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Environmental Studies Department Honors Thesis

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Abstract

Eutrophication presents a serious threat to America's aquatic ecosystems, negatively impacting both the aquatic life and the communities dependent on these bodies of water. Reducing nutrient inflow of nitrogen and phosphorus into waterways from point and non-point sources is critical in reversing the environmental degradation caused by eutrophication. Municipal wastewater treatment plants are one of the primary point sources of nutrient-rich effluent, and as such, implementing nutrient reduction strategies within the treatment process is an impactful step towards mitigating eutrophication. Grey infrastructure technologies that use mechanical or chemical treatment have historically been used for wastewater nutrient reduction. However, constructed wetlands have also been implemented for wastewater nutrient reduction. These systems mimic the biological and chemical processes that occur in natural wetlands to remove nutrients but in a more controlled environment. A life-cycle cost analysis is conducted to analyze differences between the total life cycle costs of constructed wetland systems and grey infrastructure improvements for nutrient removal from municipal wastewater treatment facilities. Furthermore, this paper evaluates whether the inclusion of ecosystem services generated by constructed wetlands significantly reduces their life-cycle costs. The results of this study suggest that CW systems are more cost-effective than grey infrastructure technologies for nutrient reduction when ecosystem services are included in the analysis. This study lays the groundwork for future research on the inclusion of ecosystem services into future life-cycle cost analysis for nutrient reduction and cost analyses for constructed wetland systems.

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1.0 Introduction

For decades, the world's oceans, rivers, lakes, and other aquatic ecosystems have been increasingly burdened and degraded due to excessive inflows of nutrients. Human activities now annually introduce equal to or greater fixed nitrogen into the environment than is naturally introduced (Smith et. al. 1999). The accumulation of high levels of phosphorus and nitrogen from agricultural, industrial, and municipal wastewater effluent is drastically decreasing the health and viability of marine and freshwater ecosystems. Eutrophication is the biological result of this influx of nutrients and is exacerbating algal blooms and the creation of dead zones in impacted marine environments (Smith et. al. 1999; Anderson et. al. 2002; Conley et. al. 2009). The explosion of algal blooms and dead zones drain aquatic ecosystems of oxygen that is critical in supporting aquatic life, and as a result, mass die-offs of aquatic species have been reported in eutrophied ecosystems (Heisler et. al. 2008). Furthermore, eutrophication not only negatively impacts aquatic species but also communities dependent upon such bodies of water for freshwater supply and economic opportunities (Chislock et. al. 2013; Michalak et. al. 2013). With algal blooms and dead zones becoming annual occurrences within many aquatic ecosystems, the United States Environmental Protection Agency (US EPA) is now calling for significant reductions in nutrient levels within wastewater effluent.

Green infrastructure is increasingly used as a viable and cost-effective alternative for a myriad of grey infrastructure (GI). Green infrastructure solutions either use or mimic the ecological functions provided by ecosystems to perform a variety of services such as wastewater treatment, source water protection, soil erosion mitigation, and urban run-off mitigation (Gartner et. al. 2013; TNC 2013). These green infrastructure solutions are seen as more beneficial than standard GI due to the additional ecosystem services they generate, a subject to be examined in depth within this study. These ecosystem services are environmental benefits performed by natural ecosystems and green infrastructure are not traded nor currently valued in traditional economic markets.

Constructed wetlands (CW) are one of the most successfully tested and implemented forms of green infrastructure. However, the benefits provided by wetland systems has only recently begun to be appreciated. Throughout past couple centuries, wetlands have been regarded as having little value beyond being a breeding ground for mosquitoes and waterborne diseases, and consequently, over fifty percent of wetlands around the globe have been converted for other uses since 1900 (Barbier 1993). In contrast to the global decline in natural wetlands across the globe, CW are increasingly built for municipal wastewater treatment (Steiner and Combs 1993; Kadlec and Knight 1996; US EPA 2000a; Vymazal and Kröpfelová 2008; Kadlec and Wallace 2009; Vymazal 2010). CW mimic the biological and chemical processes that occur in natural wetlands to treat wastewater and remove nutrients. The treatment capacity of CW can be maximized compared to traditional wetlands since the daily flow into the wetland can be controlled to ensure peak treatment efficiency (Kadlec and Knight 1996; Thom et. al. 1998; Neralla et al. 2000; Vymazal and Kröpfelová 2008; Kadlec and Wallace 2009). Furthermore, routine maintenance such as plant removal and porous media clearance ensure the constructed wetland functions properly.

As municipalities grapple with efforts to reduce nutrient levels in wastewater effluent, identifying cost-effective methods is critical. CW present such an option for municipalities trying to address new challenges in removing excess phosphorus and nitrogen from municipal wastewater. And while numerous studies have analyzed the cost-effectiveness of CW for nutrient level from agricultural run-off, no previous study has assessed the cost differences between CW and grey infrastructure (GI) upgrades for nutrient reduction at municipal wastewater treatment plants (WWTP). These grey infrastructure technologies include both mechanical and chemical processes that are implemented within existing WWTP to reduce nitrogen and phosphorus from the effluent. This paper seeks to analyze whether there are statistically significant differences between life-cycle costs of CW in comparison to GI technologies. This paper will also present an overview of the ecosystem services that CW generate (e.g. habitat creation, education, and recreation) beyond nutrient reduction, and how these benefits can make CW a less expensive option for municipalities looking to reduce nutrient levels.

The remainder of the paper is organized as follows: Section 1 will present introductory information on the causes and effects of eutrophication on aquatic ecosystems. This is followed by a broad introduction to modern centralized wastewater treatment methods used by municipalities with focus on key organic, inorganic, and nutrient compounds that wastewater treatment facilities must be capable of treating. Current GI technologies for reducing nitrogen and phosphorus levels in wastewater will also be described in this section. Following this introduction into traditional wastewater treatment, information on CW models currently implemented for wastewater treatment will be examined as well as an introduction to the biological and chemical processes that occur within CW to remove nutrients. This is followed by a description of the ecosystem services that are generated by CW, and lastly, a brief introduction is given for life-cycle cost analysis (LCCA), the economic methodology used in this paper. Section 2 will present this paper's research methodology, including the variables and regressions used in the analysis. Section 3 presents the case studies and data used in the LCCA. Section 4 will present the results of the regression, and lastly, Section 5 will examine any final conclusions and policy recommendations.

1.1 Eutrophication

1.1.1 Causes of Eutrophication

Eutrophication is the ecological response to surges in the availability of nutrients necessary for the growth of large algal and phytoplankton populations. While eutrophication can occur naturally due to sediment buildup over decades and centuries that trap nutrients within the ecosystems, point-source and non-point source pollution from human activities have drastically accelerated eutrophication globally (Carpenter 1981; Carpenter et. al. 1998; Chislock et. al. 2013; Michalak et. al. 2013). As a result, environmental governing bodies are increasingly analyzing how to reduce nutrient levels in the run-off and effluent from agricultural, industrial, and municipal wastewater sources (US EPA 2015). Significantly reducing nutrient levels from these point sources is critical for minimizing the accumulation of nutrients in the world's waterways because continued inaction will exacerbate negative impacts to aquatic ecosystems, local economies, and even human health.

1.1.2. Effects of Eutrophication

There are tremendous negative consequences that result from eutrophication, and the most evident is the impact upon the ecological health of aquatic ecosystems. The rapid growth, and subsequent death, of algal blooms and aquatic plant life promotes bacterial consumption of the decaying matter. These bacteria consume dissolved oxygen within the water, resulting in anoxic ecosystems (Chislock et. al. 2013). Due to the lack of oxygen, mass die-offs of aquatic fauna frequently occur in a condition called hypoxia (Diaz and Rosenberg 2008; Heisler et. Al. 2008). These algal blooms further inhibit aquatic plant growth as the algae that forms on the water's surface limit the amount of sunlight that can reach plants within the water, which limits the ability of plants to photosynthesize (Madden and Kemp 1996; Smith et. al. 1999). Eutrophication within the Chesapeake Bay is a classic case of the ecological damage this process causes. Decades of excessive nutrient loading from agricultural and municipal wastewater sources has drastically altered the ecological health of the Bay. Annual algal blooms have created small dead zones throughout the Bay, and once thriving populations of oysters and blue crabs have been crippled, in large part, by reduced oxygen levels (Malone et al. 1993; Boesch et. al. 2001; Kemp et. al. 2005). Without serious reductions in nutrient loading into the world's bodies of water, the continued eutrophication of aquatic ecosystems will spur further oxygen depletion and result in dire consequences for marine life.

Eutrophication not only has crippling effects upon marine life, but it also poses a threat to human health and well-being. Algal blooms can severely limit the economic productivity of eutrophic waterways, and it has been estimated that eutrophication causes over $2.2 billion United States dollars (USD) in annual damages within the United States (Dodds et. al. 2009; Chislock et. al. 2013) Recreation opportunities such as boating and other water sports can become prohibited due to dangerous cyanobacteria algal blooms, also known as blue-green algae. Furthermore, as seen in the Chesapeake Bay, eutrophication can negatively impact fish and shellfish populations, which limits the ability of local fishermen to make their living.

The human-related impacts of eutrophication go beyond economics and can affect cities ability to supply drinking water. Under certain conditions, cyanobacteria can produce cyanotoxins that can trigger serious health risks, and even death, when ingested (Paerl et. al. 2001; US EPA 2014). One dramatic example of the negative impact algal blooms can have upon drinking water supplies occurred during the summer of 2014 in Toledo, OH. With excessive levels of nitrogen and phosphorus accumulating for decades in the waters of Lake Erie, algal blooms have become a regular occurrence during the summer months. However, in 2014, the bloom covered hundreds of miles of the lake's western portion, precisely where the city of Toledo took their drinking water. With no cost effective methods available to remove the toxic algae from the water, over 400,000 residents were left without running water for several days (Lee 2014; Wines 2014; Yeager-Kozacek 2014). Without immediate actions to reduce the level of nutrients entering this nation's aquatic ecosystems, severe algal bloom, similar to the one that impacted Toledo, could very well become an annual occurrence that may disrupt the domestic water supply for millions of people. Eutrophication negatively affects both the health of aquatic ecosystems and the health and vitality of communities dependent upon such bodies of water. It is therefore imperative that steps be made to drastically reduce the nutrient levels being released from human activities into the nation's waterways.

1.1. Traditional Wastewater Treatment Processes

1.1.1. Primary Wastewater Treatment

The initial steps of centralized wastewater treatment processes involve removing large, untreatable objects that flow into the treatment plant. Large objects such as sticks, rags, and other solid debris cannot be broken down in the secondary and tertiary processes, and thus, preliminary treatment removes these objects before they can interfere with further treatment (US EPA 2004a). Screens stop these objects from flowing through the wastewater facility and are then collected and removed to appropriate solid waste facilities, such as landfills (US EPA 2004a). However, initial screening is often unable to remove smaller substances such as gravel and sand, which must also be removed before secondary treatment. In most centralized wastewater treatment systems, storm runoff mixes with residential wastewater, resulting in the accumulation of gravel and sand. Thus, another screening process performs degritting wherein these particles are stopped by smaller screens and settle to the bottom of the chamber (US EPA 2004a).

Suspended solids are often still present in wastewater after primary screening and degritting. These solids "consist of minute solid particles of matter that can be removed from the wastewater with further treatment such as sedimentation or gravity settling, chemical coagulation, or filtration" (US EPA 2004a). Settling tanks allow these solids to settle at the bottom of the tank where they form a layer of primary sludge (US EPA 2004a). Periodic removal of this sludge is necessary to prevent excessive build-up. Preliminary treatment of wastewater is necessary to allow for the biological and chemical processes used to purify the wastewater in secondary and tertiary treatment to remain effective.

1.1.2. Secondary Wastewater Treatment

The vast majority of biological and chemical treatment of wastewater occurs during secondary treatment. Anaerobic and aerobic bacterial digestion is the most common method for breaking down the organic matter within wastewater (Pescod 1992; US EPA 2004a). Aerobic digestion occurs in the presence of oxygen whereas anaerobic reactions occur without oxygen. Centralized wastewater treatment often uses suspended growth processes to create aerobic conditions for bacteria to thrive and breakdown the organic matter (US EPA 2004a). Wastewater enters the aeration tanks where oxygen-rich air is pumped in to create an aerobic environment, and allowing the bacterium to break down organic matter into activated sludge (Pescod 1992; US EPA 2004a). The activated sludge collects at the bottom of settling tanks where periodical removal prevents excessive buildup.

Steps to disinfect the wastewater also occur during secondary treatment. These techniques effectively remove harmful pathogens from the wastewater to prevent future contamination of local water systems. There are three primary methods used to disinfect wastewater. Chlorine is very effective in killing microorganisms and has been used in wastewater treatment for quite some time, however, new insight into the harmful effects of excessive chlorine in ecosystems has decreased the usage of this method (US EPA 2004a). Ozone is also effective in removing pathogens from wastewater. Furthermore, ozone has minimal long-term negative effects since the ozone breaks down into elemental oxygen (US EPA 2004a). Perhaps the most common method for disinfection is the use of Ultra Violet (UV) technology, which kills microorganisms by damaging their genetic material (US EPA 2004a). UV treatment provides the most cost-effective treatment method with the added benefit of no negative environmental by-products (US EPA 2004a).

1.2. Primary Substances Treated within the Municipal Wastewater Treatment Process

Municipal wastewater treatment facilities must be capable of removing a diverse range of organic, inorganic, and nutrient compounds. This section briefly examines a small portion of the most commonly tested substances to showcase why it is critical that any wastewater treatment strategy is capable of treating wastewater to meet US EPA effluent standards.

1.2.1 Total Suspended Solids (TSS) and Biodegradable Oxygen Demand (BOD)

TSS comprise all organic and inorganic matter that pass through primary screening in the treatment process. Excessive TSS in wastewater effluent can cause severe water degradation by decreasing the amount of sunlight that reaches the water, reducing the ability of aquatic plants to photosynthesize and lowering oxygen levels in the water (Rossi et. al 2006). Redcued oxygen levels can result in die-offs of aquatic species in these degraded waterways. Excessive TSS also settle on the bottom of waterways, prohibiting the growth of aquatic plants and fish species ability to lay eggs for reproduction (Rossi et. al 2006). To avoid such water degradation, the US EPA has established effluent standards for TSS at 45 mg/L (US EPA 2010).

Oxygen plays a critical role in allowing aerobic bacteria to treat wastewater, and BOD is the amount of oxygen required to effectively treat wastewater. It is necessary to measure BOD because if the wastewater contains, high levels of organic solids and ammonia, excessive oxygen demand for treating wastewater can leave the effluent oxygen-depleted (US EPA 2004a). Oxygen-deprived effluent can reduce oxygen in the receiving water source, affecting the ability of fish and aquatic plants to survive. Municipal wastewater treatments must minimize the BOD in wastewater to ensure that oxygen levels are not significantly depleted when the effluent leaves the WWTP. The standard measure for monitoring BOD is the five-day biological oxygen demand (BOD_5) of treated wastewater (US EPA 2004a). This is the amount of oxygen required to treat wastewater over a five day period (US EPA 2004a). To maintain a standard for environmentally safe BOD levels, the US EPA has established the maximum BOD_5 of effluent at 30 mg/L (US EPA 2010).

1.2.2. Nutrients: Phosphorus and Nitrogen

Excessive nutrient levels in effluent from wastewater treatment facilities have become a critical factor in the global degradation of rivers, lakes, and oceans. The two most impactful nutrients found in wastewater effluent are nitrogen and phosphorus. In excessive amounts, these nutrients cause eutrophication in waterways resulting in abnormally large algae blooms (US EPA 2004a). The subsequent die-off of the algal blooms fuels bacterial consumption of the decaying matter and oxygen within the water, forming dead zones that negatively impact aquatic life (US EPA 2004a; Carpenter 2008; Conley et. al. 2009). While the US EPA has ramped up efforts to reduce phosphorus and nitrogen in wastewater effluent, significant gaps persist in monitoring and limiting the nutrient levels in effluent (US EPA 2012). Even though national effluent standards are currently lacking for these nutrients, it is becoming increasingly critical that municipal wastewater systems effectively remove nitrogen and phosphorus to reduce the likelihood of future eutrophication occurring in effluent-receiving aquatic ecosystems.

1.3 Grey Infrastructure Improvements for Nutrient Reduction

As wastewater treatment facilities across the United States continue to grapple with lowering nutrient levels in their facilities' effluent discharges, technological and infrastructure improvements are increasingly important to provide additional treatment before releasing the effluent. There are currently a number of methods being implemented to achieve better nitrogen and phosphorus reductions. They can be classified into three groups (See Appendix C). Currently, the most common technology is biological nutrient removal (BNR) that improves upon the treatment facility's suspended growth treatment systems. These improvements allow bacteria within the activated sludge process to convert nitrate into non-impactful nitrogen gas (Hartman and Cleland 2007). In addition, this process binds phosphorus within the sludge that is then removed from the facility (Hartman and Cleland 2007). Furthermore, BNR can be used to allow nitrifying bacteria to convert excessive ammonia into nitrate, which can subsequently be broken down into nitrogen gas by bacteria later in the BNR process. A second GI method for nutrient reduction involves the use of a media surface for bacteria and biomass to grow on. This attached growth method allows for the bacteria to grow and perform nitrification and denitrification to remove nitrogen from wastewater effluent in either an aerobic or anaerobic environment.

There are two primary methods currently used for phosphorus removal from municipal wastewater effluent. The most common method employs chemical additives such as alum, lime or iron salts that bind to the phosphorus and cause the phosphorus to settle to the bottom of the holding chamber (Hartman and Cleland 2007). The phosphorus is then removed as part of the activated sludge. A more recent technology for phosphorus removal is enhanced biological phosphorus removal (EBPR). This technology modifies the activated sludge process within WWTP to create a cycle of aerobic, anaerobic, and anoxic conditions that typically result in high levels of phosphorus removal (Strom 2006; Hartman and Cleland 2007). EBPR provides many benefits for wastewater treatment facilities over the use of chemical additives including: reduced chemical usage, reduced energy demand, lower accumulation of settled sludge, and improved phosphorus removal (Park et. al. 1997; Hartman and Cleland 2007).

1.4. Constructed Wetlands

Wetlands are some of the most productive ecosystems on the planet, and as such, they provide a wide array of ecosystem services. While there is no singular definition for what constitutes a wetland, these ecosystems are nearly always wet year-round, or at least seasonally wet, and play host to a wide diversity of plant and animal life (US EPA 2004b). Wetlands are capable of converting common pollutants found in municipal wastewaters into simple by-products necessary for the wetland's plants to survive (Kadlec and Wallace 2009). This biological ability of wetlands to effectively treat wastewater has led to the development of constructed wetlands for the purpose of wastewater treatment and nutrient reduction. CW use the natural processes that occur within wetlands in a controlled setting to maximize treatment capability (Vymazal 2010). While CW can effectively treat and remove organic solids and nutrients present in municipal wastewater, it is often necessary that primary screening of the wastewater occurs before entering the CW (US EPA 2000). This preliminary treatment removes large, untreatable objects such as plant material and other floating trash from the wastewater to ensure that the CW maintains effective treatment capabilities.

CW share two key features. To prevent wastewater from seeping into and degrading local groundwater, non-permeable liners are placed at the bottom of the CW (Vymazal and Kropfelova 2008; Kadlec and Knight 2009). The next critical step in building a CW is identifying wetland plants that are both capable of handling the nutrient loading that occurs in the nutrient removal process and are native to the local ecology when applicable. While the plants used within the CW vary based on local preference, the most common plants used are cattails, bulrushes and reeds (US EPA 2000b). However, a wide variety of wetland plants can be utilized so long as the plants are capable of high organic and nutrient loadings, provide extensive roots and rhizomes systems for attached bacteria in both anaerobic and aerobic settings, and have sufficient biomass above the wetland's surface to provide adequate insulation during winter months, if the CW is built in a colder environment (Brix 1994a; Vymazal and Kropfelova 2008; Vymazal 2011). Substantial above-surface plant material also increases potential nutrient removal when the wetland plants are periodically harvested (Reddy and De Busk 1985; Thullen et. al. 2005; Vymazal 2011). While sharing these two features, constructed wetlands differ upon the way in which the wastewater passes through the wetland. The following section will examine the differences in the structure and treatment capabilities for free surface flow, horizontal subsurface flow, and vertical subsurface flow.

1.4.1. Free Surface Flow (FSF)

FSF CW are designed to closely mimic the appearance and function of natural wetlands. As seen in Figure 1, wastewater flows openly across the wetland where the organic matter and nutrients come into contact with emergent wetland vegetation (Kadlec and Wallace 2009).

Wastewater slowly filters across the wetland, which allows for most solid organic matter to settle at the bottom of the wetland where the plants and bacteria break down the organic solids (US EPA 2000b; Kadlec and Wallace 2009). The emergent flora also have a critical role in reducing the buildup of algae on the wetland's surface as well as providing heat insulation during the winter months (US EPA 2000b; Vymazal 2010). These services are critical in ensuring the FSF CW maintains peak treatment capabilities year-round even when the temperature drops during winter months. However, for municipalities that experience harsh winters, this heat insulation may be inadequate as FSF CW are prone to freezing over in such climates, resulting in reduced treatment capacities (Werket et. al. 2002).

Figure 1. Free Surface Flow Constructed Wetland (Kadlec and Wallace 2009)

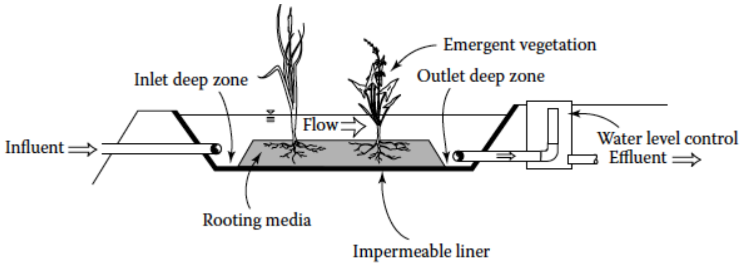


Figure 2. FSF Wetland at Arcata, CA (Humboldt State University 2016)

Since free surface flow constructed wetlands most accurately depict natural wetlands, they often provide habitat for a diverse range of wildlife (Kadlec and Knight 1996). Such wildlife habitation is often included as an ecosystem service generated by FSF CW. However, the open flow of wastewater has generated concerns regarding risk of human exposure to pathogens by coming in contact with the wastewater (Kadlec and Wallace 2009). These concerns have historically limited FSF CW to being used in wastewater treatment once pathogens have been treated by primary and secondary treatment methods (U.S. US EPA 2000a).

1.4.2. Subsurface Flow

SSF CW treat wastewater as it flows through a porous media below the surface of the wetland. Within the media, wastewater comes into contact with bacteria and plant roots that perform the biological treatment processes (Kadlec 2009; Kadlec and Wallace 2009). While studies have begun to analyze the varying levels of effectiveness for different SSF media, most SSF CW utilize a gravel medium (US EPA 2000c; Pant et. al. 2001). Due to the importance of maintaining the media, a major cost of SSF CW is clearing clogs within the media bed. Since the wastewater flows through the media in SSF CW, the effectiveness of the CW to treat the wastewater decreases with excessive buildup of organic matter (Nivala et. al. 2012). One major weakness of SSF CW is the lack of oxygen present throughout the wetland. While some oxygen is present around the plants' roots, nearly all of the treatment processes are performed in anaerobic conditions (Kadlec and Wallace 2009). Thus, the ability for these wetlands to breakdown ammonia through nitrification is a subject of concern, however, expanding the size of the CW has been shown to increase the ability of SSF CW to perform nitrification of ammonia (Vymazal 2006; Kadlec and Wallace 2009).

SSF constructed wetlands can be further categorized based upon the direction in which wastewater enters the wetland. In horizontal subsurface flow (HSSF) CW, wastewater enters from the side of wetland and continues to move through the gravel media (US EPA 2000c; Vymazal and Kropfelova 2008; Kadlec and Wallace 2009).

Figure 3. Horizontal Subsurface Flow Wetland (Kadlec and Wallace 2009).

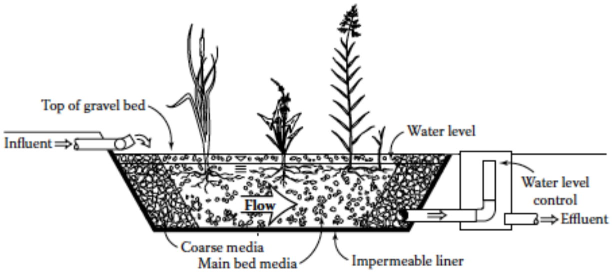


Figure 4. HSSF Constructed Wetland (Wastewater Gardens® Information Sheet 2016)

In contrast, vertical SSF (VSSF) CW, deliver a steady stream of wastewater over the top of the CW where it passes through an initial level of gravel before entering the porous media to be treated (Kadlec and Wallace 2009). This process adds oxygen to the the wastewater as it enters the gravel media, the wastewater as it enters the gravel media, allowing for aerobic nitrification to occur (Vymazal and Kropfelova 2008; Kadlec and Wallace 2009). Thus, VSSF CW are more capable of performing nitrification than HSSF CW, however, due to the incursion of oxygen into the wastewater, very little anaerobic denitrification occurs within VSSF CW (Vymazal 2010). These differences in treatment capabilities have led to considerations for hybrid CW systems that include both HSSF and VSSF to maximize treatment capabilities.

Figure 5. Vertical Subsurface Flow Wetland (Haberl 2012).

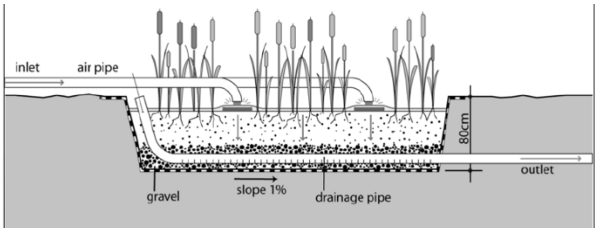


Figure 6. VSSF Constructed Wetland (Blumberg Engineers 2015).

1.5. Biological and Chemical Mechanisms for Wastewater Treatment in Constructed Wetlands

1.5.1. Total Suspended Solids (TSS) and Biodegradable Oxygen Demand (BOD) Removal

Constructed wetlands are very effective at removing total suspended solids and other organic matter from wastewater. In FSF CW, sedimentation of TSS occurs as the matter moves slowly across the wetland and steadily accumulates along the wetland's base (Kadlec and Wallace 2009). As the TSS accumulates, anaerobic bacteria breakdown the waste into harmless by-products (US EPA 1999; Kadlec and Wallace 2009). In contrast, the accumulation of TSS in SSF CW occurs within the gravel media bed. As in FSF wetlands, anaerobic bacterium break down the TSS as they accumulate in the gravel and along plant roots (Akratos and Tsihrintzis 2007; Al-Omari and Fayyad 2008; Greenway and Woolley 1999; Vymazal 2002). However, the long-term accumulation of these solids within the media can reduce the flow rate of the wastewater through the wetland (Vymazal and Kropfelova 2008; Kadlec and Wallace 2009). Within VSSF CW, additional aerobic breakdown occurs at wetland's surface as the intermittent dosing of wastewater allows for minor accumulation of organic matter along the CW’s surface (Kadlec and Wallace 2009). Aerobic bacteria then break down these TSS. Thus, there are greater maintenance requirements for SSF CW in removing excessive TSS build-up compared to FSF CW.

Constructed wetlands are efficient at keeping biological oxygen demand below Us EPA standards via anaerobic and aerobic bacteria. In HSSF CW, anaerobic bacteria attached to the aquatic plants' roots efficiently use oxygen present in the wastewater to decompose the solid waste and maintain low levels of BOD (Vymazal and Kröpfelová 2008; Crites et. al. 2010). Aerobic bacteria also break down oxygen within FSF CW due to the presence of bacteria in the litter bed that remove the oxygen present in the organic matter that accumulate along the bed (Crites et. al. 2010). Similar to FSF CW, VSSF CW are dependent upon aerobic bacteria that is attached to the media bed and plant material to minimize the amount of oxygen necessary to treat the wastewater (Crites et. al. 2010).

1.5.2. Phosphorus Removal

CW remove phosphorus from wastewater through various biological and chemical processes. A key removal mechanism is through soil accretion and plant uptake. Phosphorus accumulates in the litter bed along the base of FSF CW and within the porous media of SSF CW (Vymazal 2007). Similarly, plants within FSF and SSF CW uptake phosphorus from the wastewater (Kadlec and Knight 1996, Vymazal 2007). Studies have shown that nearly 20% of phosphorus removed by constructed wetlands occurs through accumulation in the soil base and plants of the CW (Richardson 1985; Vymazal and Kröpfelová 2008). However, excessive phosphorus accumulation along the litter bed, porous media, and plants of CW decreases the ability for the soil and plants to uptake and retain phosphorus (Kadlec and Knight 1996; Dunne and Reddy 2005; Vymazal 2007). Thus, it is necessary to routinely clear constructed wetlands' media bed and vegetation to ensure sustained phosphorus removal.

Phosphorus removal also occurs via adsorption within the gravel media of SSF CW. Adsorption is the chemical process in which the phosphorus binds to other elements within the gravel (Kadlec and Knight 1996; Yang et. al. 2001; Akratos and Tsihrintzis 2007; Martín et. al. 2013). This process is similar to the use of chemical polymers in WWTP that bind with phosphorus and cause it to settle down into the activated sludge. While there has been debate as to effectiveness of the standard gravel used in CW to react with phosphorus, higher concentrations of Al, Fe, or Ca ions in the media have been shown to increase the adsorption of phosphorus (Vymazal 2007; Vymazal and Kröpfelova 2008). Thus, it is possible for wastewater treatment operators to select media beds with high concentrations of these ions to increase the capacity of the SSF CW to bind to and remove phosphorus.

1.5.3. Nitrogen Removal

CW are efficient in removing nitrogen from wastewater effluent via biochemical processes. Ammonia is a common and environmentally-degrading nitrogen compound present in municipal wastewater. CW treat ammonia by breaking down the compound into nitrite, and subsequently nitrate, via aerobic bacteria that perform nitrification (Vymazal 2007). FSF and VSSF are more capable at breaking down ammonia than HSSF CW, which lack the oxygen necessary for aerobic bacteria to survive in sufficient numbers (Vymazal 2007; Kadlec and Wallace 2009). This is an intermediate step in the treatment process as the nitrate must subsequently be converted into gaseous nitrogen (Vymazal 2007). HSSF CW are capable of chemically converting aqueous ammonia into ammonia gas through volatilization (Saeed and Sun 2012). This process occurs when ammonia reacts with catalyzing ions in the gravel media. The reaction quickly converts aqueous ammonia into gaseous ammonia, and the gas diffuses to the CW's surface where it is safely released into the environment (Tanner et. al. 2002; Mayo and Mutamba 2004; Kadlec and Wallace 2009). There are significant differences in CWs' capacities to remove nitrate, another primary nitrogen molecule found in wastewater. For HSSF and FSF CW, the primary method for treating nitrate occurs via denitrification by anaerobic bacteria within the porous media and plant roots (Kadlec and Knight 1996; Yang et. al. 2001; Akratos and Tsihrintzis 2007). HSSF CW are more capable of performing denitrification given the anaerobic conditions present through the HSSF CW (Kadlec and Wallace 2009). In contrast, FSF and VSSF CW are less capable of treating nitrate due to the consistent inflow of oxygen within the CW. Thus, creating a hybrid CW system that utilizes a combination of these CW models can help achieve consistently effective nitrogen removal (Kadlec and Knight 1996; Vymazal 2007).

1.6. Wetlands and Ecosystem Services

Constructed wetlands provide ecosystem services not valued by traditional market costs and benefits (Boyer and Polasky 2004; Brander et. al. 2006). These services provide numerous benefits to human health and communities that encompass ecological, cultural, and economic well-being (de Groot et. al. 2012). Ecosystem services include ecological protection measures such wastewater treatment, habitat creation, nutrient removal, erosion control, climate regulation, and controlling water flow to minimize flooding (de Groot et. al. 2012). Cultural value can also stem from ecosystem services as these ecosystems provide opportunity for education, recreation, and even spiritual nourishment (de Groot et. al. 2012). Furthermore, some ecosystem services provide economic resources such as food, fresh water, medicinal supplies as well as raw materials such as timber for building (de Groot et. al. 2012). Ecosystems differ on the amount of ecosystem services generated, however, wetlands have been shown to generate significant ecosystem services covering a wide range of ecological, cultural, and economic services (Constanza et. al. 1997; Barbier et. al. 1997; Brouwer 1997; Brouwer 2000; Ghermandi et. al. 2009; Constanza et. al. 2014). The primary ecosystem services generated by wetlands are habitat creation, wastewater treatment, flood prevention, and erosion control (de Groot et. al. 2012). Furthermore, wetlands provide some of the highest values for recreation and education amongst the world's ecosystems (de Groot et. al. 2012). Wetlands generate significant ecosystem services that benefit both human health and livelihoods, and as such, these services must be given greater focus when analyzing the life-cycle benefits of such ecosystems and green infrastructure systems such as CW that mimic the biologically functioning of natural wetlands.

CW and other green infrastructure systems generate ecosystem services through mimicking natural ecosystems. Thus, CW provides similar ecosystem services to natural wetlands such as nutrient reduction, erosion control, flood abatement in addition to cultural services such as education and recreation. Previous studies have analyzed the values of ecosystem services generated by CW including habitat biodiversity, cultural education, nutrient removal, recreation and carbon sequestration (Hansson et. al. 2005Anderson and Mitsch 2006; Ghermandi et. al. 2009; Moore and Hunt 2012). It is critical that these services are accounted for in the overall life-cycle costs of CW systems because failing to do so could significantly undervalue the life-cycle costs of CW in comparison to GI systems. By incorporating these ecosystem services into the life-cycle cost analysis of CW for wastewater nutrient reduction, policy makers and wastewater management personnel can begin to quantify the true life-cycle costs of CW systems.

1.7. Life-cycle Cost Analysis (LCCA)

LCCA is an effective methodology for analyzing and comparing the total costs incurred by a project over operational life-cycle. LCCA has increasingly been used in analyzing the cost-effectiveness and treatment capacity of CW systems over a set time period (Balkema et. al. 2002; Dixon et. al. 2003; Machado et. al. 2007; Zhou et. al. 2009; Fuchs et. al. 2011; Corominas et. al. 2013). This methodology provides a strong means to calculate the potential cost savings presented by CW for nutrient reduction as compared to GI improvements. Given varying levels of implementation and O&M costs between CW and GI systems, LCCA allows the total costs of the systems to be compared over a life-cycle as opposed to simply comparing these values directly. For example, in this analysis, it is anticipated that CW will have higher implementation costs than GI nutrient reduction technologies. However, the anticipated lower O&M costs for CW over their life-cycle very well may make CW a less expensive alternative to GI technologies with higher annual O&M costs. This thesis employs this methodology to analyze the cost effectiveness of CW for nutrient reduction as compared to GI wastewater treatment plant improvements over a twenty-year life cycle.

2.0 Methodology

This analysis expands upon prior use of LCCA research into CW by including both CW and GI systems for municipal wastewater nutrient reduction to test whether CW are a cheaper alternative to GI methods for municipal wastewater nutrient reduction. The case studies used in the study were researched using secondary sources from a range of online scientific journals, primary literature and online data sets on CW and GI. Furthermore, for select CW, direct correspondence with the systems' managers was also used for data collection (Corona 2016; Finden 2016; Helton 2016; Huebotter 2016; McNerney 2016; Pomroy 2016; Sees 2016; Wilson 2015). The results of the LCCA will reveal whether CW are a cheaper alternative for nutrient reduction at a statistically significant level.

Prior to formulating the regressions used in the analysis, the mean values for the dependent and independent variables were calculated to identify noticeable patterns and differences between the values (See Appendix D). While the mean total cost values showed that GI systems were the cheaper option, the variations in the ecosystem services generated by the CW systems suggested that some systems may cost less than GI technologies when these services are included in life-cycle costs and benefits. Furthermore, the mean values for daily flow did not present any identifiable trends. To more accurately test for the significance of ecosystem services upon the total cost of CW systems, regressions are used in the LCCA. Using regressions allows multiple variables to be tested against select dependent variables in order to determine whether such variables significantly impact the values and difference in values between GI and CW systems. The regressions run in this analysis also provide more accurate results into whether the total costs of GI and CW systems differ based upon their daily flow.

Three regressions are run in this analysis. The first regression uses the total lifetime cost for the systems, TC, as the dependent variable. The TC values were computed using the one-time implementation costs for the projects added together with the annual O&M costs. The O&M costs were extrapolated for a twenty-year lifetime. This time-frame was selected due to other wastewater technologies LCCA analyzing CW and wastewater treatment technologies using a twenty-year life cycle (Abraham 2003; Ugarelli et. al. 2008; Molinos-Senante et. al. 2013). While GI and CW systems can remain in operation beyond this time, the twenty-year life cycle provides standardization to the values used in the regression, specifically the O&M costs and ecosystem services values (Abraham 2003; Ugarelli et. al. 2008; Molinos-Senante et. al. 2013). The first regression (Equation 1) will therefore use the independent variable TC to analyze if there are significant differences in the lifetime costs of CW and GI technologies.

The second regression in this analysis will include ecosystem services as net benefit for CW. The dependent variable, TC_Eco, will show whether the inclusion of ecosystem services in the LCCA of CW systems produce significantly lower total costs for CW in comparison to when such values are excluded. As previously mentioned in the introduction section, CW provide numerous ecosystem services on top of nutrient reduction and wastewater treatment. The values used to generate the ecosystem services for the CW system were calculated in de Groot et. al. 2012. These values were computed for inland wetlands systems, therefore it is necessary to use the benefit transfer method to use these values for CW. The benefit transfer method is used to transfer the value of ecosystem services calculated in previous studies to case studies where such values have yet to be calculated (Plummer 2009; Jenkins et. Al. 2010; Brander et. al. 2013). In this analysis, the values for three ecosystem services were chosen: habitat creation (2806 USD/ha/year), education (1477 USD/ha/year), and recreation (2527 USD/ha/year). These services were selected due to the high number of CW managers who reported that these were the primary ecosystem services generated by their respective CW systems (Corona 2016; Finden 2016; Helton 2016; Huebotter 2016; McNerney 2016; Pomroy 2016; Sees 2016; Wilson 2015). To create the values for TC_Eco, the lifetime ecosystem services generated by the CW systems were calculated for a twenty-year life-cycle using the three ecosystem services listed above. Once the lifetime ecosystem services values were calculated, these values were subtracted from the original TC value to generate the TC_Eco values. The second regression will therefore test whether the inclusion of ecosystem services in the valuation of CW systems significantly impact whether CW are cheaper than GI technologies for nutrient reduction over their life-cycle.

The third regression will analyze the average value of ecosystem services provided by CW systems over their life-cycle as well as the impact of the other observed variables on the generation of ecosystem services. As previously addressed, CW provide additional benefits beyond wastewater treatment and nutrient reduction. This regression will provide insight into whether the independent variables significantly affect the value of ecosystem services generated by CW. These results may suggest that certain conditions maximize ecosystem services generated by CW systems for nutrient reduction. The Eco_Services values are equal to the values of the recreation, education, and habitat creation ecosystem services generated by the CW over the life-cycle. For GI technologies, the value 0 was assigned since ecosystem services are only generated by natural ecosystems and green infrastructure systems. Therefore, it is assumed that GI technology do not generate ecosystem services (Isely et. al. 2010; Tiwary and Kumar 2014). The regression ultimately provides wastewater treatment managers a life-cycle valuation for the ecosystem services provided by a potential CW for nutrient reduction and whether certain independent variables impact the amount of ecosystem services generated by CW systems.

Each regression is run with robust standard errors to correct for heteroskedasticity. To reduce omitted variable bias, the observed variables listed below are included in each regression model. Reducing omitted variable bias aids in controlling for variables not included in the regression that may impact cost values. Furthermore, all monetary values are given in terms of United States Dollars (USD) and are adjusted for inflation up to 2015.

To conduct the regression, it is necessary to create dummy variables to separate the CW systems and the GI case studies. The variable, CW, will be one independent variable in the regressions. CW case studies will be designated as (0), while GI improvements are designated as (1). This variable will show whether the TC for CW are significantly cheaper or more expensive compared to GI technologies for nutrient reduction.

The second independent variable is the daily flow of the treatment systems. The variable, D_flow, is included to see if the daily flow through the treatment system significantly impacts the systems' life-cycle costs. Municipalities can use these results to determine whether a CW or a GI technology would be cheaper nutrient reduction given the WWTP's daily wastewater load. The third variable, GreyxD_Flow, is generated by assigning the system's D_Flow value with the system's corresponding CW dummy variable. This variable is tested to observe whether increases in the daily flow impact the costs of CW and GI technology differently.

With the independent and dependent variables describes, three regressions are run to test for statistical significance between CW and GI technologies for nutrient reduction. Epsilon is the error term for all variables not included in the regression.

Equation 1. Regression Model with TC as Independent Variable

$$TC= \beta_0+\beta_1*CW+\beta_2*D\_Flow+\beta_3*GreyxD\_Flow +\varepsilon$$

Equation 2. Regression Model with TC_Eco as the Independent Variable

$$TC\_Eco= \beta_0+\beta_1 CW*+\beta_2*D\_Flow+\beta_3*GreyxD\_Flow +\varepsilon$$

Equation 3. Regression Model with Eco_Services as the Independent Variable

$$Eco\_Services=\beta_0+\beta_1 CW^{*}+\beta_2{*}D\_Flow+\beta_3{*}GreyxD\_Flow+\varepsilon$$

The independent variable for the EPA Region within each treatment system is located was included in three additional regressions. This variable was initially excluded from the analysis due to the potential for the same sample size per region, which could have fixed effects upon the results. Nevertheless, the results will suggest whether regional location significantly impact systems' life-cycle costs. For the purpose of this analysis, the regions were assigned using the EPA's ten regions across the United States (see Appendix D). Table 1 lists the dummy variable assigned to each U.S. EPA region. The region is listed in the top row, while the corresponding dummy variable is listed in the row below.

Table 1: Dummy Variables for Region Values

EPA Region	9	4	8	6	3	2	10	7	1
Dummy Variable	0	1	2	3	4	5	6	7	8

EPA Region 9 was given the variable (0) due to the high frequency of CW systems in the states of California and Arizona. Therefore, the results from the analysis will show whether there are significant differences in the total costs of systems in EPA Region 9 as compared to those located in other regions.

With the inclusion of the independent variable Region included, three additional regressions are run to test for statistical significance between CW and GI technologies for nutrient reduction.

Equation 4. Regression Model with TC as Independent Variable

$$TC=\beta_0+\beta_1{*}CW+\beta_2{*}D\_Flow+\beta_3{*}GreyxD\_Flow+\beta_4{*}Region+\varepsilon$$

Equation 5. Regression Model with TC_Eco as the Independent Variable

$$TC\_Eco=\beta_0+\beta_1 CW^{*}+\beta_2{*}D\_Flow+\beta_3{*}GreyxD\_Flow+\beta_4{*}Region+\varepsilon$$

Equation 6. Regression Model with Eco_Services as the Independent Variable

$$\text{Eco\_Services}=\beta_0+\beta_1\text{CW}^*+\beta_2{}^*\text{D\_Flow}+\beta_3{}^*\text{GreyxD\_Flow}+\beta_4{}^*\text{Region}+\varepsilon$$

Epsilon is the error term for all variables not included in the regression. Positive coefficients for the variable CW will suggest CW systems are cheaper in comparison to GI improvements because CW are assigned the dummy variable (0). In contrast, a negative coefficient will suggest that GI technologies are cheaper than CW systems. It is expected that the coefficient will be positive for CW in both regressions, however, due to the inclusion of ecosystem services as net benefits for CW, and it is expected that the coefficient will be greater in Equation 2 as compared to Equation 1. For Equation 3, the coefficient is expected to be negative since GI technologies do not generate ecosystem services. The coefficient will therefore suggest the average value of ecosystem services generated by CW systems over the twenty-year life-cycle.

For the variable Daily_load, the coefficient is anticipated to be positive. This will suggest that the cost savings from nutrient reduction technologies are higher for WWTP that have higher daily flow rates. This is also anticipated in Equation 3 as higher daily flow are typically associated with larger CW systems, which maximizes the ecosystem services generated. The coefficient for the variable GreyxD_Flow is anticipated to be positive such that a 1 MGD increase in daily flow will generate higher costs for traditional infrastructure in comparison to CW. With the inclusion of this variable, the total cost increase per 1 MGD increase in wastewater flow for traditional infrastructure technologies will be calculated from the coefficients $\beta_1 + \beta_3$. Thus, a positive value for the coefficient β_3 will suggest that the cost increases resulting from 1 MGD increase in daily flow are greater for GI than CW systems.

For the variable Region, it is anticipated that all of the Regions will have negative coefficients. This suggests that nutrient reduction technologies outside of EPA Region 9 are more costly than those located within Region 9. This is predicted since larger FSF CW can be built in the warmer climates of California, Arizona, and Nevada, which increases the ecosystem services generated by these CW systems.

3.0 Data

See Appendix 3 for the complete list of case studies, values, and sources used in this study. The case studies used in this analysis received effluent flow levels ranging between 0.5 MGD to 20 MGD, serving communities ranging from 1000 to 29,000 people. Therefore, these systems provide nutrient reduction service for small to medium-sized towns and cities. Treatment systems serving communities smaller than and larger than this threshold were excluded to keep the scope of this study focused on small to mid-sized towns.

3.1 Constructed Wetland Data

After an extensive review of the available literature, fifty-three constructed wetland systems were selected for the analysis. These CW were selected due to their use in receiving primary and secondary treated wastewater from centralized WWTP and use in removing excess nutrient from the wastewater effluent. To ensure a sense of uniformity amongst the cases studies, only systems located in the United States are included in this analysis. Previous analyses on the costs of wastewater treatment have separated projects between those in developed countries and those in developing countries. This is primarily due to the majority of CW in developed countries being used for secondary or tertiary wastewater treatment. In contrast, CW in developing countries are increasingly used for primary wastewater treatment (Haberl 1999; Kivaisi 2001; Massoud et. al. 2009; Hernández-Sancho et. al. 2015).

Cost function equations were used to compute select implementation and O&M cost values in a select number of CW systems due to a gap in available data. In order to compute missing values for the implementation costs of CW, the equation computed by Kadlec and Knight 2009 was utilized for FSF and SSF CW:

$$\text{FSF: Implement Costs} = 194 * A^{0.690}$$

$$\text{SSF: Implement Costs} = 652 * A^{0.704}$$

A=area of CW (ha)

For missing yearly operation and maintenance (O&M) costs of FSF CW, the equation presented in US EPA 2000b was chosen:

$$\text{O\&M Costs} = 1533 \text{ USD/ha/year}$$

The equation given in US EPA 2000c was selected to compute missing O&M costs for SSF systems:

$$\text{O\&M Costs} = 60{,}000\text{/year/1 MGD}$$

Using these equations allowed for missing cost values to be computed and increase the number of CW systems that could be included within the analysis.

3.2. Grey Infrastructure Data

Following a review of existing literature and databases, eighty-one case studies for GI improvements for nitrogen and phosphorus removal from wastewater treatment plants were selected for this analysis. Three larger analyses provided the values for the implementation and yearly maintenance costs of these upgrades (Chesapeake Bay Program 2002; Hartman and Cleland 2007; US EPA 2015). The Chesapeake Bay Program, a multi-organization partnership focused on the protection and restoration of the Chesapeake Bay, calculated the costs that would be incurred by regional wastewater treatment plants in order to meet varying levels of nitrogen and phosphorus reduction, from Tier I to Tier IV. These tiers were established by the Chesapeake Bay Program to create varying standards of nutrient reduction for wastewater treatment plants to achieve. This progression allows the plants to cost-effectively increment plant upgrades to lower effluent nutrient levels. For the purpose of this analysis, the costs required for the lowest level of nutrient reduction was selected to keep the level of nutrient reduction between GI technologies and CW systems constant. Thus, the valued costs for the technology improvements were either for reductions to Tier II or Tier III (See Table 1).

Table 2. Nutrient Reduction Levels for Tier II and III (Chesapeake Bay Program 2002)

	Total Nitrogen (mg/L)	Total Phosphorus (mg/L)
Tier II	TN=8	TP=1.0 or permit limit if less
Tier III	TN= 3.0	TP=0.5 or permit if less

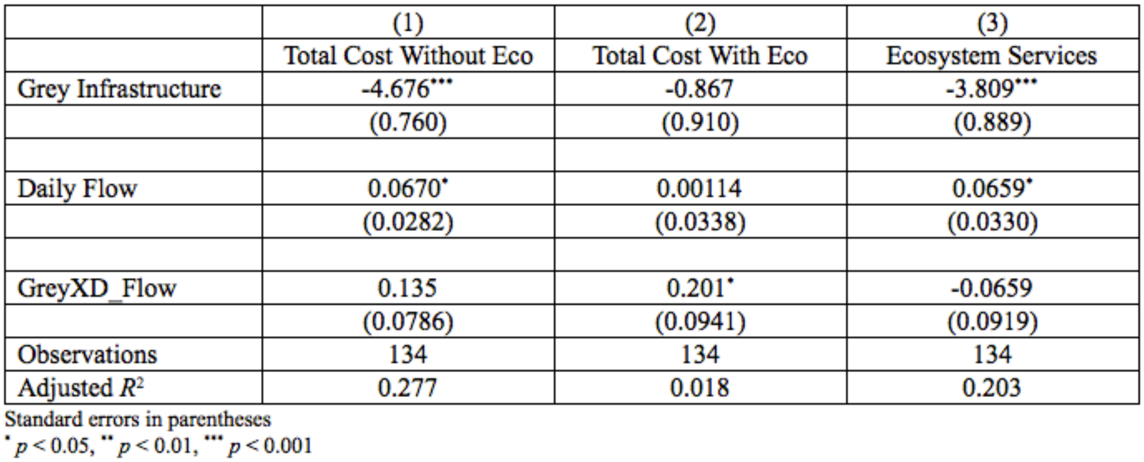
Furthermore, only improvements at "Significant Facilities" with discharge flows greater than 0.5 MGD were selected for the purpose of this analysis to ensure the daily flows of these systems were similar to those of the CW case studies.

4.0 Results

The regression results for Equations 1-3 are presented in Table 3. The results suggest that the inclusion of ecosystem services does impact whether CW systems are less costly over their life-cycle as compared to GI technologies. The results from Equation 1 indicate that GI systems are less costly than CW systems for nutrient reduction when only implementation and maintenance costs are included in the life-cycle analysis. The coefficient value for the observed variable CW, -4.676, suggests that GI technologies generated nearly 4.7 million USD in cost-savings as compared to CW systems. This result does not support the hypothesis of this paper, however, the results from Equation 2 paint an intriguing picture in regards to the role ecosystem services have on the life-cycle costs of CW systems.

Equation 2 included ecosystem services generated by CW systems as a net benefit generated over the CW's life-cycle and lowering the total life-cycle costs of the systems. The result from the regression suggest that the life-cycle costs of CW systems are comparable to GI technologies when ecosystem services are included in the analysis. The coefficient for the observed variable CW, -0.867, suggest that, when ecosystem services are included in the life-cycle analysis, GI technologies only generated approximately 800,000 USD in cost-savings over the life-cycle. This difference in total costs are negligible and therefore suggest that the inclusion of ecosystem services make CW systems a cost comparable solution to GI technologies for nutrient reduction from municipal wastewater. And lastly, the results from Equation 3 for the observed variable CW, -3.809, suggest that CW systems generated approximately 3.8 million USD in ecosystem services over a twenty-year life-cycle.

Table 3. Regression Results for Equations 1-3



	(1) Total Cost Without Eco	(2) Total Cost With Eco	(3) Ecosystem Services
Grey Infrastructure	-4.676^{***}	-0.867	-3.809^{***}
	(0.760)	(0.910)	(0.889)
Daily Flow	0.0670^{*}	0.00114	0.0659^{*}
	(0.0282)	(0.0338)	(0.0330)
GreyXD_Flow	0.135	0.201^{*}	-0.0659
	(0.0786)	(0.0941)	(0.0919)
Observations	134	134	134
Adjusted R^2	0.277	0.018	0.203

Standard errors in parentheses
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

The additional observed variables provide intriguing results in regards to their impact on life-cycle total costs. Within each regression, the coefficients for the variable Daily Flow are all positive, which is not unexpected given that an increase in the wastewater flow handled by the system would require greater yearly maintenance costs to ensure maximum treatment capability. However, in both Equation 1 and 2, the coefficients for Daily Flow are below 100,000 USD per increase of 1 MGD over the system's life cycle. Thus, the results suggest that, while higher daily flows inevitably increase the life-cycle costs of nutrient reduction systems, these cost increases are relatively low per 1 MGD increase in wastewater flow. The results from Equation 3 suggest that a daily flow increase of 1 MGD increases the value of a CW's ecosystem services by nearly 60,000 USD over its life-cycle. This result was anticipated as CW systems with higher daily flows often require larger area footprints, thereby increasing the ecosystem services generated by the CW.

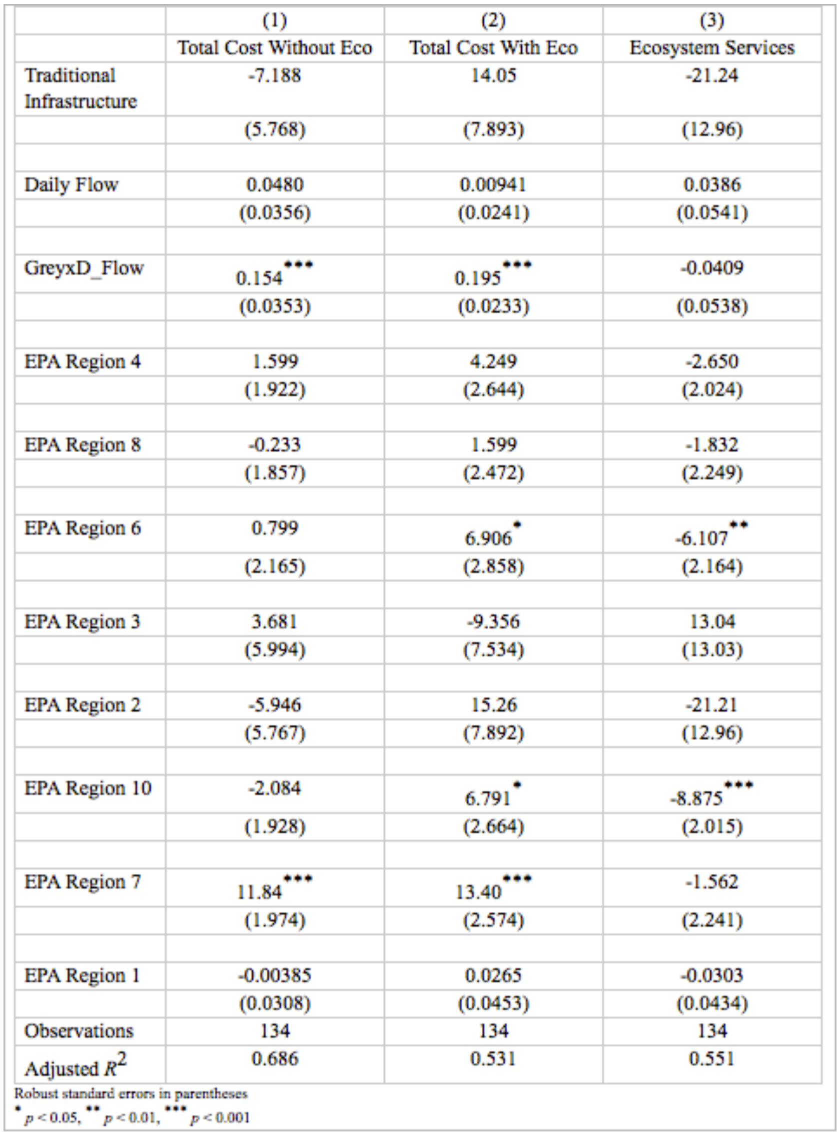
The coefficients for the observed variable GreyxD_Flow suggest that increases in daily flow generate greater total costs for GI technologies than for CW systems at a statistically significant level. The coefficient values for Equation 1 and 2 are 0.154 and 0.195, respectively. Thus, a 1 MGD increase in daily flow increases the total costs for GI improvements by over 130,000 USD as compared to CW and by 200,000 USD when CW ecosystem services are included in the life-cycle analysis. These results suggest that life-cycle costs for CW systems increase less than those of GI technologies when daily flow increases.

The results obtained from the second set of three regressions are presented in Table 4. It is important to note that the addition of the independent variable Region did have fixed effects upon the results and produced high robust standard errors. This is assumed to be caused by the small sample size of values included per region group. The regression results for Equations for the variable CW differ by a substantial margin from the results of Equations 1, 2 and 3. The results' robust standard errors for the variable CW are all much greater than 1, therefore, the results are less precise due to the varying sample sizes within the variable Region. In Equation 4, the coefficient for CW, -7.188, suggests that that GI systems are cheaper than CW for nutrient reduction when only implementation and maintenance costs are included in the life-cycle analysis by approximately 7 million USD. Compared to the coefficient for Equation1, this is a difference of 2.512, approximately 2.5 million USD. In Equation 5, the coefficient for the independent variable CW, 14.05, suggest that, when ecosystem services are included in this life-cycle analysis, CW systems for nutrient reduction generate over 14 million USD in life-time cost savings compared to GI nutrient reduction technologies. While the coefficient for CW in Equation 2 suggests that the difference in the life-cycle costs of GI and CW systems are negligible, the results from Equation 5 suggest that CW generate over 14 million USD in cost savings over the life-cycle in comparison to GI technologies. Lastly, the coefficient for CW from Equation 6, -21.24, suggest that CW systems generate over 21 million USD in ecosystem services over the life-cycle. This is a difference of approximately17 million USD in comparison to the results in Equation 3. Thus, the inclusion of the variable Region had a significant impact on the results and produced high robust standard errors, and therefore, the primary focus of these results is on the statistical significance of the results for the variable, Region.

The results for the observed variable Region provide insight into the effect that location has upon the total cost and ecosystem services provided by CW. In Equation 1, the results suggest that overall, CW systems outside of EPA Region 9 are costlier than those located within California, Arizona, and Nevada. However, for systems in Regions 2 and 10, the results suggest that these systems' total costs are lower than those within EPA Region 9. The value of these variables coefficients, -5.946 and –2.084, suggest that CW systems in these regions are much cheaper than those located in Region 9. These results are most likely due to the smaller size of CW systems located in these regions. This analysis is supported by the coefficients for the variables in Equation 2 and 3 when ecosystem services are included within the independent variable. For Region 2, the coefficients for Equation 2 and 3 are 15.26 and –21.21, respectively, while the coefficients for Region 10 are 6.791 and –8.875. Therefore, the results suggest that CW systems in EPA Region 9 generate greater ecosystem services than those in Regions 2 and 10.

This could be attributed to larger FSF CW being used in Region 9 due to the warmer climate. As previously discussed, a larger area footprint increases the ecosystem services generated by CW. Thus, the cheaper total cost of CW in Region 2 and 10 from Equation 1 can be attributed to these systems being smaller than those within Region 9, which is why the ecosystem services generated by these CW are less than those in Region 9 systems. Overall, the results suggest that CW systems located in EPA Region 9 are generally cheaper than those elsewhere in the United States, and this is due in large part to the high ecosystem services generated by these systems.

Table 4. Regression Results for Equations 4-6



	(1)	(2)	(3)
	Total Cost Without Eco	Total Cost With Eco	Ecosystem Services
Traditional Infrastructure	-7.188	14.05	-21.24
	(5.768)	(7.893)	(12.96)
Daily Flow	0.0480	0.00941	0.0386
	(0.0356)	(0.0241)	(0.0541)
GreyxD_Flow	0.154^{***}	0.195^{***}	-0.0409
	(0.0353)	(0.0233)	(0.0538)
EPA Region 4	1.599	4.249	-2.650
	(1.922)	(2.644)	(2.024)
EPA Region 8	-0.233	1.599	-1.832
	(1.857)	(2.472)	(2.249)
EPA Region 6	0.799	6.906^{*}	-6.107^{**}
	(2.165)	(2.858)	(2.164)
EPA Region 3	3.681	-9.356	13.04
	(5.994)	(7.534)	(13.03)
EPA Region 2	-5.946	15.26	-21.21
	(5.767)	(7.892)	(12.96)
EPA Region 10	-2.084	6.791^{*}	-8.875^{***}
	(1.928)	(2.664)	(2.015)
EPA Region 7	11.84^{***}	13.40^{***}	-1.562
	(1.974)	(2.574)	(2.241)
EPA Region 1	-0.00385	0.0265	-0.0303
	(0.0308)	(0.0453)	(0.0434)
Observations	134	134	134
Adjusted R^2	0.686	0.531	0.551

Robust standard errors in parentheses

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

5.0 Analysis and Recommendations for Future Research

The results from the life-cycle analyses provide useful suggestions into the costs of different methods for municipal wastewater nutrient reduction as a means to minimize the anthropogenic eutrophication of the nation's aquatic ecosystems. GI technologies are the cheaper technology when ecosystem services are excluded from the LCCA. Thus, given that most cost-benefit analyses do not take into account ecosystem services, the results suggest that wastewater treatment managers are more likely to select a GI improvement over a CW system. The primary explanation for these results is that the GI improvements for nutrient reduction often require simply upgrading the existing wastewater treatment plant as opposed to having to implement a new CW system. Furthermore, these nutrient removal upgrades may only present minor increases in O&M costs due to increased energy and chemical usage or the need for increased sludge removal. In contrast, CW require consistent maintenance of the wetland's plants and porous media in order to ensure peak treatment capacity. Thus, the results of the first LCCA disproved the hypothesis that CW systems are cheaper than GI technologies for nutrient removal, however, this analysis also suggests that the inclusion of ecosystem services provided by a CW has a strong impact on such systems lower life-cycle costs.

The inclusion of ecosystem services in LCCA of constructed wetlands is critical when comparing the costs of technologies for nutrient reduction. The results from the analysis suggest that the average CW system will generate over 20 million USD in ecosystem services over a twenty-year life-cycle. The impact of ecosystem services on the total life-cycle costs of CW systems was shown in the second regression analysis and suggests that CW systems are cheaper than GI improvements when ecosystem services are included as net benefits. Thus, CW systems present a cheaper solution for nutrient reduction when ecosystem systems services are included in the LCCA. However, this analysis only used the values for three ecosystem services generated by the CW: habitat creation, education, and recreation. Thus, it is probable that this analysis has undervalued the total ecosystem services generated by the CW used in the analysis. It is not unrealistic to predict that, if the complete range of ecosystem services generated by the CW were included in this analysis, that the value of ecosystem services would be much higher and further decrease the life-cycle costs of CW systems in comparison to GI technologies.

This study has shown how including ecosystem services in a LCCA provides a more comprehensive comparison between CW systems and GI technologies for wastewater nutrient reduction. For too long, the ecosystem services generated by green infrastructure such as CW have not been included when analyzing the life-cycle costs of infrastructure projects. Creating a more comprehensive analytic methodology is critical in allowing wastewater treatment managers to analyze whether a CW is the cheaper option for nutrient reduction. While value benefit transfer was used to compute the ecosystem services provided by the CW used in this analysis, developing more precise methodologies for computing the ecosystem service provided specifically by CW for wastewater nutrient reduction will allow future LCCA to incorporate site-specific ecosystem services. It is important that wastewater treatment managers considering a constructed wetland system can accurately value the ecosystem services the wetland will produce over its life-cycle. For example, a smaller HSSF CW in the Midwest will most likely not generate the same services as a large FSF CW in a warmer climate, and therefore, these CW will produce different values and types of ecosystem services. In my correspondence with site managers who have previously decided to implement a constructed wetland for nutrient reduction, none of the mangers stated that the value of potential ecosystem services had been calculated to the system being implemented. Rather, these CW systems were selected because the alternative GI solution proved too costly for the municipalities. Thus, it is apparent that even in cases where CW systems are selected for nutrient removal, there are not concrete evaluations for the potential ecosystem services that will be generated over the constructed wetland's life cycle. Therefore, it is imperative that wastewater treatment managers have the resources necessary to accurately value the expected ecosystem services a constructed wetland system will generate.

There are a range of variables not included in the scope of this analysis that future research can analyze to continue the discussion on the role of ecosystem services in LCCA for both CW and other green infrastructure systems. While the independent variable Region was included in the second set of regressions, it would be intriguing for future research to incorporate the mean temperature of the CW system within the LCCA. Using temperature, rather than region, would provide results on whether variances in a system's temperature and location significantly impact their life-cycle costs and generated ecosystem services. Thus, this variable could allow future research to suggest whether CW in warmer southern states produce significantly higher ecosystem services as compared to those located in cooler northern climates. Furthermore, this variable could suggest whether there are certain temperatures that help to maximize ecosystem services, and if so, then municipalities in such climates such begin to take a serious look at implementing a constructed wetland system. A second variable future research should analyze are the type of constructed wetland and how the costs and ecosystem services generated by these CW models differ. As explained in the introduction section of this paper, there are three primary CW models: free-surface flow (FSF), horizontal subsurface flow (HSSF), and vertical subsurface flow (VSSF); however, due to the sample size of CW used in this analysis, it was decided to include all CW models together. Thus, future research should seek to analyze whether different ecosystem services are generated by the different CW models. If the research suggests this, city officials and policy makers could determine which CW model would generate the most ecosystem services that are deemed most important for the surrounding community.

It is also important to keep in consideration that the wastewater treatment systems both the GI and CW technologies were being implemented for were treating populations ranging from approximately 1,000 to 29,000 community members. Thus, these communities are primarily small to middle-sized municipalities, and so, much smaller communities, single-sized homes, and larger metropolitan cities are not included in the scope of this analysis. Future research focused on these sized towns and cities could suggest new recommendations on whether CW systems are a less costly option for nutrient reduction depending on various population levels. This research will greatly expand the scope of LCCA for CW systems to include a larger range of population sizes. Ultimately, the results from such studies will allow wastewater treatment managers from various sized communities to analyze the amount of ecosystem services and the total life-cycle costs of potential CW systems.

As eutrophication continues to threaten the health of both aquatic ecosystems and communities dependent on them, it is important that future research is dedicated to understanding how wastewater treatment plants can cost-effectively reduce nutrient levels in wastewater effluent. This analysis has suggested that ecosystem services generated by CW systems greatly reduce the life-cycle cost of these systems, therefore, future research on valuation methodologies for ecosystem services will enhance the ability for wastewater treatment managers to accurately value the ecosystem services a potential CW could provide over its life-cycle. The results of this analysis suggests that CW systems should be viewed as a cost-effective solution for municipal wastewater nutrient reduction and an integral part for reversing the rampant anthropogenic eutrophication that has inflicted devastating impacts upon the world's aquatic ecosystems.

Appendix A: Abbreviations

BNR: Biological nutrient removal
BOD: Biodegradable oxygen demand
CW: Constructed wetlands
EBPR: Enhanced biological phosphorus removal
FSF: Free-surface flow
GI: Grey infrastructure
HSSF: Horizontal sub-surface flow
LCCA: Life-cycle cost assessment
MGD: Million gallons per day
O&M Costs: Operation and Maintenance costs
SSF: Sub-surface flow
TSS: Total Suspended Solids
VSSF: Vertical sub-surface flow
USD: United States Dollars
US EPA: United States Environmental Protection Agency

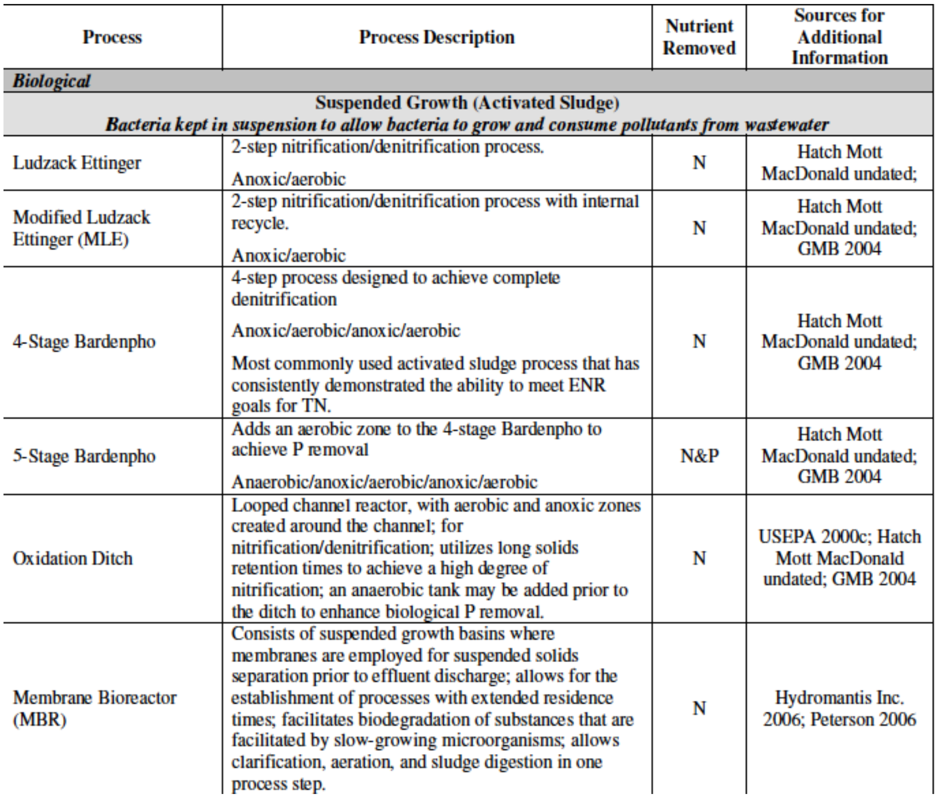
Appendix B: Primary Biochemical Processes for Nutrient Removal in CW

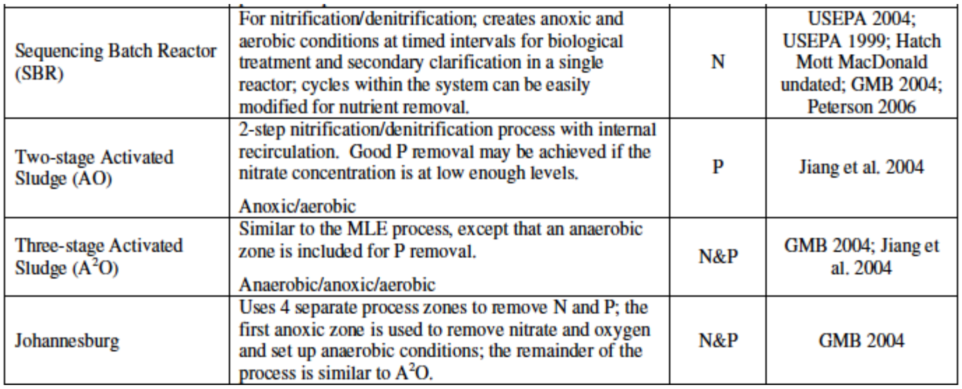
This table provides a brief overview of the primary biochemical processes that occur within CW to remove nitrogen and phosphorus from wastewater effluent. The "Process" column identifies the name of the biochemical process, while the "Chemical Equation/Description" column describes the chemical reaction that occurs within the process. For processes that utilize bacteria to breakdown the nutrients, the "Environmental Conditions" column shows whether the process occurs in aerobic (with oxygen) conditions or in anaerobic conditions (without oxygen). For some of the processes, the environmental conditions are not applicable (N/A). Lastly, the "Nutrient Removed" column states whether the biochemical process removes nitrogen (N) or phosphorus (P) from the wastewater effluent.

Process	**Chemical Equation/Description**	**Environmental Conditions**	**Nutrient Removed**
Nitrification	ammonia-N→nitrite-N→ nitrate-N	Aerobic	N
Denitrification	nitrate-N→nitrite-N→ gaseous N2, N2O	Anaerobic	N
Volatization	ammonia-N (aq)→ammonia-N (g)	N/A	N
Adsorption	P binds to Al, Fe, or Ca ions in the gravel media, causing the P ions to settle within the media	N/A	P
Soil Accretion	P ions settle and bind with ions located within the CW's soil	N/A	P
Plant Uptake	The wetland's plants uptake phosphorus through its roots as P is a necessary nutrient for plant growth. Peak P uptake usually occurs early in a plant's growing season.	N/A	P

Appendix C: Grey Infrastructure for Nutrient Reduction (Hartman and Cleland 2007)

This table provides more in-depth information on current GI technologies for wastewater nutrient reduction. The technologies are grouped together according to the method in which they remove nutrients: biologically, physically, and chemically. The "Process" column identifies the common name for the technologies. "Process Description" provides a brief description into the mechanism in which nutrient reduction occurs. Lastly, the "Nutrient Removed" column identifies whether the process is used to remove nitrogen (N), phosphorus (P), or both nutrients (N&P).





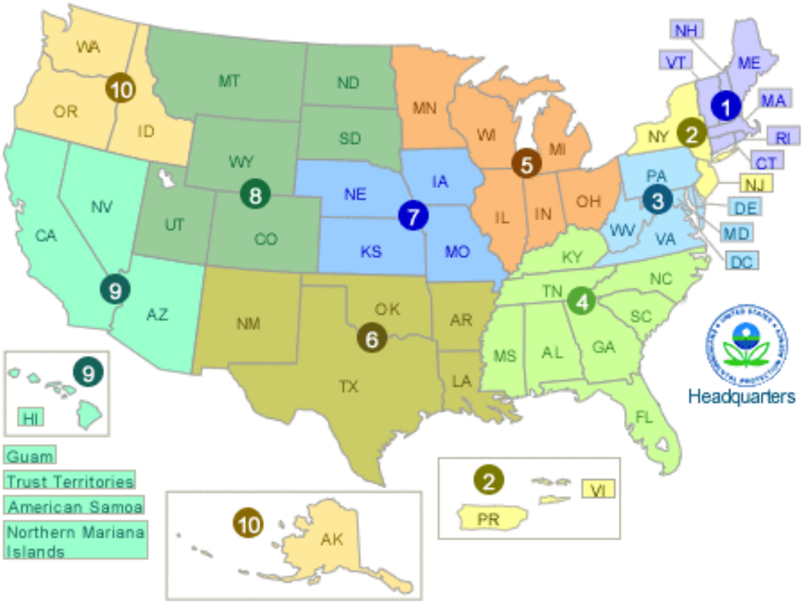
Process	Process Description	Nutrient Removed	Sources for Additional Information
Biological			
	Suspended Growth (Activated Sludge) ***Bacteria kept in suspension to allow bacteria to grow and consume pollutants from wastewater***		
Ludzack Ettinger	2-step nitrification/denitrification process. Anoxic/aerobic	N	Hatch Mott MacDonald undated;
Modified Ludzack Ettinger (MLE)	2-step nitrification/denitrification process with internal recycle. Anoxic/aerobic	N	Hatch Mott MacDonald undated; GMB 2004
4-Stage Bardenpho	4-step process designed to achieve complete denitrification Anoxic/aerobic/anoxic/aerobic Most commonly used activated sludge process that has consistently demonstrated the ability to meet ENR goals for TN.	N	Hatch Mott MacDonald undated; GMB 2004
5-Stage Bardenpho	Adds an aerobic zone to the 4-stage Bardenpho to achieve P removal Anaerobic/anoxic/aerobic/anoxic/aerobic	N&P	Hatch Mott MacDonald undated; GMB 2004
Oxidation Ditch	Looped channel reactor, with aerobic and anoxic zones created around the channel; for nitrification/denitrification; utilizes long solids retention times to achieve a high degree of nitrification; an anaerobic tank may be added prior to the ditch to enhance biological P removal.	N	USEPA 2000c; Hatch Mott MacDonald undated; GMB 2004
Membrane Bioreactor (MBR)	Consists of suspended growth basins where membranes are employed for suspended solids separation prior to effluent discharge; allows for the establishment of processes with extended residence times; facilitates biodegradation of substances that are facilitated by slow-growing microorganisms; allows clarification, aeration, and sludge digestion in one process step.	N	Hydromantis Inc. 2006; Peterson 2006
Sequencing Batch Reactor (SBR)	For nitrification/denitrification; creates anoxic and aerobic conditions at timed intervals for biological treatment and secondary clarification in a single reactor; cycles within the system can be easily modified for nutrient removal.	N	USEPA 2004; USEPA 1999; Hatch Mott MacDonald undated; GMB 2004; Peterson 2006
Two-stage Activated Sludge (AO)	2-step nitrification/denitrification process with internal recirculation. Good P removal may be achieved if the nitrate concentration is at low enough levels. Anoxic/aerobic	P	Jiang et al. 2004
Three-stage Activated Sludge (A^2O)	Similar to the MLE process, except that an anaerobic zone is included for P removal. Anaerobic/anoxic/aerobic	N&P	GMB 2004; Jiang et al. 2004
Johannesburg	Uses 4 separate process zones to remove N and P; the first anoxic zone is used to remove nitrate and oxygen and set up anaerobic conditions; the remainder of the process is similar to A^2O.	N&P	GMB 2004

Attached Growth (Fixed Film) ***Utilizes media to provide a surface for biomass to grow and perform nitrification & denitrification***			
Trickling Filter	For nitrification/denitrification; involves a tank, usually filled with a bed of rocks, stones or synthetic media, to support bacterial growth used to treat wastewater.	N	USEPA 2000d; USEPA 2004; USACE 2001
Rotating Biological Contactor	For nitrification/denitrification; series of disks attached to a central axis that rotates and exposes biomass on disks to both air (aerobic conditions) and wastewater (anoxic conditions).	N	GMB 2004; USACE 2001; Peterson 2006
Denitrification Filter	Utilize granular media to remove nitrates after nitrification. May be added to existing treatment systems that use biological processes to convert nitrate-N to N gas; physical/chemical treatment may be added using chemical phosphorus precipitation to achieve TP as low as 0.3 mg/L.	N	Hatch Mott MacDonald undated; Freed 2007
Fluidized Bed Reactor	For nitrification/denitrification; utilizes small media which is kept in suspension by aeration or mixing action; aerobic and anaerobic reactors may be arranged in series for complete BNR OR may be added to an existing BNR process for additional denitrification.	N	GMB 2004
Physical			
Reverse Osmosis	Performs micro-filtration processes using membrane; will remove nearly all particulate P and 95-99% soluble P; membrane treatments may be expensive.	P	USACE 2001
Sand Filtration	Traps some solids remaining in the wastewater after secondary treatment. Can reduce P adsorbed to solid particles, such as particles remaining in suspension after chemical addition (see below). In some cases, biological nitrification/denitrification is also achieved with sand filters. Performance may be adversely affected by cold weather.	P	Hydromantis Inc. 2006
Chemical			
Chemical Addition	Also commonly referred to as chemical precipitation or flocculation. Metal salts (e.g. alum, iron) or other chemicals may be used as coagulants and precipitating agents to enhance the formation and separation of solids that can removed from wastewater stream. Chemicals are added to either a conventional secondary treatment process (e.g., activated sludge) or may be incorporated as part of a tertiary treatment technology (e.g., sand filtration); the effectiveness of the agent will be a function of the solids separation efficiency and its effectiveness in forming a solid-associated contaminant.	P	USEPA 2000e; Hydromantis Inc. 2006

Appendix D: Mean Value Table for Dependent and Independent Variables

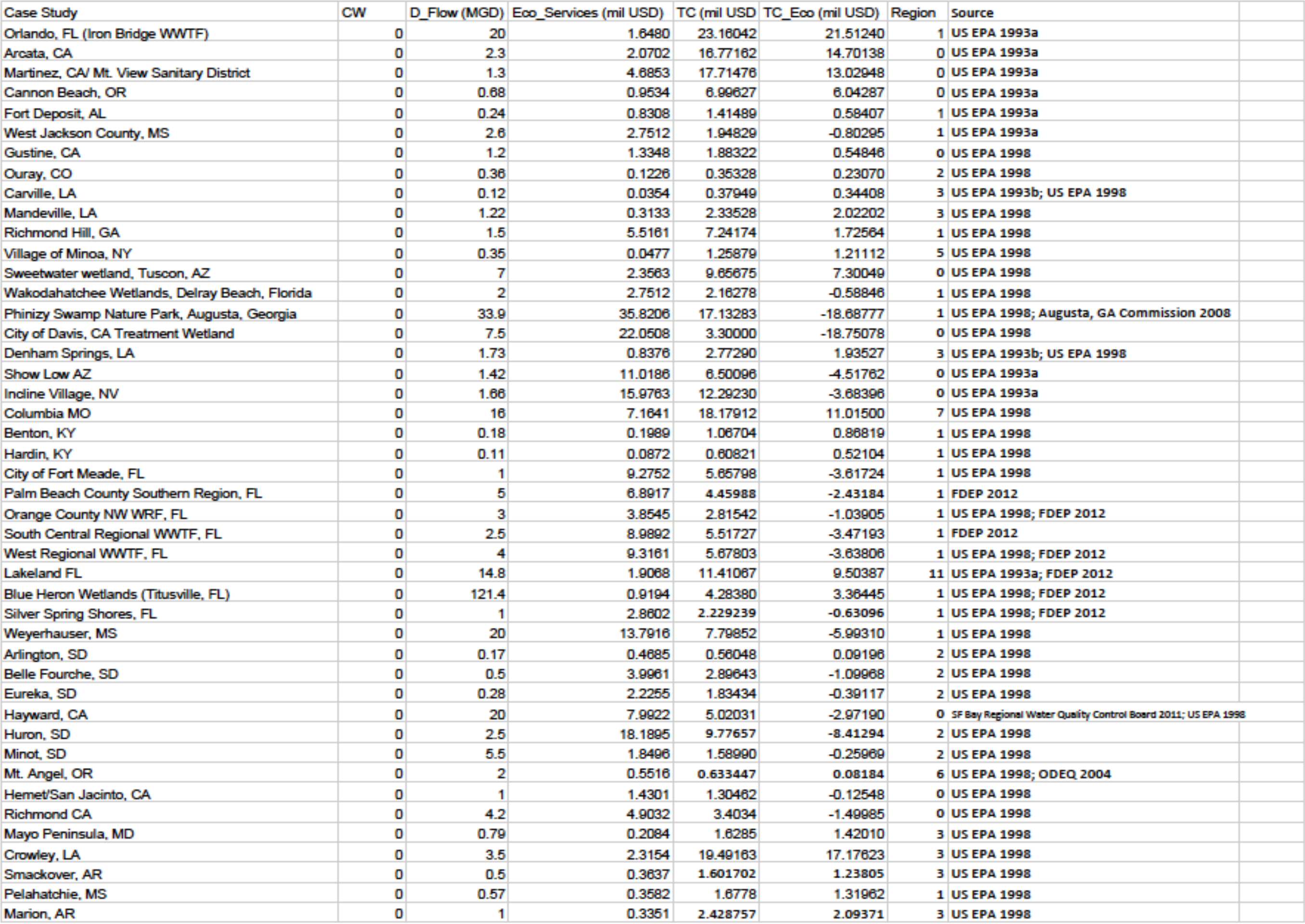
	Grey Infrastructure			Constructed Wetlands			
	# of Obs.	**TC (mil USD)**	**D_Flow (MGD)**	**# of Obs.**	**TC (mil USD)**	**Eco_Services (mil USD)**	**D_Flow (MGD)**
EPA Region 9	2	2	9.9	10	7.78	7.38	4.76
EPA Region 4	0	n/a	n/a	18	5.90	5.99	12.99
EPA Region 8	1	0.10	0.5	6	2.84	4.48	1.55
EPA Region 6	2	0.37	1.83	15	2.75	0.40	0.92
EPA Region 3	64	1.12	5.54	0	n/a	n/a	n/a
EPA Region 2	0	n/a	n/a	1	1.26	0.05	0.35
EPA Region 10	0	n/a	n/a	2	3.82	0.75	1.34
EPA Region 7	0	n/a	n/a	1	18.18	7.16	16
EPA Region 1	1	0.12	0.6	0	n/a	n/a	n/a

Appendix E: EPA Regions Map (US EPA 2015)

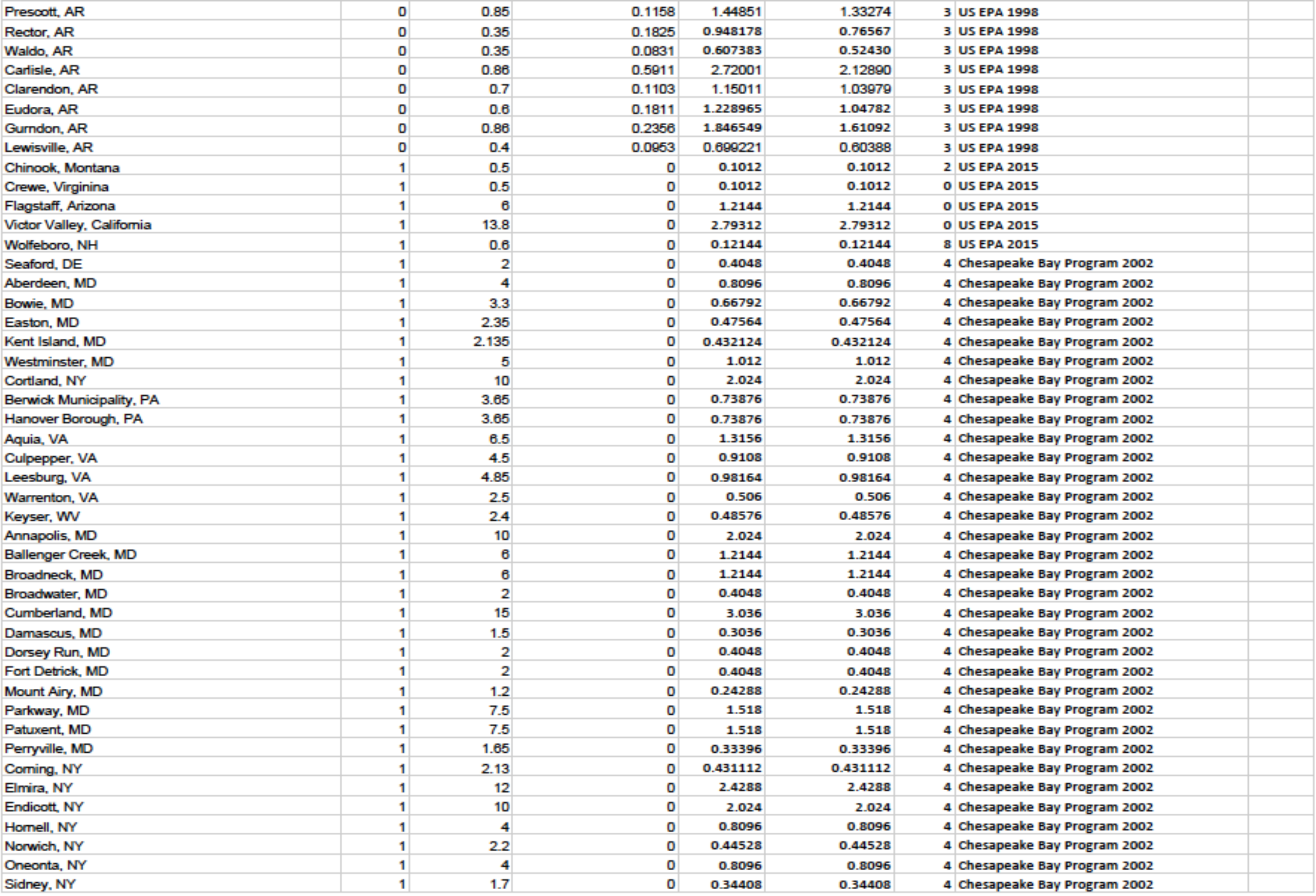


Appendix F: List of Case Studies and Values

The following table provides an overview of the case studies and values used in this study. The "Case Study" column provides the wastewater treatment that the CW or GI improvements were built for. The "CW" column identifies whether the case study is a CW (0) or a GI improvement (1). "D_Flow" provides the values for the million gallons per day (MGD) daily flow of wastewater into the systems. The "Eco_Services" column lists the value for the ecosystem services generated by the CW systems in million USD. "TC" are the total cost values for the systems, and "TC_Eco" are the total cost values for the systems when the "Eco_Services" values are included. The "Region" column identifies the US EPA Region the system is located in using the variable values identified in the study's Methodology section. Lastly, the sources that provided the values used in this study are presented in the "Source" column.



Case Study	CW	D_Flow (MGD)	Eco_Services (mil USD)	TC (mil USD	TC_Eco (mil USD)	Region	Source
Orlando, FL (Iron Bridge WWTF)	0	20	1.6480	23.16042	21.51240	1	US EPA 1993a
Arcata, CA	0	2.3	2.0702	16.77162	14.70138	0	US EPA 1993a
Martinez, CA/ Mt. View Sanitary District	0	1.3	4.6853	17.71476	13.02948	0	US EPA 1993a
Cannon Beach, OR	0	0.68	0.9534	6.99627	6.04287	0	US EPA 1993a
Fort Deposit, AL	0	0.24	0.8308	1.41489	0.58407	1	US EPA 1993a
West Jackson County, MS	0	2.6	2.7512	1.94829	-0.80295	1	US EPA 1993a
Gustine, CA	0	1.2	1.3348	1.88322	0.54846	0	US EPA 1998
Ouray, CO	0	0.36	0.1226	0.35328	0.23070	2	US EPA 1998
Carville, LA	0	0.12	0.0354	0.37949	0.34408	3	US EPA 1993b; US EPA 1998
Mandeville, LA	0	1.22	0.3133	2.33528	2.02202	3	US EPA 1998
Richmond Hill, GA	0	1.5	5.5161	7.24174	1.72564	1	US EPA 1998
Village of Minoa, NY	0	0.35	0.0477	1.25879	1.21112	5	US EPA 1998
Sweetwater wetland, Tuscon, AZ	0	7	2.3563	9.65675	7.30049	0	US EPA 1998
Wakodahatchee Wetlands, Delray Beach, Florida	0	2	2.7512	2.16278	-0.58846	1	US EPA 1998
Phinizy Swamp Nature Park, Augusta, Georgia	0	33.9	35.8206	17.13283	-18.68777	1	US EPA 1998; Augusta, GA Commission 2008
City of Davis, CA Treatment Wetland	0	7.5	22.0508	3.30000	-18.75078	0	US EPA 1998
Denham Springs, LA	0	1.73	0.8376	2.77290	1.93527	3	US EPA 1993b; US EPA 1998
Show Low AZ	0	1.42	11.0186	6.50096	-4.51762	0	US EPA 1993a
Incline Village, NV	0	1.66	15.9763	12.29230	-3.68396	0	US EPA 1993a
Columbia MO	0	16	7.1641	18.17912	11.01500	7	US EPA 1998
Benton, KY	0	0.18	0.1989	1.06704	0.86819	1	US EPA 1998
Hardin, KY	0	0.11	0.0872	0.60821	0.52104	1	US EPA 1998
City of Fort Meade, FL	0	1	9.2752	5.65798	-3.61724	1	US EPA 1998
Palm Beach County Southern Region, FL	0	5	6.8917	4.45988	-2.43184	1	FDEP 2012
Orange County NW WRF, FL	0	3	3.8545	2.81542	-1.03905	1	US EPA 1998; FDEP 2012
South Central Regional WWTF, FL	0	2.5	8.9892	5.51727	-3.47193	1	FDEP 2012
West Regional WWTF, FL	0	4	9.3161	5.67803	-3.63806	1	US EPA 1998; FDEP 2012
Lakeland FL	0	14.8	1.9068	11.41067	9.50387	11	US EPA 1993a; FDEP 2012
Blue Heron Wetlands (Titusville, FL)	0	121.4	0.9194	4.28380	3.36445	1	US EPA 1998; FDEP 2012
Silver Spring Shores, FL	0	1	2.8602	2.229239	-0.63096	1	US EPA 1998; FDEP 2012
Weyerhauser, MS	0	20	13.7916	7.79852	-5.99310	1	US EPA 1998
Arlington, SD	0	0.17	0.4685	0.56048	0.09196	2	US EPA 1998
Belle Fourche, SD	0	0.5	3.9961	2.89643	-1.09968	2	US EPA 1998
Eureka, SD	0	0.28	2.2255	1.83434	-0.39117	2	US EPA 1998
Hayward, CA	0	20	7.9922	5.02031	-2.97190	0	SF Bay Regional Water Quality Control Board 2011; US EPA 1998
Huron, SD	0	2.5	18.1895	9.77657	-8.41294	2	US EPA 1998
Minot, SD	0	5.5	1.8496	1.58990	-0.25969	2	US EPA 1998
Mt. Angel, OR	0	2	0.5516	0.633447	0.08184	6	US EPA 1998; ODEQ 2004
Hemet/San Jacinto, CA	0	1	1.4301	1.30462	-0.12548	0	US EPA 1998
Richmond CA	0	4.2	4.9032	3.4034	-1.49985	0	US EPA 1998
Mayo Peninsula, MD	0	0.79	0.2084	1.6285	1.42010	3	US EPA 1998
Crowley, LA	0	3.5	2.3154	19.49163	17.17623	3	US EPA 1998
Smackover, AR	0	0.5	0.3637	1.601702	1.23805	3	US EPA 1998
Pelahatchie, MS	0	0.57	0.3582	1.6778	1.31962	1	US EPA 1998
Marion, AR	0	1	0.3351	2.428757	2.09371	3	US EPA 1998



Prescott, AR	0	0.85	0.1158	1.44851	1.33274	3	US EPA 1998	
Rector, AR	0	0.35	0.1825	0.948178	0.76567	3	US EPA 1998	
Waldo, AR	0	0.35	0.0831	0.607383	0.52430	3	US EPA 1998	
Carlisle, AR	0	0.86	0.5911	2.72001	2.12890	3	US EPA 1998	
Clarendon, AR	0	0.7	0.1103	1.15011	1.03979	3	US EPA 1998	
Eudora, AR	0	0.6	0.1811	1.228965	1.04782	3	US EPA 1998	
Gurndon, AR	0	0.86	0.2356	1.846549	1.61092	3	US EPA 1998	
Lewisville, AR	0	0.4	0.0953	0.699221	0.60388	3	US EPA 1998	
Chinook, Montana	1	0.5	0	0.1012	0.1012	2	US EPA 2015	
Crewe, Virginina	1	0.5	0	0.1012	0.1012	0	US EPA 2015	
Flagstaff, Arizona	1	6	0	1.2144	1.2144	0	US EPA 2015	
Victor Valley, California	1	13.8	0	2.79312	2.79312	0	US EPA 2015	
Wolfeboro, NH	1	0.6	0	0.12144	0.12144	8	US EPA 2015	
Seaford, DE	1	2	0	0.4048	0.4048	4	Chesapeake Bay Program 2002	
Aberdeen, MD	1	4	0	0.8096	0.8096	4	Chesapeake Bay Program 2002	
Bowie, MD	1	3.3	0	0.66792	0.66792	4	Chesapeake Bay Program 2002	
Easton, MD	1	2.35	0	0.47564	0.47564	4	Chesapeake Bay Program 2002	
Kent Island, MD	1	2.135	0	0.432124	0.432124	4	Chesapeake Bay Program 2002	
Westminster, MD	1	5	0	1.012	1.012	4	Chesapeake Bay Program 2002	
Cortland, NY	1	10	0	2.024	2.024	4	Chesapeake Bay Program 2002	
Berwick Municipality, PA	1	3.65	0	0.73876	0.73876	4	Chesapeake Bay Program 2002	
Hanover Borough, PA	1	3.65	0	0.73876	0.73876	4	Chesapeake Bay Program 2002	
Aquia, VA	1	6.5	0	1.3156	1.3156	4	Chesapeake Bay Program 2002	
Culpepper, VA	1	4.5	0	0.9108	0.9108	4	Chesapeake Bay Program 2002	
Leesburg, VA	1	4.85	0	0.98164	0.98164	4	Chesapeake Bay Program 2002	
Warrenton, VA	1	2.5	0	0.506	0.506	4	Chesapeake Bay Program 2002	
Keyser, WV	1	2.4	0	0.48576	0.48576	4	Chesapeake Bay Program 2002	
Annapolis, MD	1	10	0	2.024	2.024	4	Chesapeake Bay Program 2002	
Ballenger Creek, MD	1	6	0	1.2144	1.2144	4	Chesapeake Bay Program 2002	
Broadneck, MD	1	6	0	1.2144	1.2144	4	Chesapeake Bay Program 2002	
Broadwater, MD	1	2	0	0.4048	0.4048	4	Chesapeake Bay Program 2002	
Cumberland, MD	1	15	0	3.036	3.036	4	Chesapeake Bay Program 2002	
Damascus, MD	1	1.5	0	0.3036	0.3036	4	Chesapeake Bay Program 2002	
Dorsey Run, MD	1	2	0	0.4048	0.4048	4	Chesapeake Bay Program 2002	
Fort Detrick, MD	1	2	0	0.4048	0.4048	4	Chesapeake Bay Program 2002	
Mount Airy, MD	1	1.2	0	0.24288	0.24288	4	Chesapeake Bay Program 2002	
Parkway, MD	1	7.5	0	1.518	1.518	4	Chesapeake Bay Program 2002	
Patuxent, MD	1	7.5	0	1.518	1.518	4	Chesapeake Bay Program 2002	
Perryville, MD	1	1.65	0	0.33396	0.33396	4	Chesapeake Bay Program 2002	
Corning, NY	1	2.13	0	0.431112	0.431112	4	Chesapeake Bay Program 2002	
Elmira, NY	1	12	0	2.4288	2.4288	4	Chesapeake Bay Program 2002	
Endicott, NY	1	10	0	2.024	2.024	4	Chesapeake Bay Program 2002	
Hornell, NY	1	4	0	0.8096	0.8096	4	Chesapeake Bay Program 2002	
Norwich, NY	1	2.2	0	0.44528	0.44528	4	Chesapeake Bay Program 2002	
Oneonta, NY	1	4	0	0.8096	0.8096	4	Chesapeake Bay Program 2002	
Sidney, NY	1	1.7	0	0.34408	0.34408	4	Chesapeake Bay Program 2002	

Ashland Municipality, PA	1	1.3	0	0.26312	0.26312	4	Chesapeake Bay Program 2002	
Bellefonte Borough, PA	1	3.22	0	0.651728	0.651728	4	Chesapeake Bay Program 2002	
Bloomsburg Municipality, PA	1	4.29	0	0.868296	0.868296	4	Chesapeake Bay Program 2002	
Carlisle Borough, PA	1	7	0	1.4168	1.4168	4	Chesapeake Bay Program 2002	
Clearfield, PA	1	4.5	0	0.9108	0.9108	4	Chesapeake Bay Program 2002	
Columbia, PA	1	2	0	0.4048	0.4048	4	Chesapeake Bay Program 2002	
Derry Township, PA	1	5	0	1.012	1.012	4	Chesapeake Bay Program 2002	
Dover Township, PA	1	4	0	0.8096	0.8096	4	Chesapeake Bay Program 2002	
Greater Hazelton, PA	1	8.9	0	1.80136	1.80136	4	Chesapeake Bay Program 2002	
Huntingdon Borough, PA	1	3.75	0	0.759	0.759	4	Chesapeake Bay Program 2002	
Lebanon City, PA	1	8	0	1.6192	1.6192	4	Chesapeake Bay Program 2002	
Lower Allen Township, PA	1	5.95	0	1.20428	1.20428	4	Chesapeake Bay Program 2002	
Mechanicsburg, PA	1	2.08	0	0.420992	0.420992	4	Chesapeake Bay Program 2002	
Middletown, PA	1	2.2	0	0.44528	0.44528	4	Chesapeake Bay Program 2002	
Penn Township, PA	1	4.2	0	0.85008	0.85008	4	Chesapeake Bay Program 2002	
Sayre, PA	1	1.94	0	0.392656	0.392656	4	Chesapeake Bay Program 2002	
Scranton, PA	1	28	0	5.6672	5.6672	4	Chesapeake Bay Program 2002	
Shamokin-Coal Township, PA	1	7	0	1.4168	1.4168	4	Chesapeake Bay Program 2002	
Shippensburg Borough, PA	1	2.75	0	0.5566	0.5566	4	Chesapeake Bay Program 2002	
Wellsboro Municipality, PA	1	2	0	0.4048	0.4048	4	Chesapeake Bay Program 2002	
York City, PA	1	26	0	5.2624	5.2624	4	Chesapeake Bay Program 2002	
Buena Vista, VA	1	2.25	0	0.4554	0.4554	4	Chesapeake Bay Program 2002	
Clifton Forge, VA	1	2	0	0.4048	0.4048	4	Chesapeake Bay Program 2002	
Covington, VA	1	3	0	0.6072	0.6072	4	Chesapeake Bay Program 2002	
Dale City #1, VA	1	4	0	0.8096	0.8096	4	Chesapeake Bay Program 2002	
Dale City #8,VA	1	4	0	0.8096	0.8096	4	Chesapeake Bay Program 2002	
Farmville, VA	1	2.4	0	0.48576	0.48576	4	Chesapeake Bay Program 2002	
Fisherville, Va	1	2	0	0.4048	0.4048	4	Chesapeake Bay Program 2002	
Fredericksburg, Va	1	3.5	0	0.7084	0.7084	4	Chesapeake Bay Program 2002	
Front Royal, VA	1	4	0	0.8096	0.8096	4	Chesapeake Bay Program 2002	
Harrisonburg, Va	1	16	0	3.2384	3.2384	4	Chesapeake Bay Program 2002	
Lexington, VA	1	4	0	0.8096	0.8096	4	Chesapeake Bay Program 2002	
Little Falls Run, VA	1	4	0	0.8096	0.8096	4	Chesapeake Bay Program 2002	
Lynchburg, VA	1	22	0	4.4528	4.4528	4	Chesapeake Bay Program 2002	
Middle River, VA	1	6.8	0	1.37632	1.37632	4	Chesapeake Bay Program 2002	
Proctors Creek, VA	1	21.5	0	4.3516	4.3516	4	Chesapeake Bay Program 2002	
Quantico, Va	1	2.2	0	0.44528	0.44528	4	Chesapeake Bay Program 2002	
Totopotomoy, Va	1	5	0	1.012	1.012	4	Chesapeake Bay Program 2002	
Waynesboro, VA	1	4	0	0.8096	0.8096	4	Chesapeake Bay Program 2002	
Berkely County, WV	1	2.35	0	0.47564	0.47564	4	Chesapeake Bay Program 2002	
Martinsburg, WV	1	5	0	1.012	1.012	4	Chesapeake Bay Program 2002	
Stephenville, TX	1	3	0	0.6072	0.6072	3	Hartman and Cleland 2007	
Clifton, TX	1	0.65	0	0.13156	0.13156	3	Hartman and Cleland 2007	

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