted from "Wetland Technologies for Nursery and Greenhouse Compliance with Nutrient Regulations" by White, S. (2013), HortScience, 48, p. 1106-1107.

The focus of this experiment was specifically on FTWs. As the name suggests, FTWs consist of a floating mat of vegetation with the roots of the macrophytes extending into the water column. These FTWs are designed to treat the surface water in which they float, with their roots and substrate serving as a matrix for microbes to colonize and treat the water, similar to the processes in a natural wetland (Shahid et al., 2020). Additionally, the plants would be able to extract nutrients directly from the water through contact with their roots.

For this project, FTWs were chosen for the design over other surface and subsurface flow designs. While all can be effective treatment methods, the primary factors influencing this decision were initial cost, materials, and land availability. Implementing an FTW, as opposed to a different surface flow or subsurface flow design, would require significantly less labor at startup and involve fewer modifications to the existing stream and channel. In the case of the English Coulee, one significant advantage of an FTW over other designs is its relative ease of installation on existing water bodies (White, 2013). Other benefits include the ability to adapt to temporary changes in water level by floating on the surface. FTWs can withstand higher inflow velocities than other types of constructed wetlands (Shen et al., 2021), and their suspension on the water allows for a greater root surface area, providing a larger surface area to interact with the target pollutants.

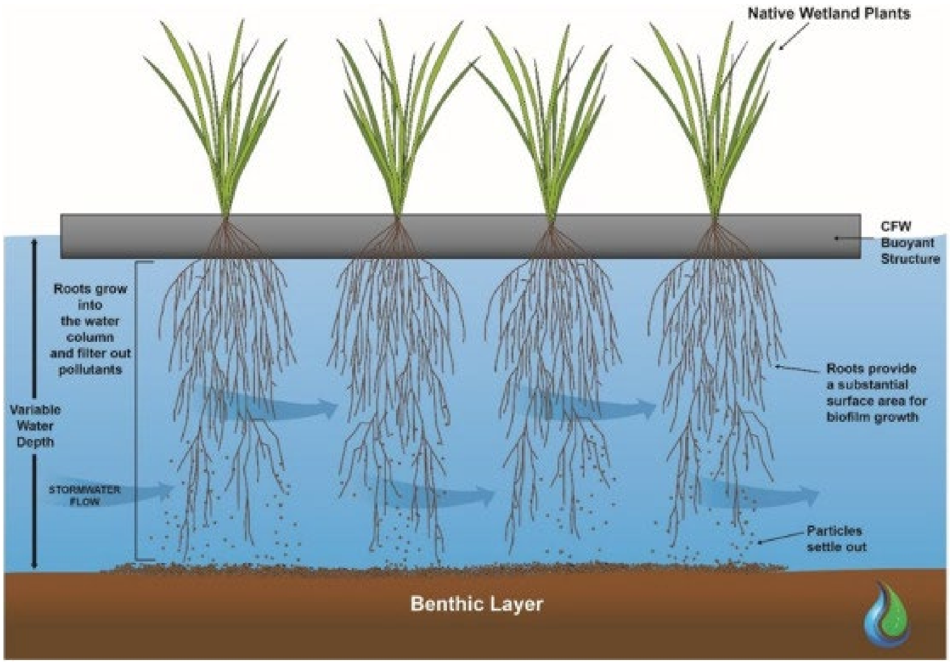

Figure 13. Floating Treatment Wetland (FTW) Diagram. Adapted from "Experimental designs of field-based constructed floating wetland studies: A review" by Lucke, T., et al. (2019), Science of the Total Environment, 660, p. 199-208.

Similar to natural wetlands, FTWs have proven effective in removing a wide variety of pollutants. Numerous studies have been conducted over the years to assess their effectiveness and design in various applications. These applications commonly include the treatment of urban stormwater runoff, municipal and industrial wastewater, landfill leachate, and mine drainage (Shahid et al., 2020).

While FTWs are highly effective in remediating natural systems, there are some limitations that must be considered. While compared to other designs, FTWs can better withstand temporary changes in water level, if the water level becomes too deep, the system's effectiveness diminishes because a smaller fraction of the water column interacts with the FTW (Shen et al., 2021). Conversely, the water level cannot fall too low. If the roots make contact with the bed, there is a risk of attachment and subsequent damage when the FTW moves and breaks the roots (Borne et al., 2015).

Another aspect to consider is the impact of high coverage. FTWs block sunlight and can affect benthic communities and habitats in areas with significant coverage (Borne et al., 2015). Light is not the only factor influencing the area below FTWs. Similar to natural wetlands, FTWs can lead to oxygen depletion in the area immediately beneath the floating mat (Borne et al., 2015).

Based on the existing site conditions, available information on design practices, and the fundamental mechanics of how wetlands improve water quality, FTWs appear to be a viable option for nutrient removal and the most appealing variant for an artificial wetland system in the case of the English Coulee. Nevertheless, several unique challenges specific to the English Coulee—such as its harsh cold-weather climate, unique hydrology, and seasonal flow patterns—could make the successful implementation of an FTW system very difficult.

Chapter 3: Methodology

The objectives of the study is threefold. First, to determine the effects of a small-scale FTW system on downstream nutrient concentrations in the English Coulee. Second, to identify the most suitable macrophytes for a potential large-scale design. Third, to determine, through small-scale experiments and modeling, the required size of an FTW system in the English Coulee to achieve a treatment objective of at least 50% phosphate removal. These objectives aim to test the hypothesis that the installation of an FTW system in the English Coulee will result in a noticeable and measurable decrease in downstream nitrate and phosphate concentrations, and to determine the necessary size of the wetland to achieve the treatment objective. The research can be divided into two main phases: pilot-study data collection and modeling.

3.1 Pilot Study Wetlands Design

During the summer of 2021, a pilot-scale FTW system was established in the English Coulee. The purpose of this small-scale design was to gain insights into its treatment efficiency and help determine the requirements for scaling up to a full-scale design. The location for the pilot-scale wetlands was carefully selected along the stretch of English Coulee that runs through the University of North Dakota campus to allow for easy monitoring. The specific area chosen was continuously flooded year-round by a stock dam located a little further upstream to accommodate the depth requirements of the FTW system. The location is depicted in Figures 14 and 15.

The islands in this experiment were constructed using 5.08 cm diameter PVC pipes. Each of the two islands consisted of three smaller segments that were interconnected. These individual segments measured 1.5 m by 1.5 m and featured cross supports along their edges. They were assembled using PVC cement and primer. To enhance buoyancy, the interior of the PVC pipes was filled with two-part expandable foam, and the exterior was wrapped with standard 7.112 cm diameter polyethylene foam pool noodles. Plastic deer fencing was used as a base for the planting area and secured to the frame with weather-resistant zip ties with a tensile strength of 222.41 Newtons. To contain the substrate and prevent environmental degradation, the frames and planting areas were covered with landscaping fabric, which allowed water to permeate through to the substrate and enabled the perennial root systems to penetrate into the water below. Figure 17 provides an illustration of the frame and fabric setup.

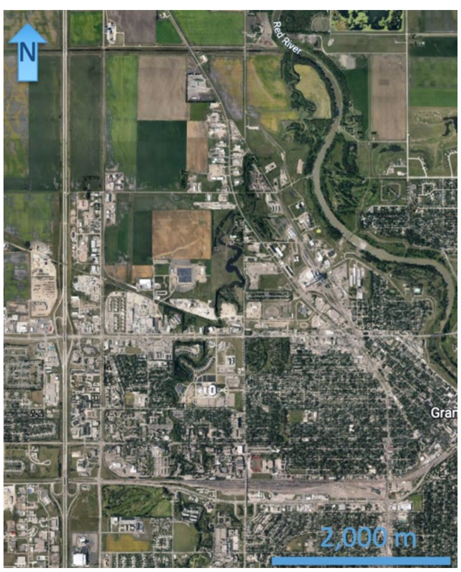

Figure 14. Map of Grand Forks. Aerial imagery taken from Google Maps, 2021.

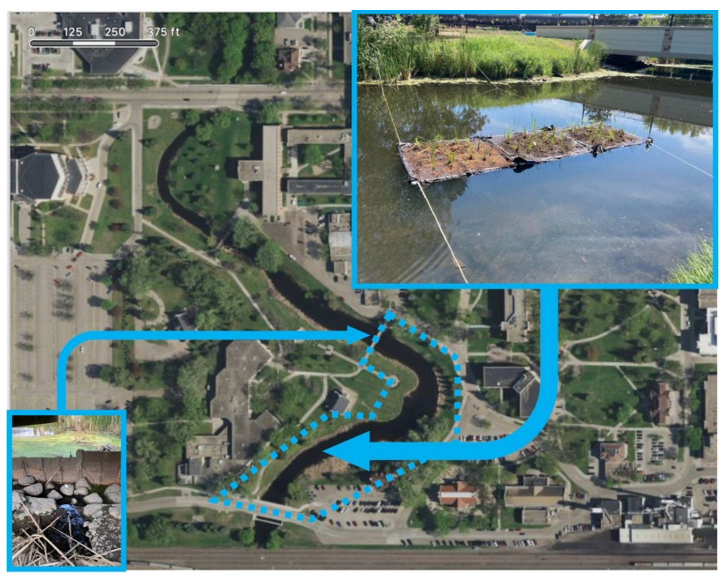

Figure 15. Site Map. Aerial imagery taken from Google Maps, 2021. Photographs by Michael Rosati. Taken July 9th, 2021 (upper right) and July 17th, 2021 (lower left).

Once the six segments were constructed, they were transported to the English Coulee and placed on the shore. Substrate and vegetation were added to the islands before they were gradually introduced to the water. Each island consisted of three segments, which were securely connected using 45.4 kg strength ropes. They were anchored to the shore from each corner with 7.6 cm by 38.1 cm earth anchors. Sufficient slack was provided in the lines to accommodate variations in water levels, allowing the islands to raise and lower as needed. The islands were aligned in a straight line, parallel to the flow of the stream, as illustrated in Figure 17.

Figure 16. Island Frame with Fabric Cover
Photograph by Michael Rosati. Taken July 5th, 2021.

Figure 17. Island Orientation in the English Coulee
Photograph by Michael Rosati. Taken July 9th, 2021.

In terms of FTW implementation, there is no universal standard design, and in each situation, it must be tailored to the specific environment and desired outcome. First, the physical environment must be considered. This includes factors such as root-depth ratio, flow velocity, hydraulic retention time, and channel dimensions (Borne et al., 2015). These factors determine the interaction times between the FTW and the water being treated. If the flow rate is too high, this could result in damage to the FTW, but if the flow rate is too low, mixing is limited, and less water interacts with the FTW, resulting in reduced nutrient removal efficiency (Shen et al., 2021). Root-depth ratio, placement, and channel dimensions are similarly important. If the root depth ratio is too small or the FTW doesn't stretch the width of the channel, water can flow around the FTW without interacting with it, thus bypassing treatment (Tirpak et al., 2022).

Next, the FTW itself should be considered. Parameters such as size, macrophytes used, and substrate also affect treatment and should also be considered. Increasing the size of the FTW provides more space for macrophytes and a larger matrix for the microbes to colonize, allowing for higher removal (Borne et al., 2015; Tirpak et al., 2022). However, this comes at a cost. In addition to the increased monetary cost, a larger FTW would require higher maintenance as well as have a larger impact on the existing environment and benthic community (Borne et al., 2015).

Six species of wetland plants that are commonly used in constructed wetlands were selected for this experiment. The plants were purchased as 5-cm plugs from Marshland Transplant Aquatic Nursery in Berlin, Wisconsin. The species used include *Sparganium eurycarpum*, *Juncus effusus*, *Iris virginica*, *Leersia oryzoides*, *Acorus americanus*, and *Schoenoplectus tabernaemontani*. All of the species were monocots, and five of the six are considered native to the Red River Valley region based on data from the USDA's PLANTS database (USDA, 2022), with the only outlier being *Iris virginica*, which was selected based on the advice of a wholesale plant dealer. Prior to placing the segments in the English Coulee, the tops of the vegetation were trimmed back. This was done to stimulate root growth and help the plants become established. A table of the selected plants and their quantities can be found below.

Table 1. Initial Macrophyte Count

Macrophyte	Family	Plant Functional Type	Number of Individuals
Sparganium eurycarpum	*Typhaceae*	Forb	20
Juncus effusus	*Juncaceae*	Sedge	40
Iris virginica	*Iridaceae*	Forb	20
Leersia oryzoides	*Poaceae*	Grass	20
Acorus americanus	*Acoraceae*	Grass	20
Schoenoplectus tabernaemontani	*Cyperaceae*	Sedge	30

Macrophyte selection is another crucial aspect of FTW design. When planning an FTW, it's essential to conduct a study of existing wetlands in the area. Plants thriving in these natural wetlands should be adapted to the local pests and climate (Borne et al., 2015). Ideally, selected plants should possess a substantial root system that extends from the floating mat, while remaining relatively low in height. Taller plants may catch the wind, potentially causing the FTW to shift if not anchored properly or, worse, causing structural damage (Borne et al., 2015).

Beyond these physical traits, plant selection can become quite technical, depending on the desired outcome. The chosen plants must also be capable of surviving the conditions to which they will be subjected, including variations in pH, salinity, and nutrient concentrations (Greenway, 2007). Moreover, macrophytes interact with the substrate and with each other. Some species can modify substrate conditions by creating reducing or oxidizing environments through the release of oxygen via their root systems (Brix, 1994, 1997). This oxygen release alters the soil's chemistry, affecting microbial colonization. Additionally, certain wetland plant species can produce antibiotics or compounds that either enhance or hinder the growth of neighboring plants (Brix, 1997).

Planting density is another critical factor affecting treatment performance. Increasing planting density enhances interactions with phosphates and suspended solids, leading to increased sorption (Tirpak et al., 2022). Understanding these interactions will be invaluable in designing an optimal wetland for specific site conditions and treatment objectives.

The substrate chosen for this experiment was Premier Compressed Sphagnum Peat Moss without added fertilizers or chemicals, sourced from Premier Tech located at 1 Avenue Premier,

Premier Tech Campus, Rivière-du-Loup, Quebec, and purchased from Menards in Grand Forks, North Dakota. The peat provides stability for the plants and serves as a matrix for beneficial microbes to grow.

A total of 0.78 m^3 of peat covered the 14 m^2 of planting area on the islands, resulting in a 5.6 cm-thick layer of peat. Prior to being placed, the peat needed to be saturated. Dry peat is initially hydrophobic and will float when water seeps through the bottom fabric. Pre-soaking it prevented the peat from rising and floating when initially installed in the English Coulee.

The choice of substrate in FTW design is a critical consideration due to its role in the sorption of phosphate, which is one of the primary mechanisms for reactive phosphate removal, alongside biological removal and sedimentation (Shen et al., 2022). As discussed in Chapter 2, various factors influence adsorption and impact removal efficiency. Understanding the species of adsorbents present and their quantities is essential for determining the adsorption rate. If these concentrations are naturally low in the substrate, additional adsorbents, such as iron oxides (Borggaard et al., 2005), can be introduced to enhance adsorption.

3.2 Sampling and Methods of Collection

The sampling equipment for this experiment consisted of two Hach AS950 samplers, manufactured by Hach and produced at 100 Dayton Ave, Ames, IA. One sampler was positioned 15.24 m upstream of the islands, and the other was placed 27.42 m downstream. Sampling was conducted at a depth of 7.62 m from the western shoreline. These distances were chosen to ensure the complete mixing of the FTW impact, treating it as a point source plume on the surface. The selection of these distances was based on calculations from the 'Handbook on

Mixing in Rivers' and involved a combination of physical measurements and simplified geometric assumptions, as described by Rutherford (1981).

$$X_m \cong 0.1 \frac{Ub^2}{D_z} \quad (9)$$

Where X_m is the downstream distance, *U* is velocity, *b* is width, and *Dz* is a constant dispersion coefficient.

Samples were collected once daily throughout the testing period. These samples were either promptly tested or refrigerated for a maximum of 48 hours, following the guidelines for the NitraVer 5 nitrate test and the PhosVer 3 phosphate test (HACH, 1996). The testing period for this project commenced on July 22nd, 2021 and concluded on August 27th, 2021. An illustration of one of the sampling stations is provided in Figure 18.

For each collection, a 500 ml sample was taken and tested for nitrates and phosphates using a Hach DR/2010 Spectrophotometer with NitraVer 5 and PhosVer 3 reagent pillows (Hach, 1996). The tests were conducted following the Hach DR/2010 handbook (Hach, 1996).

Plans to continue observing macrophyte growth and monitoring nutrient levels in the spring of 2022 were disrupted by intense flooding caused by melting snowpack and increased precipitation. The English Coulee water levels rose significantly, with a sharp rise in flowrate. Despite expectations that the islands would adapt to the elevated water level, the force of the high flow and debris collisions dislodged the earth anchors on the southern bank, resulting in the complete loss of all macrophytes and substrate. Consequently, data from the pilot-scale study is only available for the summer of 2021.

Figure 18. Downstream Sampling Station.
Photograph by Michael Rosati. Taken July 10th, 2021, on the UND campus.

Chapter 4: Modeling and Data Compilation

4. 1 HEC-RAS and HEC-HMS Models

Throughout this research, several models were employed to investigate the key mechanisms for nutrient removal, as outlined in Chapter 2. These models were developed using HEC-RAS and HEC-HMS software, which are products of the US Army Corps of Engineers, headquartered in Washington, D.C., designed for hydraulic modeling. Specifically, HEC-RAS and HEC-HMS models were constructed by the University of North Dakota to simulate the hydrology of the English Coulee and its basin. The data generated from these models are subsequently used in sizing calculations for denitrification and in estimating macrophyte contributions to nutrient removal. Additionally, channel data from HEC-RAS and HEC-HMS will be utilized in the adsorption model, which is constructed using Visual MINTEQ. Visual MINTEQ is a chemical equilibrium modeling software developed for the US Environmental Protection Agency and currently maintained by the Royal Institute of Technology in Stockholm, Sweden. The outcomes of these various models and calculations will be employed to determine the optimal size requirements for a theoretical FTW system.

The HEC-RAS model focuses on the main channel of the English Coulee and its surrounding banks. This model was initially constructed for a previous project at the University of North Dakota. It utilizes surveyed cross sections from the Army Corps of Engineers, an example of which is illustrated in Figure 21, and compiles them to create a 3D model of the English Coulee. HEC-RAS is particularly valuable for this type of modeling because it is dynamic and can account for changes in water level. As the water level fluctuates, the channel's volume adjusts accordingly. Given that the channel and overbank do not have convenient geometries, performing these calculations manually for each cross section would be highly time consuming. Figures 20 and 21 display examples of the cross sections used to build the model and the side profile of the composite channel in HEC-RAS.

Figure 20 illustrates the channel profile from the low-head dam to the point where the coulee discharges into the Red River. This profile presents channel distance, elevation, and water surface levels for various scenarios, incorporating structures like bridges, culverts, and stock dams. A more detailed description of the scenario depicted in Figures 19 and 20 will be provided in Chapter 5. Inputs for this HEC-RAS model draw from outputs generated by an HEC-HMS model.

HEC-HMS is another hydraulic modeling tool that aids in determining the hydraulic parameters necessary for sizing calculations. Unlike HEC-RAS, HEC-HMS takes a comprehensive watershed approach. This enables the incorporation of precipitation inputs, allowing for the calculation of runoff from the surrounding area into the system. A visual representation of the model is presented in Figure 21, which illustrates the drainage basin for the English Coulee. The figure displays various inputs into the basin, different sinks within the basin, area and topographical information for catchment areas, and the primary reaches of the English Coulee, depicted as thick blue lines, with smaller black lines connecting the various inputs and sinks.

Utilizing historical weather data in the HEC-HMS model provides the necessary inputs for the HEC-RAS model to simulate flow in the channel over a specific period. In the context of this project, the sampling period between July 22nd, 2021 and August 27th, 2021 was simulated using precipitation data sourced from the National Oceanic and Atmospheric Administration (NOAA).

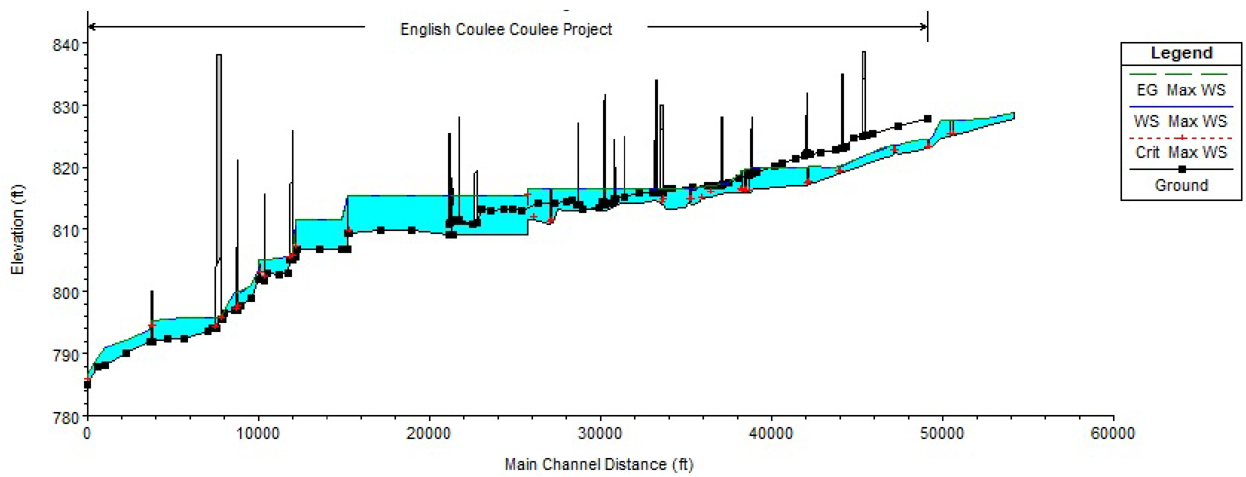

Figure 19. English Coulee HEC-RAS Channel Profile.

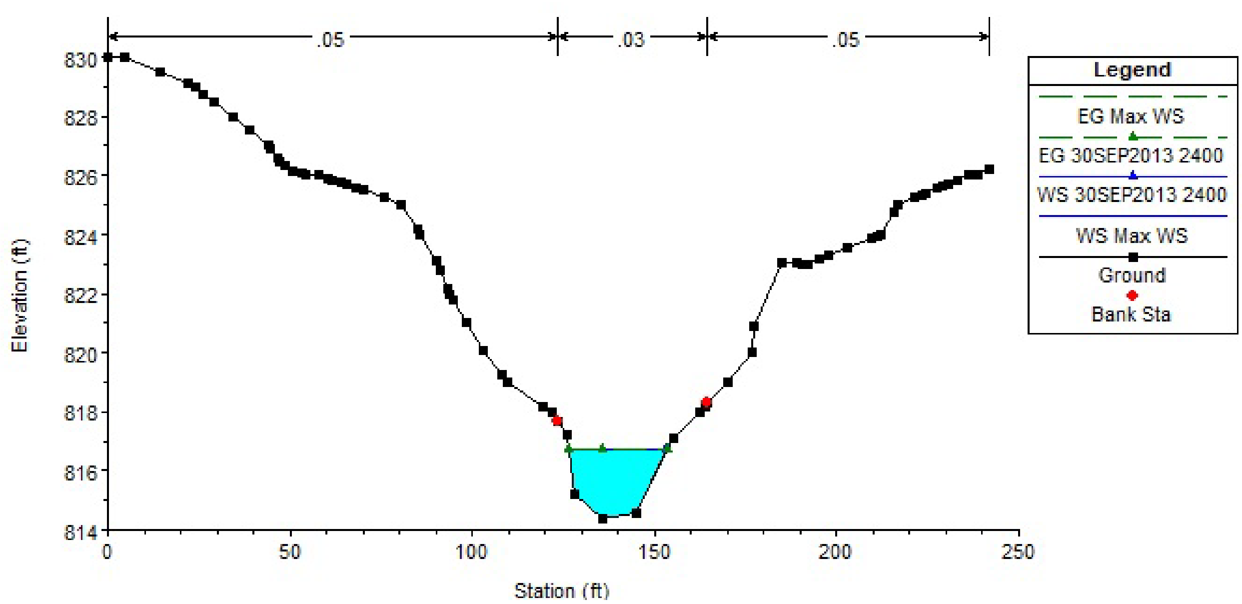

Figure 20. Sample HEC-RAS Channel Cross Section.

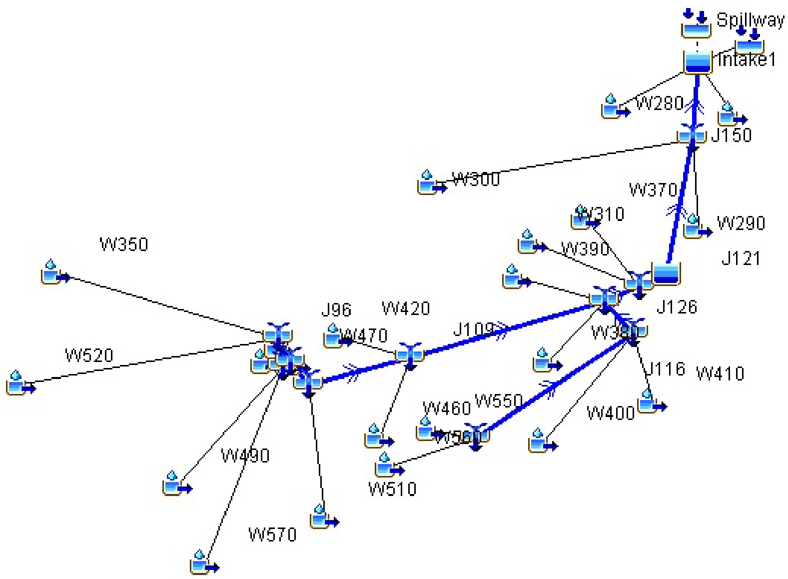

Figure 21. HEC-HMS Basin Model.

4.2 Nutrient Removal Model

When modeling nutrient removal in a hypothetical treatment wetland, it is crucial to begin by assessing the current conditions and defining the treatment objectives. In this case, the objective is to achieve a 50% removal rate for phosphate. This choice is grounded in the fact that, in freshwater systems, phosphorus is typically the most limiting nutrient, and aiming for a 50% phosphate removal aligns with established removal efficiencies observed in previous FTW studies (Shen et al., 2022; Van de Moortel et al., 2010).

The next step involved considering and simulating the primary removal mechanisms to determine the wetland's required size to meet this goal. Initially, loading rates for both Nitrogen and Phosphorus needed to be established. These values were sourced from a 2022 study conducted for the City of Grand Forks by the engineering firm AE2S, situated in Grand Forks, North Dakota (AE2S, 2022). This study determined loading rates for the urban portions of the English Coulee to be 5.79 kg/day for total nitrogen and 5.83 kg/day for total phosphorus. These values were derived from a combination of city zoning records, WinSLAMM data, city zoning information, and NRCS soil survey data (AE2S, 2022).

4.2.1 Nitrate Removal

The primary mechanism for nitrate removal in FTWs is through microbial activity and biological uptake. To simulate the removal of nitrate from water through biological uptake, existing nutrient uptake data was employed to determine a theoretical uptake rate per unit area, which will be used to calculate the wetland's size requirements.

On the other hand, assessing nitrate removal through microbial activity is more intricate because MINTEQ lacks a specific method for this process. Instead, the denitrification rate was calculated manually. This involved treating the segment of the English Coulee as an extended bioreactor, as outlined in Raboni et al. (2020).

$$SDNR_T = \frac{Q * \Delta N}{V * MLVSS} \tag{10}$$

Where $Q * \Delta N$ is the load of nitrogen removed in denitrification, V is volume of the denitrification reactor, and $MLVSS$ is the mixed liquor volatile suspended solids in denitrification and $SDNR_T$ is the specific denitrification rate.

4.2.2 Phosphate Removal

Modeling phosphate removal will follow a similar approach to that of nitrate removal. Existing data collected online will be utilized to determine an uptake rate per unit area for the macrophytes used in the system. Instead of focusing on microbial activity, which was the case for nitrate removal, this time the emphasis will be on adsorption, and removal will be determined through adsorption isotherms generated using MINTEQ.

Similar to nitrate modeling, phosphate solubility is influenced by various factors, including the minerals present in the soil and substrate, temperature, pH, and more. Data for these inputs was collected on October 3rd, 2021, using a YSI Multiparameter Sonde probe, produced by YSI, a Xylem brand at 1725 Brannum Lane, Yellow Springs, OH, 45387, and measurements along with sampling locations can be found in Figure 22 and Table 2 below. Flow information will be determined using the results obtained from the HEC-RAS and HEC-HMS models.

Table 2. Water Quality Parameters

Sample	pH	DO (mg/L)	Conductivity (µS/cm)
1	8.1	7.93	970
2	8.4	9.7	903
3	8.2	8.05	909
4	8.2	10.45	1046

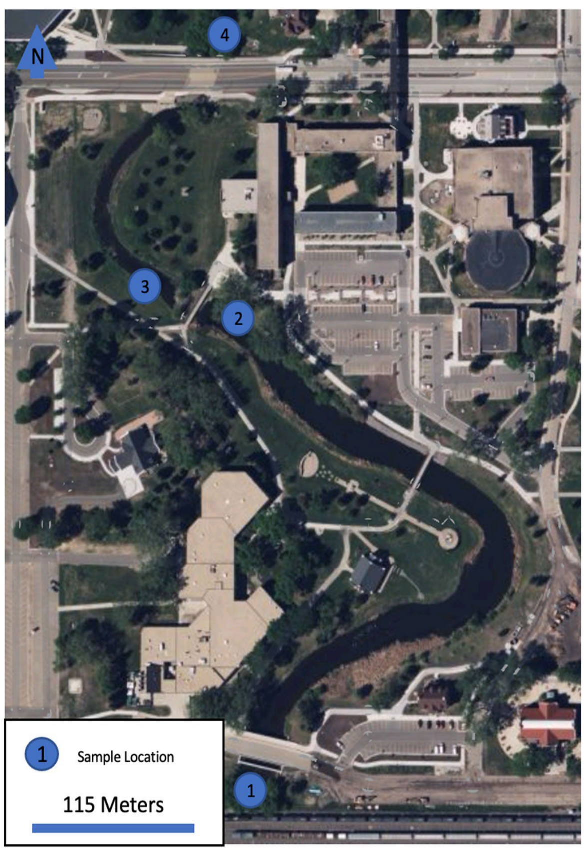

Figure 22. Water Quality Parameters Sampling locations. Aerial imagery provided by Google Earth, 2021.

4.2.3 Modeling Plant Uptake:

To simulate the influence of macrophytes on nutrient concentration, existing data on nutrient uptake specific to the macrophytes used in this experiment will be compiled. The most successful plants in this study, soft rush and rice cut grass, are commonly utilized in FTWs and other artificial wetland systems. As a result, there is a wealth of available data. However, it is important to note that due to the diverse environments and growing conditions in various experiments, results have shown significant variations, as indicated in Table 2 (Taylor et al., 2020; Spangler et al., 2019).

Table 3. Nutrient Uptake for Selected Macrophyte Species

Macrophyte Nutrient Uptake				
	Upper Approximation		Lower Approximation	
	N Uptake ($g/m^2/d$)	P Uptake ($g/m^2/d$)	N Uptake ($g/m^2/d$)	P Uptake ($g/m^2/d$)
Leersia oryzoides	0.42	0.144	0.0669	0.0246
Juncus effusus	0.505	0.057	0.0358	0.00953
Combined Approximate Removal for 50/50 Split Population(Kg/d)	73.95	10.58	6.10	1.78

In this model, a mix of soft rush and rice cut grass were analyzed. However, in reality, the English Coulee boasts a diverse range of native vegetation. Two species, cattails (likely *Typha angustifolia* or a hybrid) and duckweed (*Lemna minor*), were prominently observed in the study area. These species will exhibit their own influences on nitrate and phosphate concentrations, which we will discuss in greater detail in Section 5.1.3.

Chapter 5: Results and Discussion

5.1 Pilot Study Results

5.1.1 Nutrient Results

Data collection and testing took place from July 22nd, 2021, to August 27th, 2021, with a consistent daily sampling frequency of one sample from upstream and one from downstream. The collected data is depicted in Figures 23 and 24, and the raw data can be found in Appendix A.

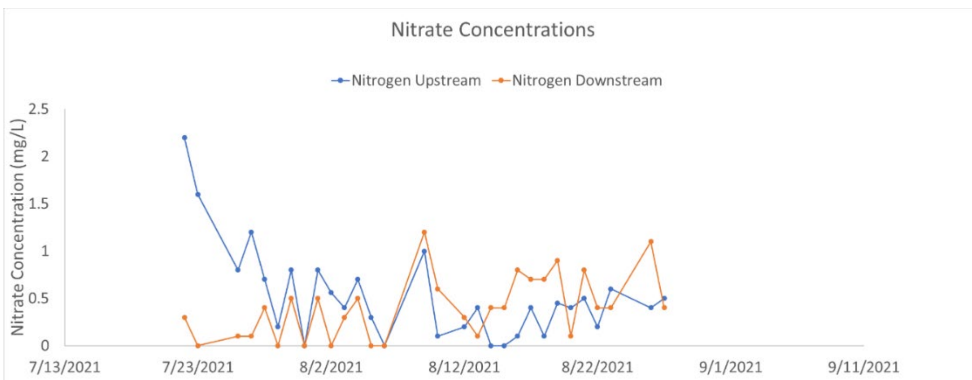

Figure 23. Nitrate Concentrations.

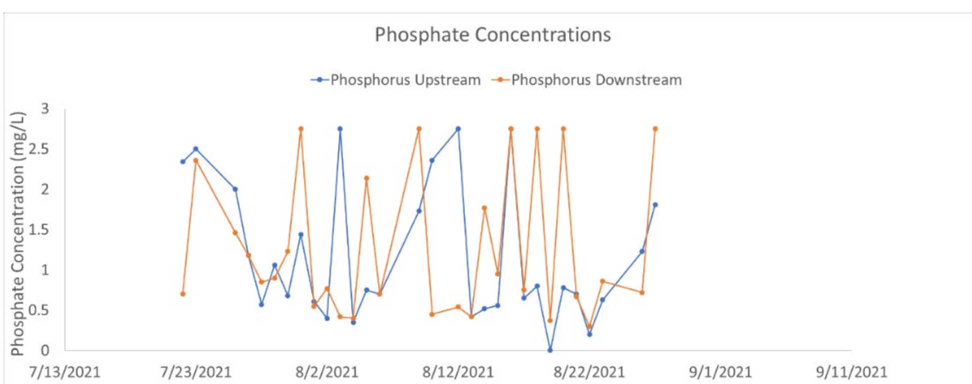

Figure 24. Phosphate Concentrations.

The distribution of nitrate and phosphate concentrations for both upstream and downstream samples can be observed in the following box plots.

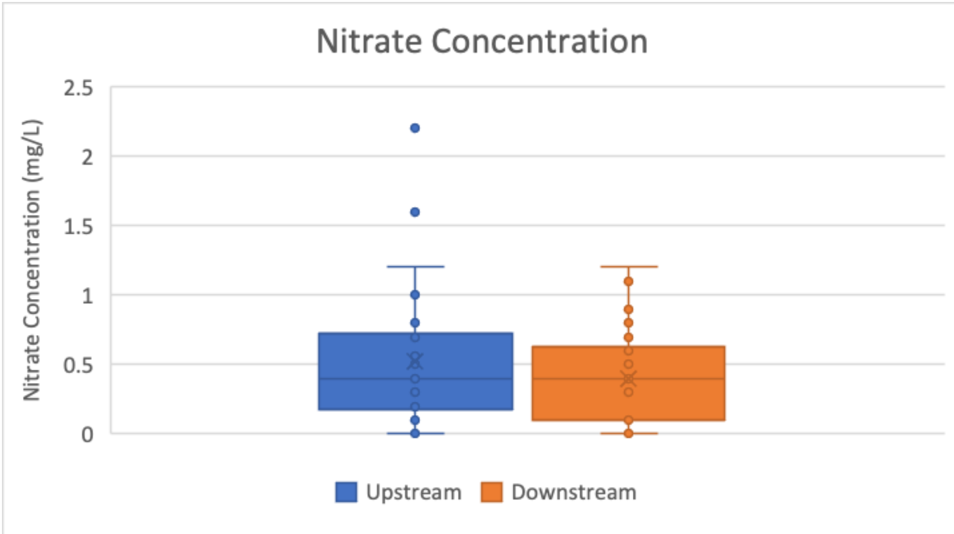

Figure 25. Nitrate Concentrations Box and Whiskers Plot.

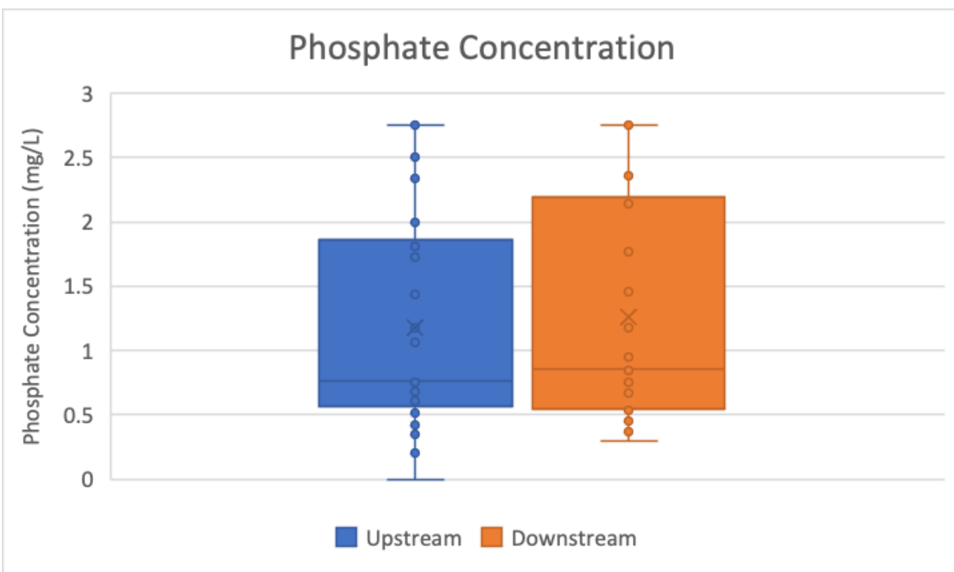

Figure 26. Phosphate Concentrations Box and Whiskers Plot.

The mean phosphate concentration for upstream samples was 1.174 ± 0.85 mg/L, and the mean nitrate concentration was 0.52 ± 0.49 mg/L. In contrast, the mean phosphate concentration for downstream samples was 1.265 ± 0.9 mg/L, and the mean nitrate concentration was 0.4 ± 0.34 mg/L. This dataset was analyzed using Microsoft Excel to assess whether there was a significant difference in nutrient concentrations between the upstream and downstream locations.

To evaluate the statistical difference between the upstream and downstream sample groups, a paired two-sample t-tests was conducted. The results of the Nitrate t-tests indicated no significant difference in nitrate concentrations between the upstream (M=0.52 mg/L, SD=0.49 mg/L) and downstream (M=0.4, SD=0.34 mg/L) conditions; $t(29) = 1.7$, $p = 0.145$. Likewise, the Phosphate t-tests revealed no significant difference in phosphate concentrations between the upstream (M=1.174, SD=0.85 mg/L) and downstream (M=1.265, SD=0.9 mg/L) conditions; $t(29) = 1.7$, $p = 0.321$. These findings indicate that there is no statistical difference between the mean concentrations of nitrate or phosphate in the upstream and downstream sample groups. In both cases, the p-values exceeded the 0.05 (5%) alpha threshold, suggesting that the null hypothesis is supported, and there is no statistical difference between the means of the upstream and downstream groups. This outcome is likely influenced by multiple factors, which will be discussed in more detail in Chapter 5.

Table 4. Tabular Results of t-test

Nutrient	Upstream (Mean ± SD)	Downstream (Mean ± SD)	t-value	Degrees of Freedom (df)	p-value
Phosphate	1.174 ± 0.85 mg/L	1.27 ± 0.9 mg/L	1.7	29	0.321
Nitrate	0.52 ± 0.49 mg/L	0.4 ± 0.34 mg/L	1.7	29	0.145

5.1.2 Plant Survival Rates

Out of the 150 plants initially planted at the beginning of the experiment, only two species managed to survive until winter dormancy: soft rush and rice cutgrass. The low survival rate of the other species can be attributed to a combination of environmental factors. One noteworthy factor that was frequently observed was herbivory. Shortly after installing the islands in the English Coulee, muskrats were observed grazing on and carrying away plants from the islands. While this was not ideal for the nutrient removal aspect, it provided valuable data regarding plant selection that might have been overlooked in a controlled laboratory setting. One of the attractive features of FTWs is their low maintenance, and having species that are resistant to grazing is critical for long-term success. Specific survival numbers for the various species can be found in Table 4.

Table 5. Observed Macrophyte Survival Rate

Macrophyte	Initial Count	Final Population	Survival Percentage
Sparganium eurycarpum	20	0	0%
Juncus effusus	40	35	88%
Iris virginica	20	0	0%
Leersia oryzoides	20	10	50%
Acorus americanus	20	4	20%
Schoenoplectus tabernaemontani	30	0	0%

Based on the results of the survival rate, the selected species for the modeling portion will be *Juncus effusus* and *Leersia oryzoides*. The decision to employ two species, rather than creating a monoculture, has several advantages. First, it can enhance the aesthetics of the English Coulee, which runs through the heart of the University of North Dakota campus. Additionally, a polyculture, compared to a monoculture, can provide increased resilience to environmental stressors, pests, and diseases (Kadlec et al., 2009). Another potential benefit is improved ecosystem quality, offering a more diverse range of food sources and habitats for native fauna (EPA, 2000).

From a nutrient removal perspective, whether a monoculture or polyculture is more effective in treatment wetlands has produced mixed results. It is hypothesized that combining plant species with complementary functional traits can enhance treatment efficiency (Mandal et al., 2018). Eviner and Chapin (2003) propose that an increase in plant diversity in treatment wetlands can enhance tolerance to changing conditions and provide improved biogeochemical stability (Eviner et al., 2003; Borgström et al., 2022). Moreover, differences in seasonal growth patterns could lead to improved efficiency year-round, with different species being more productive at different times of the year (Coleman et al., 2001; Fraser et al., 2004; Picard et al., 2005). However, in the context of replicated experimental measurements, results have been contradictory. Other studies suggest that using a combination of plant species does not necessarily increase treatment efficiency in treatment wetlands (Brisson et al., 2015; Luo et al., 2023). Conversely, studies by Zhang et al. (2010) and Zhu et al. (2010) do suggest that an increase in the number of plant species can lead to improved nitrogen removal.

Considering the advantages previously outlined, a polyculture was selected for the modeling calculations in this project. While specific data on phosphate and nitrate removal for these two species in this particular context is not readily available, insights can be derived from existing information about the plant species themselves. Both species are grasses and occupy similar niches in their environment. However, certain characteristics may influence their growth and, consequently, their capacity to remove nitrates and phosphates from the water.

One such characteristic is a potent oxidative effect caused by the release of oxygen by the roots of *Juncus effusus*, up to 0.5 mg/hour per plant (Wießner et al., 2002; Blossfeld et al., 2011). Additionally, the root system for *Leersia oryzoides* metabolically acclimates to lower oxygen levels in the soil, leading to the decay of the original roots during anaerobiosis. While *Leersia* *oryzoides* can survive by compensating with adventitious roots, nutrient delivery is diminished (Koontz et al., 2013). If planted alongside *Juncus effusus*, it may help maintain a higher level of nutrient delivery, thereby increasing the removal of both nitrates and phosphates from the water. However, further studies would be necessary to precisely determine the nature of this relationship.

5.1.3 Discussion of Pilot Study Results

Based on the results of the analysis of the pilot-scale study, it was observed that nutrient concentrations both upstream and downstream of the pilot-scale FTWs were similar. A review of FTWs in the literature specifically for river systems, provides insights into their usage and optimization for nutrient and organic extraction from river systems.

The average design specifications for FTWs, gathered from various case studies, include a coverage of 53.9% ± 28.5%, a water depth of 1.15 ± 0.89 m, and a hydraulic retention time (HRT) of 8.01 ± 6.77 days. The typical efficiency of FTWs in purifying natural water is outlined by average removal rates for various pollutants: TN (46.5% ± 23.7%), $NH4^{+}$-N (62.5% ± 26.4%), $NO3^{-}$-N (52.2 ± 29.3%), TP (50.6% ± 25.0%), and COD (42.4% ± 23.8%) (Shen et al., 2021).

Design recommendations for FTWs in river purification suggest a coverage spectrum of 25–60%, surpassing the 5–45% coverage suggested for lakes, ponds, or reservoirs (Shen et al., 2021). Elevated FTW coverage may impede hydrological characteristics, decelerating flow and impacting flood discharge and storage functions. Variances in root structure and absorption among macrophytes contribute to differences in pollutant removal efficiency, with studies revealing different floating mat coverage requirements for distinct plant species (Shen et al., 2021).

The efficiency of pollutant removal is notably influenced by water-plant interactions, incorporating factors such as the amount of water and duration. Plant root characteristics play a crucial role, with species like *Typha angustifolia* showcasing superior treatment performance owing to their thin and loose roots, facilitating effective water-plant interactions (Shen et al., 2021).

While FTWs demonstrate effectiveness, especially in river systems, studies suggest they may be less efficient in deep and static water ecosystems due to inadequate interactions between plant roots and the water column. The pivotal role of hydraulic retention time (HRT) in influencing pollutant removal efficiency is underscored (Olguin et al., 2017; Xiao et al., 2016).

Among the studies reviewed, Zhou and Wang (2010) demonstrated optimal removal efficiencies for a river system. An FTW with a coverage of 80-90% and a depth of 0.38 m, planted with *O. javanica*, achieved removal efficiencies of 90.78% for total nitrogen and 76.47% for total phosphorus. While other studies reported comparable or better removal efficiencies, comprehensive parameter information was not consistently provided. This reinforces the importance of design parameters, plant species selection, and water-plant interactions in optimizing FTW performance for water purification in rivers.

Several factors likely contributed to the lower efficiencies observed in this pilot-scale study. First, the plants used were all juvenile 5-cm plugs. Their young age and small size meant their root systems were also limited in size. Given the importance of root size in nutrient uptake and efficiency, the uptake in these juvenile plants is expected to be lower compared to fully grown, more established plants (Bar-Tal et al., 1997). As plants grow, their demand for nutrients increases; however, during the experiment's measurement stage, the uptake from juvenile plants was likely modest. Additionally, due to the small size and recent planting of the plants, their root systems did not penetrate deep into the water column, resulting in a smaller area for microbial colonization and reduced contact area, further contributing to lower nutrient removal compared to established FTWs.

Second, the pilot-scale wetlands were relatively small compared to the water body they were intended to treat. With a size of 13.9 m^2, these FTWs were the largest that could be created with the available resources, whereas ideal coverage should be approximately 50% (Borne et al., 2015). In relation to the total surface area of approximately 25.9 ha of the English Coulee, from the dry dam to the Red River, based on data from the national wetlands inventory (USFWS, 2023), the pilot-scale FTWs were considerably undersized, limiting their overall impact. Using the 50% coverage recommended by Borne et al., 2015, an FTW of 12.99 ha would be needed and based of the 50% phosphate and nitrate removal treatment objectives and modeling results, a 4.69 ha FTW would be required to achieve these results. Constructing an FTW system large enough to achieve the desired removal efficiency would be extremely impractical, considering the area and shape of the English Coulee.

In addition to sizing and macrophyte-related issues, challenges were encountered in isolating the effects of the islands from the surrounding environment. Achieving complete isolation from the surroundings was made difficult by the presence of surrounding vegetation. Patches of wetland plants that performed similar nutrient removal processes to the FTWs can be found along the English Coulee. Using remote sensing data acquired from the U.S. Geological Survey's Earth Explorer website (https://earthexplorer.usgs.gov/), riparian vegetation patches in the study area between the upstream and downstream sampling stations were identified and extracted using ArcGIS Pro. This analysis revealed approximately 65 m^2 of natural vegetation on the banks in the study area located between the upstream and downstream sampling stations.

Based on removal efficiencies reported in previous studies (Shen et al., 2021), it was estimated that this riparian vegetation accounted for approximately 0.74 g/hour of nitrate and 0.037 g/hour of phosphate removal. In contrast, the 13.9 m^2 FTW was expected to contribute roughly 0.16 g total-N/hour and 0.008 g total-P/hour removal. Because of the significant size difference, any potential effect of the FTW was likely overshadowed by the presence of existing riparian vegetation, such as cattails or duckweed, which were both present in the sampling area throughout the sampling period. Notably, there was a substantial stand of cattails on the southeast bank of the English Coulee at the pilot-scale study site, covering an estimated area of approximately 550 m^2, as determined from GIS data in ArcPro. This riparian vegetation, with the potential to remove phosphorus ranging from 3 g P/m^2 to 6 g P/m^2 annually (Grosshans et al., 2011), could lead to an average daily removal between 8.2 mg P/day and 16.4 mg P/day, which might have influenced the collected data.

Additionally, duckweed, is a native plant frequently found in the study area and throughout the English Coulee. Duckweed thrives in small aquatic environments with slow-moving water. These plants propagate rapidly, forming extensive floating mats. Duckweed can play a vital role in breaking down pollutants and absorbing excess phosphates and nitrates from the water. A single square meter of duckweed can remove between 30 to 200 mg of total phosphorus per day and 40 to 310 mg of Total Kjeldahl Nitrogen (TKN) per day, depending on environmental conditions (Iqbal et al., 2019). However, when duckweed dies and decomposes, the nutrients are released back into the water source. This presents an opportunity for future research—exploring methods to harvest and efficiently use duckweed for complete removal of excess phosphates and nitrates from the ecosystem. Given duckweed's potential as a food source, fertilizer, and even as a biofuel, further investigation into its harvesting and utilization is a promising avenue for research (Spiegel et al., 2013; Cheng et al., 2009).

In addition to the presence of native vegetation, certain uncertainties in the pilot-scale study results could be attributed to existing infrastructure. Several meters upstream of the upstream sampling station, there was a PVC discharge pipe that released water with a chemical composition that could not be assessed with the available equipment for this research.

Another source of uncertainty likely stemmed from variations in flow within the English Coulee. The placement of the samplers was determined based on a measured low flow rate of 0.0142 cms, with the assumption that mixing would occur within that velocity, and the effects of the islands would be entirely mixed by the time they reached the downstream sampler.

Unfortunately, gauge data in the study area were limited, and the flow was inconsistent for a significant portion of the experiment. Flow rates ranged from being stagnant to significantly higher than the measured 0.0142 cms, often due to rain events. While USGS gage data is not available for sampling period, historical data from 2008 to 2009 at Sampling station 05082590 near Thompson Road, ND, indicates a range of flowrates from 0 cms to 3.397 cms (USGS, 2023). These fluctuations in flow rates and mixing patterns, combined with the setup of the sampling stations, likely contributed to data that may not accurately reflect the effects of the FTWs, potentially explaining the observed higher downstream nutrient concentrations.

5.2 Results of Model

5.2.1 HEC-RAS and HEC-HMS Model Results

Utilizing a simulated flow rate of 0.6455 cms and precipitation data recorded from July 22, 2011, to August 27, 2011, our models revealed a storage volume of 217,932 m^3 and an associated travel time of 93.24 hours. These factors will play a crucial role in determining loading information and nutrient removal.

5.2.2 Adsorption Model Results

The MINTEQ model determined an adsorption rate of 49.6%. This method relies on a ferrihydrite CDM adsorption database integrated into MINTEQ. The selection of this specific method and database was based on its frequent occurrence in impaired waters. Ideally, trace metal concentrations would be obtained from laboratory experiments. However, in this case, the inputs were determined using historical data and data from similar systems.

5.2.3 Nitrate Modeling Results

Based on methods outlined in Chapter Four, field measurements from the English Coulee, HEC-HMS and HEC-RAS outputs, and historical data approximations, a denitrification rate of 0.0157 kg NO_3^-/kg MLVSS/ day was determined, with MLVSS indicating microbial presence. MLVSS, which stands for Mixed Liquor Volatile Suspended Solids, plays a significant role in indicating the presence and health of the microbial community responsible for denitrification. MLVSS represents the concentration of microorganisms, particularly bacteria, within the system. These microorganisms are essential for the denitrification process, as they facilitate the conversion of nitrates (NO_3^-) in water treatment systems to harmless nitrogen gas (N_2) under anoxic conditions (Tchobanoglous, 2014).

5.2.4 Macrophyte Uptake Results

Given the substantial variability in reported values for the selected macrophytes, a cautious approach was taken, and the lower values from Table 3 were utilized in the calculations to determine the required FTW size specifications. Assuming equal coverage of the two species, a 4.69 ha wetland would yield a nitrogen removal rate of 287.3 kg/d and a phosphorus removal rate of 83.86 kg/d, based on the available uptake data.

However, the use of stand-in values introduces a considerable degree of uncertainty, even though the selected species match those used in this experiment. Therefore, it is strongly recommended to gather data under the specific conditions these plants will experience in the English Coulee to establish a more accurate removal rate.

5.2.5 Discussion of Model Results

The results derived from these various models and calculations were then used to establish a minimum size requirement. It is important to note that this is an approximation, based on models featuring simplifying assumptions, stand-in values from other research, assumptions regarding water and soil chemistry, and data collected during the late summer and early fall of 2021. Several key assumptions were made when sizing the FTW:

1. Phosphate, as the most limiting nutrient and a primary contributor to problematic algal blooms, was the focus of design considerations.
2. Nitrate levels in the English Coulee, based on data from the pilot-scale study, were found to be close to compliance with EPA standards.
3. A significant portion of nitrate removal is attributed to microbial fixation and macrophyte uptake.

4. A substantial, if not exclusive, portion of phosphate removal is attributed to adsorption and macrophyte uptake.

Due to the variability in available macrophyte data regarding nutrient removal, the adsorption model served as the primary tool for sizing the wetland to achieve a 50% removal rate for phosphate. This led to a minimum required area of 4.69 ha, necessitating 28,703 m^3 of peat and covering 18.5% of the water's surface from the Red River to the dry dam.

5.3 Discussion on Design and Optimization

The flow and hydrology in the pilot-scale study presented challenges, ultimately leading to the complete loss of the FTW in a flood event during the spring of 2022. This aspect is notably problematic in the context of the English Coulee. Seasonal flooding, resulting from snowmelt and aggressive flow control measures like dams and diversion channels further upstream, leads to extreme variations at different times of the year. These variations include low flow and water levels in the summer and fall, and high water levels and increased flow during the spring. Designing for these conditions can be challenging. However, as previously emphasized in this report, these challenges should ideally be overcome by FTWs, which are designed to thrive under such conditions.

The islands might have survived the flood event if more durable and buoyant materials were employed in conjunction with protective measures against debris strikes, such as the use of collection nets in front of the wetlands to mitigate the impact of larger debris. These are all critical factors to consider, and given more time and a larger budget, implementing such measures could significantly enhance the long-term success of an FTW system. Nevertheless, further research and analysis would be necessary to provide definitive conclusions in this context.

In addition to flow and hydrology, careful plant selection is a crucial factor to consider for optimizing success, particularly in the unique environment of the English Coulee. Plant selection is a critical aspect in every FTW design, as it directly influences nutrient uptake and provides a matrix for microbial colonization, which enhances water treatment as it comes into contact with plant roots and associated microbes. Selecting native plants adapted to the local environment can significantly increase the chances of success. These plants have already evolved to thrive in the local conditions, possess specific life histories, and carry genetic traits tailored to the region. This 'local versus non-local' concept is well-supported, with numerous benefits associated with choosing native species (Jones, 2013).

Another significant consideration is the restoration of natural wetlands and vegetation in the riparian zone. As previously mentioned in Chapter 1, a substantial portion of the English Coulees watershed has been converted to cropland, resulting in only approximately 14.1% of forested and wetland areas within 30.5 m of each bank. Restoring some or all of this buffer zone could lead to a substantial reduction in nutrient loads, particularly for phosphates and nitrates (Lee & Schultz, 2003). This restoration effort could potentially reduce the size of the artificial wetland required. For instance, restoring just 50% of the 30.5 m buffer on each side of the English Coulee along the lower 14.5 km reach that passes through the campus would introduce an additional 0.4 km^2 of wetland and vegetated areas, contributing to enhanced nutrient removal and buffer capacity.

Further enhancements in efficiency for this FTW system could involve dosing the substrate with an adsorbent like iron oxide. This addition would provide additional binding sites for phosphates, ultimately improving phosphate removal (Tchobanoglous, 2014).

5.4 Further Research

The small-scale pilot experiment did not achieve the desired reduction in nutrients in the English Coulee. However, this outcome can be attributed to limitations stemming from time and resource constraints. To definitively determine the feasibility and success of an FTW system in the English Coulee, further research is essential. Scaling up the design as per the models would demand more substantial resources and funding. Yet, with the utilization of more durable equipment and well-established plants on a larger scale, the wetlands could be substantial enough to produce quantifiable differences.

Another approach to lowering nutrient concentrations in the English Coulee involves restoring the flushing effect that is currently obstructed by various hydraulic structures. There are two potential solutions based on the research and modeling from this experiment: removing stock dams and lowering the intake structure. The HEC-RAS model indicates that removing stock dams could facilitate drainage of impounded areas without increasing flood risk in overbank areas. However, the feasibility of lowering the earthen dam inlet structure would require further research and extensive modeling to accurately assess flood hazards and redesign the intake. While this project's scope does not encompass such detailed research, it represents a promising area for future investigation.

Additionally, this paper did not investigate the ecosystem effects of hydraulic modifications, such as the removal of stock dams. These changes could potentially restore the English Coulee to a more natural state but may also impact the existing ecosystem that has adapted to the current conditions. Further research into the ecosystem effects of removing or altering these structures represents a valuable avenue for future investigation.

5.5 Alternative Methods

Phosphorus concentrations in the English Coulee are much higher than EPA recommendations and reducing them to the recommended levels would require a very large FTW. Given the English Coulee's shape and volatile flood conditions in the spring, maintaining a wetland of that size would be cumbersome. Because of this an FTW alone is likely not the answer to the water quality issues facing the English Coulee. Instead, it could possibly be supplemented with other practices such as restoring riparian wetlands and vegetation in the buffer area. Another option would be to remove the stock dams that are retaining water in these reservoir areas. As stated in the previous section, removing the stock dams would allow the English Coulee to return to a more natural state. Based on the HEC-HMMS model results, this would have no negative effects from a flood control perspective and improve drainage. This would allow the flushing effect to resume, addressing issues related to algal blooms in the English Coulee.

However, this approach would likely result in a significant and uncertain ecological shift in the current English Coulee ecosystem. Moreover, it may not effectively address the core issue, as the retained nutrients could still flow into the Red River, further impacting Lake Winnipeg, which is already grappling with nutrient loading and algal bloom problems (Binding et al., 2018). An alternative option would be to address the source of the phosphates and nitrates and reduce the loading rates into the English Coulee. Given the non-point source origins of nitrate and phosphate pollution in the English Coulee watershed, alternative methods, such as Better Management Practices (BMPs), could be employed. BMPs have demonstrated their effectiveness in reducing nitrate and phosphate loading without the need for extensive construction projects (Bosch et al., 2014; Cooper et al., 2004).

There is a wide range of Best Management Practices (BMPs) that can be employed to help reduce nitrate and phosphate loading into the English Coulee. While we have previously discussed riparian buffers, other BMP strategies are focused on agricultural producers. Effective strategies include using cover crops, employing precision agriculture techniques, and reducing fertilizer usage to decrease phosphate and nitrate loading (Khanal et al., 2015).

On the city level, enhanced stormwater management and urban planning are two approaches to reduce loading rates not only of phosphates and nitrates but also of other pollutants, such as *E. coli*. In the case of the English Coulee, elevated levels of *E. coli* have been observed in both runoff samples and at sampling stations throughout the area (NDDEQ, 2016). As previously mentioned, the city of Grand Forks is currently engaged in an ongoing restoration effort for the English Coulee, which includes the implementation of BMPs. The analysis phase of this project has recently been completed, and the city is now progressing to the next phase which includes creating a more detailed hydraulic model to assess potential hydraulic changes as well as a more detailed sediment analysis.

Chapter 6: Conclusions

In this study, a comprehensive investigation into the feasibility of implementing an FTW system in the English Coulee, Grand Forks, ND, to address elevated nutrient pollution, specifically nitrates and phosphates was performed. The objectives encompassed evaluating the effects of a small-scale FTW system, identifying suitable macrophyte species, and determining the necessary FTW size for at least 50% phosphate removal. From these analysis, several conclusions can be drawn.

The pilot-scale study results showed no statistical difference in nitrate or phosphate concentrations between upstream and downstream locations in the English Coulee. Several factors, such as the young age of the plants, the pilot-scale FTW size, and the presence of riparian vegetation, likely contributed to these findings. Furthermore, variations in flow within the English Coulee and the hydrology of the region presented additional challenges for the effectiveness of the pilot-scale FTW.

The modeling results provided valuable data indicating a 4.69 ha FTW would be required to achieve the desired nutrient removal objectives. However, the study revealed that even a 4.69 ha FTW would not be practical given size and space requirements, as well as the unique cold region hydrology and flood patterns of the English Coulee.

To optimize the potential success of an FTW in the English Coulee, careful consideration should be given to plant selection, the restoration of natural wetlands and vegetation in the riparian zone, and the use of adsorbents like iron oxide to enhance phosphate removal.

Exploring alternative or a combination of methods, such as restoring riparian wetlands, removing stock dams, and implementing BMPs to reduce non-point source nutrient runoff, offers promising solutions for addressing water quality issues in the English Coulee. During Phase 1 of the English Coulee Renewal project led by the city of Grand Forks in collaboration with AE2S, proposed approaches included creating a water quality treatment basin to mitigate upstream phosphorus, sediment, and bacteria loading before reaching the city. Modifying the diversion channel could reduce or eliminate the connection between the agricultural watershed and the city. Addressing phosphorus and sediment loading from the city may involve small-scale maintenance and dredging, narrowing stagnant areas, or removing stock dams to enhance flow in the English Coulee. Augmenting flow with an additional water source during dry periods is another potential solution, but careful consideration of the water source is imperative (AE2S, 2022). Future research efforts are necessary to develop a holistic and effective solution to the nutrient pollution problem in the English Coulee and its downstream impacts on the environment.

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Appendix A: Supplementary Data

Table A1. Nutrient Concentrations Raw Data

	Upstream		Downstream	
Sample Date	Sample (mg/L N NO3-)	Sample (mg/L P PO4 3-)	Sample (mg/L N NO3-)	Sample (mg/L P PO4 3-)
7/22/21	2.2	2.34	0.3	0.7
7/23/21	1.6	2.5	0	2.36
7/26/21	0.8	2	0.1	1.46
7/27/21	1.2	1.18	0.1	1.18
7/28/21	0.7	0.57	0.4	0.85
7/29/21	0.2	1.06	0	0.9
7/30/21	0.8	0.68	0.5	1.23
7/31/21	0	1.44	0	2.75
8/1/21	0.8	0.61	0.5	0.55
8/2/21	0.56	0.4	0	0.77
8/3/21	0.4	2.75	0.3	0.42
8/4/21	0.7	0.35	0.5	0.4
8/5/21	0.3	0.75	0	2.14
8/6/21	0	0.7	0	0.7
8/9/21	1	1.73	1.2	2.75
8/10/21	0.1	2.36	0.6	0.45
8/12/21	0.2	2.75	0.3	0.54
8/13/21	0.4	0.42	0.1	0.42
8/14/21	0	0.52	0.4	1.77
8/15/21	0	0.56	0.4	0.95
8/16/21	0.1	2.75	0.8	2.75
8/17/21	0.4	0.65	0.7	0.75
8/18/21	0.1	0.8	0.7	2.75
8/19/21	0.45	0	0.9	0.37
8/20/21	0.4	0.78	0.1	2.75
8/21/21	0.5	0.7	0.8	0.67
8/22/21	0.2	0.2	0.4	0.3
8/23/21	0.6	0.63	0.4	0.86
8/26/21	0.4	1.23	1.1	0.72
8/27/21	0.5	1.81	0.4	2.75
Average:	0.520333333	1.174	0.4	1.265333333

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