Bioretention Facilities and Constructed Wetlands Efficiency and Use: A Review

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Abstract

Population growth and human-induced landscape manipulation has degraded water quality around the world. This is due in large part to the replacement of natural wetlands and associated pollution attenuation benefits with impervious surfaces and pollutant-producing development. Natural mechanisms and engineering technologies have been combined to form best management practices that aim to improve water quality. Bioretention and constructed wetlands are common facilities that show promise in improving water quality in urban, suburban, industrial, municipal, and agricultural environments. These facilities can be used separately or together to achieve maximal nutrient removal efficiencies. Research has shown high variability in the efficiency of these treatment systems. This review aims to present recent research on the type and removal processes associated with bioretention facilities and constructed wetlands and to aid in improving their design. Nitrogen and phosphorus are the most targeted pollutants in studies regarding bioretention and constructed wetlands due to their ubiquitous occurrence in effluent and their effects on eutrophication. Removal of these nutrients is dependent on treatment system design that considers hydrology, vegetation, and substrate. Hydrology in regards to hydraulic load rate, retention time, and aerobic/anaerobic status is the main force behind the operation of these systems. Selection of appropriate vegetation and substrate is a critical design consideration for nutrient assimilation and sequestration. Maintenance and management of treatment systems are often required and may increase costs and direct design choices. The

literature reviewed for this paper exhibits conflicting information regarding best methodology for nutrient removal within constructed wetlands and bioretention facilities; however, current research can be employed to create successful systems within reasonable expectations. Further studies are recommended to understand the mechanisms responsible for optimum removal efficiencies and development of constants to predict outcomes than can better tune design.

Introduction

Anthropogenic manipulation of the landscape has led to degraded water quality in many parts of the world (Davis, 2008). Manipulation such as development, agricultural practices, and wastewater effluent negatively alter and replace the treatment capabilities of the natural environment. These alterations tend to result in elevated pollutant loading, reduced infiltration and treatment, sedimentation, and voluminous peak flows that natural systems cannot completely absorb (Davis, 2008; Passeport et al., 2009). As populations continue to expand, natural filtering mechanisms become stressed. This effect is compounded by the documented loss of natural wetland systems, which is estimated to be approximately 50 or greater percent worldwide (Mitsch and Gosselink, 2000). The loss of naturally existing filtering mechanisms promoted by wetlands has led to the loading of excess nutrients and pollutants into water bodies causing water quality degradation.

In an effort to improve water quality and offset loss of natural wetland functions, a hybridization of natural capacities with engineering technologies has become a popular best management practice (BMP) to treat polluted discharges and attenuate hydrology changes (Hunt et al., 2006; James and Dymond, 2011). The two most common treatment systems are bioretention facilities and constructed wetlands. Bioretention is typically considered a “dry” system where water residence time is limited, while constructed wetlands retain water for longer periods. Bioretention and constructed wetland systems have been shown to effectively remove pollutants from point and non-point sources (Fink and Mitsch, 2004; Stone et al., 2004; Cui et al., 2008). These systems differ in form and function to improve water quality through similar natural mechanisms.

Bioretention and constructed wetlands are gradually becoming integrated within modified landscapes as components of new development or add-on systems to existing infrastructure (Tanner, 1996). The design and use of bioretention and constructed wetlands is being considered a prevalent method to treat pollutants in urban, suburban, and agricultural environments (Brooks et al., 2000; Hunt et al., 2006; Lieyu Zhang et al., 2011; Song et al., 2010). Although the practical application and scientific research regarding treatment system design is still developing, regulatory agencies are progressively requiring their use (Hunt et al., 2006; Passeport et al., 2009; White et al., 2011). For example, the United States Environmental Protection Agency (EPA) has been working to set total maximum daily loads of pollutants into waterways to curb concentration increases (White et al., 2011). These limits trigger the use of BMPs to treat runoff as a means to meet EPA guidelines. Additionally, there is a trend towards the use of sustainable methods of water treatment, such as low-impact development (LID), that reduce costs and resource consumption (Hunt et al., 2006; Wong, 2006). Implementation of bioretention and constructed wetlands affords the opportunity to utilize natural processes to purify water in a sustainable, cost-effective, conservation-oriented manner (D. Zhang et al., 2009).

Constructed water treatment facilities have been shown to be effective in the removal of excessive nutrient pollutants and harmful bacteria (Song et al., 2010; White et al., 2011; Lan Zhang et al., 2011). The removal efficiencies of bioretention and constructed wetlands are highly variable and considerable research has focused on determining their true effectiveness. Removal efficiencies of common nutrients such as nitrogen (N) and phosphorus (P) and fecal coliform and *Escherichia coli* bacteria strains have been reported to be negligible by some, while other research has elucidated pollutant abatement in excess of 80% (Lan Zhang et al., 2011; Passeport et al, 2009). As implementation of treatment systems becomes commonplace and regulations are set to achieve a determined removal efficiency target, it is critical that complex processes controlling removal are identified and design is conducted sufficiently to apply them.

Although the use of treatment systems is promising, the application of these facilities for purposes of water quality improvement is not without complications. Dynamic processes within wetlands related to temporal and spatial variation inhibit the prediction of nutrient flux and resultant water quality improvement (Thorén et al., 2004). It has been reported that constructed systems are not likely to mimic ideal natural conditions from the outset (Campbell et al., 2002; Fennessy et al., 2008; Reeder, 2011). With this in mind, it may be difficult to assign performance standards to a constructed system when reference conditions are not achievable. Short-term results should not be expected to indicate success, only an indication of a trend towards the intended goals.

The optimum design characteristics of these systems is not fully understood (Fink and Mitsch, 2004). Many factors are at work in the removal of pollutants within a constructed wetland and selection of design details need to coincide with as many as possible. Proper design is dependent on detailed holistic data collection, which is paramount to the implementation of a treatment system that will achieve desired goals (Campbell et al., 2002; Reeder, 2011; Lieyu Zhang et al., 2011). Many constructed systems fail due to lack of understanding of biogeochemical processes, overall insufficient research regarding pollutant removal, and a dependency on statistical models (Lieyu Zhang et al., 2011). Although constructed wetlands and bioretention serve beneficial functions, advancements in research and design techniques are not far enough along for constructed facilities to completely replace or sufficiently emulate natural wetland functions. It may be advisable to consider constructed systems as a “jump-start” or attendant feature to a fully functional system and probably shouldn’t be relied upon for complete nutrient removal.

The purpose of this paper is to conduct a literature review regarding bioretention facilities and constructed wetlands, including the types and processes that govern removal efficiencies. The information provided herein is intended to assist parties interested in utilizing constructed systems for water quality improvement. The review spans the features, benefits, and potential problems discovered by recent research. Hopefully this information yields improved decisions concerning constructed treatment system design.

Treatment System Characteristics

Bioretention areas and constructed wetlands are being used and researched in numerous configurations to better understand what design attributes achieve the best use efficiency. Although the chosen design form depends heavily on impetus of use, there are generalized characteristics that are attributed to them. The functions realized by these systems are not mutually exclusive and the combination of bioretention and constructed wetlands is a beneficial approach.

Bioretention Facilities

Bioretention facilities are commonly referred to as raingardens and bioswales and are most often used as initial runoff treatment systems and typically only detain runoff for a short period of time. Therefore, these “dry” systems contribute to pollutant removal during short pulses associated with precipitation. Bioretention has been found to be best suited for treatment of minor, less intense storm events (James and Dymond, 2011). Although this may be considered a limitation, it makes these systems suitable for use in confined and numerous areas where other alternatives are not feasible.

Typically these systems consist of a soil filter media that is characterized by a high hydraulic conductivity, which is responsible for rapid infiltration of stormwater and reduced residence time. During percolation, pollutants are removed from the water by the filtration media. Since bioretention systems rely on hydraulic conductivity to draw down pollutants through filter material, restricted drainage would result in increased surface runoff and lack of pollutant removal (James and Dymond, 2011). Oftentimes, a drain system is constructed within or below the filter media to ensure aerobic conditions persist (Figure 1).The use of a drainage system is particularly necessary in locations where the facility is underlain by a less pervious soil and risk of perching exists or where the water table may encroach within the filter profile (James and Dymond, 2011). A drainage system is also required where an impermeable liner intended to inhibit translocation of pollutants to groundwater is incorporated between the filter media and native soil.

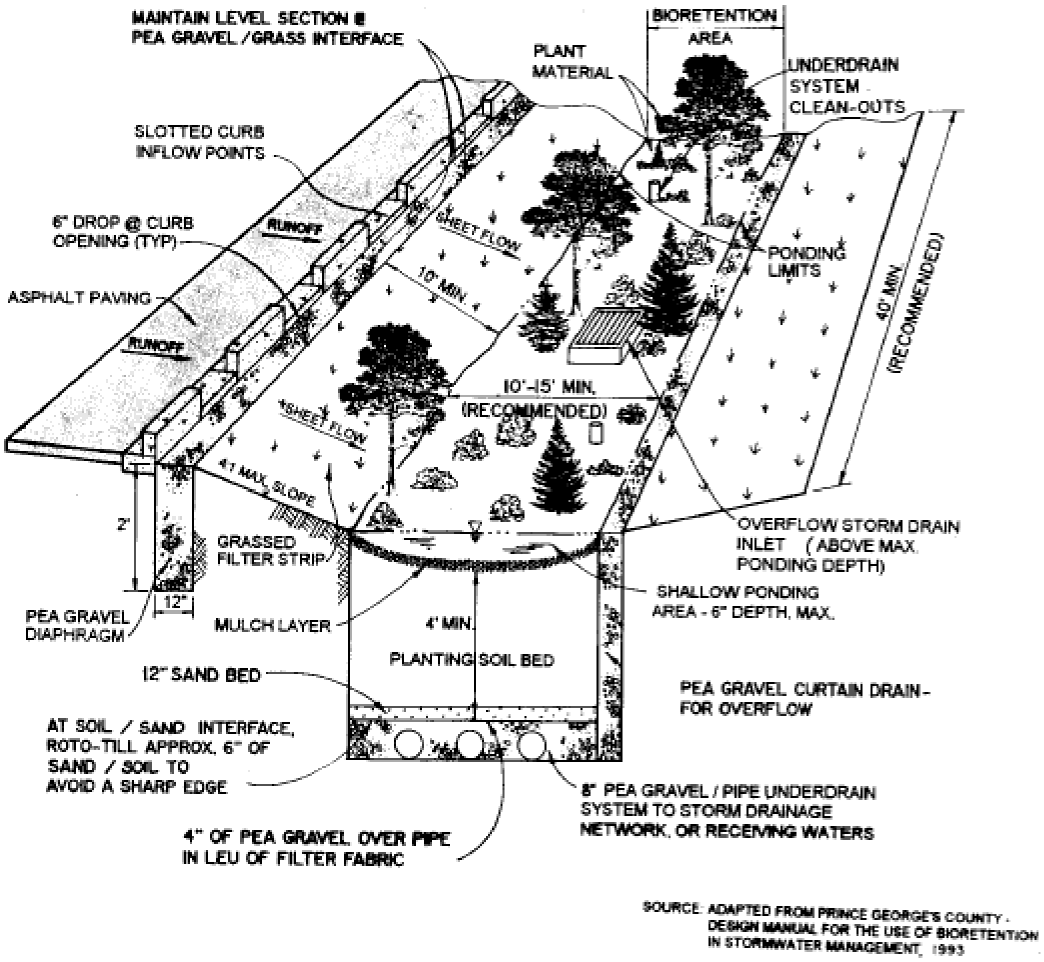

Figure 1: Typical bioretention facility to treat stormwater runoff (Claytor and Schueler, 1996)

In addition to soil media and drainage, plants and mulch are typical features implemented for both aesthetic and nutrient removal benefits. The plants are efficient at nutrient assimilation and mulch is a beneficial energy source for microbial respiration and water retention. Added benefits of mulch include moisture retention for plant and microbial functions, contributions to soil organic matter (OM) content, and aesthetics. Use of bioretention is especially useful in areas that are heavily traveled and aesthetics are important (Hunt et al., 2006). They allow otherwise static, hard-surfaced areas to become both useful and attractive. Due to the aesthetic value and pollutant removal properties, these bioretention facilities are often selected for use in areas focusing on LID, where a "softer" approach is desired (Hunt et al., 2006). A photograph of an attractive a functional bioretention system treating shopping mall parking lot runoff in Maryland is provided as Figure 2.

Figure 2: Bioretention facility treating parking lot runoff in Maryland (Davis, 2004)

Because bioretention is effective as initial runoff treatment, these systems can be effectively used as pre- and post-treatment of runoff and effluent prior to and after constructed wetland treatment systems (Stone et al., 2004). Pre- and post-treatment plays an important role in affecting nutrient loading into wetlands and the receiving wetland's removal efficiency. Since the dynamics related to precipitation and temperature directly influence the rate, species, and concentration of nutrients into and exiting wetland systems, pre- and post-treatment can have a distinct effect on transformation and fate prior to reaching the wetland proper or receiving waters (Thorén et al., 2003). Combining bioretention as pre- and/or post-treatment with constructed wetlands allows for dedicated aerobic and anaerobic benefits (Ballantine and Tanner, 2010). However, bioretention areas have also been designed to incorporate anaerobic zones to take advantage of fate pathways where they are not combined with constructed wetlands (Hunt et al., 2006).

Constructed Wetlands

In contrast to bioretention systems that are mainly used to treat moderate stormwater runoff, constructed wetlands are utilized to treat runoff and effluent composed of high pollutant concentrations in larger scales. These systems rely on longer hydrologic residence time to capitalize on anaerobic effects. Constructed wetlands have been used to treat runoff and wastewater effluent from agricultural, industrial, residential (e.g, urban, suburban), commercial, and municipal sources (Brooks et al., 2000; Davis et al., 2003; Kohler et al., 2004; Mbuligwe, 2004; Thorén et al., 2004; Hunt et al., 2006; D. Zhang et al., 2009; Dickopp et al., 2011; Lieyu Zhang et al., 2011).

There are numerous constructed wetland types that can be employed and are generally described based on flow. Commonly used constructed wetland types include surface, subsurface, vertical, or horizontal flow systems (Tanner, 1996; Kadlec, 1997). Of these systems, surface horizontal flow, or surface-flow, systems are intended to mimic natural wetlands (Burchell et al., 2007); while subsurface systems are a hybrid between bioretention and surface-flow systems. Figure 3 provides a photograph of a constructed subsurface flow wetland designed to treat septic tank effluent in New Zealand (Tanner and Sukias, 2002). Inundation is a common design specification within surface flow systems. Surface-flow wetlands rely heavily upon macrophytes and the soil-water interface for nutrient removal mechanism. Subsurface wetlands are designed to have higher hydraulic conductivity to permit vertical exposure of nutrients to substrate profiles, although not exclusive of vegetation removal pathways. Improved drainage integrated into subsurface systems generally precludes design with standing water. A typical schematic depicting subsurface and surface-flow constructed wetlands is provided as Figure 4.

Figure 3: Constructed subsurface flow wetland treating septic tank effluent (Tanner and Sukias, 2002)

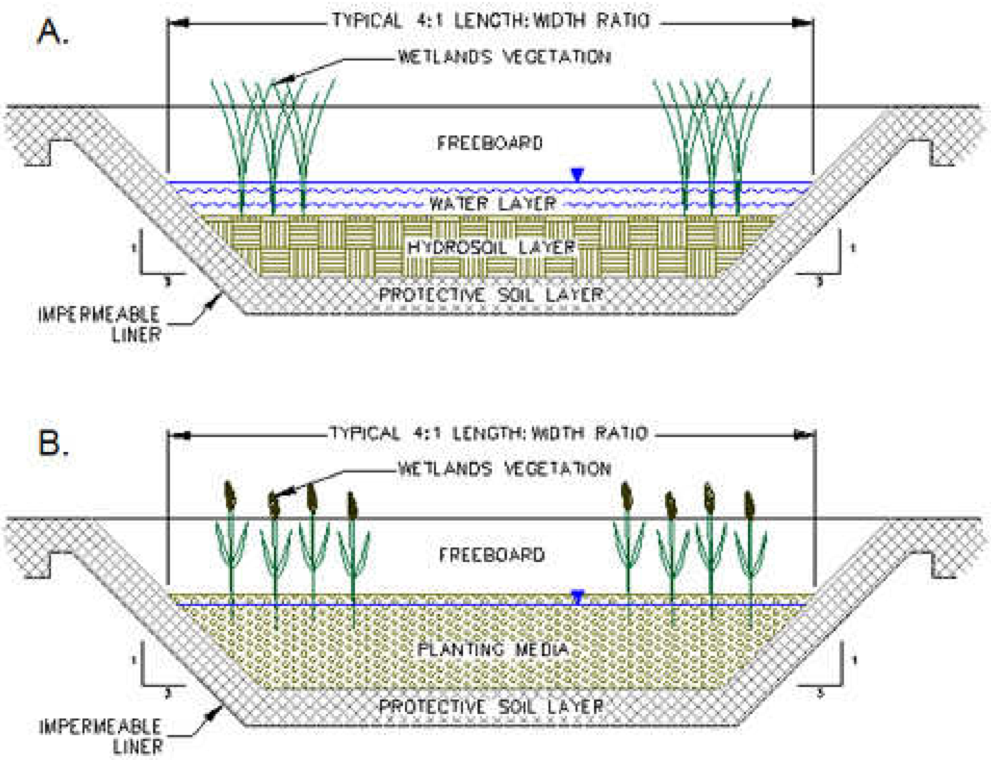

Figure 4: Typical schematic of surface-flow constructed wetlands (United States Department of Energy, n.d.)

The use of constructed wetlands to effectively achieve desired removal rates may require larger areas than what has conventionally been used for effluent treatment and bioretention. This is primarily due to the tendency for pollutant concentrations to decline with increased distance from the input source and with saturation of adsorption sites within existing substrate (Stone et al., 2004).

Whereas bioretention often aligns filtering mechanisms with aesthetic appeal, constructed wetlands are left more as natural or "wild" in appearance. Bioretention relies primarily on below-surface removal mechanisms, leaving vegetation as an attendant benefit. In constructed wetlands the use of vegetation is equally important to nutrient removal as substrate. Therefore, prolific growing herbaceous vegetation used in constructed wetlands does not require aesthetic maintenance. The inundated state typically associated with constructed wetlands also precludes the ability to conduct regular maintenance. Large land requirements, retention of water increasing chance of nuisance insects (e.g., mosquitoes), and a “wild” appearance often prohibits the use of these systems in areas with high human populations or confined spaces.

Functions and Components

Nutrients

Water quality degradation is due heavily in part by the influx of excess nutrients from landscape changes and human activities (Owens et al., 2007; White et al., 2011). Bioretention and constructed wetlands have been the focus on research for the sustainable nutrient removal benefits they provide. Although bioretention and constructed wetlands are capable of reducing a wide range of pollutants, N and P are considered the most ubiquitous nutrients to be related to water quality degradation and will be focused on herein (Brooks et al., 2000; Geta et al., 2004; Hunt et al, 2006). Of N species that are of most interest, ammonium, nitrite, and nitrate are the most targeted; while phosphate and orthophosphate are the most discussed forms of P focused upon (Lieyu Zhang et al., 2011). These nutrients are found in heavy quantities of fertilizers and agricultural and human waste that are often related to point and non-point source pollution (Taylor et al., 2006; White et al., 2011). N and P are directly linked to primary productivity with even minor concentrations exacerbating eutrophication (Brooks et al., 2000; Hunt et al., 2006; White et al., 2011). N and P flux and removal within wetlands occurs through numerous complex pathways and varies depending on environmental factors. According to Hunt et al. (2006) design of treatment systems currently lacks the substantial removal efficiencies that are often expected. The nutrient input species and concentrations into a constructed treatment facility must be assessed to determine the appropriate and reasonable removal rates. Study of the controlling removal factors and continued research focused on pollutant removal and fate mechanisms is critical to successful design of these systems.

Hydrology

Hydrology is the primary controlling factor that determines the nutrient removal characteristics and functional capacity of treatments systems (Thorén et al., 2004; Song et al., 2010). As stated earlier, bioretention systems are typically aerobic in design to only receive hydrologic inputs of short duration. Nitrification processes and substrate adsorption are the dominant pathways in aerobic conditions to remove N and P. When hydrology is manipulated in lieu of an aerobic condition the fate pathways for N and P differ as is typically associated with constructed wetlands.

Unmixed and reduced flow conditions within constructed wetlands leads an anaerobic water column. Coupled with increased water depths, diffusion of oxygen is limited. This anoxic condition shifts N removal to denitrification and liberates P that is otherwise strongly held in aerobic sediments (Ballantine and Tanner, 2010). Denitrification is an important water quality benefit and is regulated by the microbial community within anaerobic zones (Song et al., 2010; Lieyu Zhang et al., 2011). Shallow wetland systems have been shown to become a source for N export; therefore, it is recommended that water flow magnitude (flow and depth) be varied as an additional maintenance function to reduce N export tendencies (Thorén et al., 2004; Song et al., 2010).

Manipulated hydrology that allows drying and rehydration cycles leads to elevated rates of mineralization and removal of N through leaching and flushing (Song et al., 2010). Intermittent hydroperiod also affects the microbial community structure and relationship to denitrification processes (Song et al., 2010). The anaerobic microbial community and resultant denitrification benefits occur maximally at the point where saturation and inundation are juxtaposed. During a drying cycle, the transformation of nitrate-N to gas is halted and nitrification is reinstated, which causes surplus nitrate-N that is available for uptake or mobilization (Song et al., 2010). Song et al. (2010) suggest that frequent hydrologic manipulation resulting in drastic dry and saturated conditions severely alters the microbial community and denitrification processes that may lead to degraded water quality. Ensuring hydrologic inputs and predicting or planning hydrological manipulations is important since constructed wetlands are typically intended as filtration structures prior to discharge. Reduction of denitrification in wetlands intended to treat nitrate-N can lead to deleterious effects within the wetland and receiving waters (Song et al., 2010). Whether it is aerobic or anaerobic, a constructed system can only attenuate a particular hydrologic input and pollution concentration over a specified period of time. The pollution concentration, hydraulic loading rate, and residence time will determine the pollutant removal pathways and design requirements within a system.

The amount of surface water entering the system, or hydraulic loading rate, determines the input required for the system to effectively remove enough pollutant to achieve a desired outflow concentration. Based on nitrification/denitrification and P immobilization processes, researchers have identified hydraulic loading rates that correlate to nutrient removal for a particular treatment system. Hydraulic load rates of 0.1 – 0.3md^{-1} were found to be the point at which maximum N export occurs in various constructed wetlands (Thorén et al., 2004; Spieles and Mitsch, 2000). Although, even at considerably lower loading rates (0.02 md^{-1}) N can still be removed at acceptable rates (35%), but P removal may not be as efficient (8%) (Stone et al., 2004).

The hydraulic load rate may need to be fine tuned as time progresses for a particular system to continually achieve desired removal targets. Fine control of hydraulic entry may not always be feasible, which may result in elevated outflow nutrient concentrations. Hunt et al. (2006) found that effluent concentrations of P were elevated in North Carolina bioretention systems during the wet season when hydraulic loading was higher and flow was unrestricted. Similarly, other research has indicated an increased loss of P during wet weather seasons compared to dry season flows (Fink and Mitsch, 2004). This may be due to anaerobically induced P liberation or translocation of eroded P-bound soil particles. In either scenario, it is evident that differences in hydraulic loading rate will affect nutrient mobility. Additionally, the concentration gradient of nutrients will dissipate as distance increases from the influent source (Stone et al, 2004; Burchell et al, 2007). Flushing of a system with rapid inputs releases and exports nutrients with larger amounts exporting during initial flushes (Thorén et al., 2004). Nutrient concentrations are reduced due to increased water volume, but removal efficiency is unengaged (Thorén et al, 2004). Therefore, when hydraulic rates exceed the system's assimilative capacity, threat of nutrient export may occur.

Residence time needs to be assessed in concert with loading rate to allow the system ample time to process nutrients before outflow (Thorén et al., 2004). Residence time may be reduced in constructed systems designed to encourage surface flow that results in high hydraulic loading rates. This may reduce the ability for P to be removed from a system. For this reason, Brooks et al. (2000) chose a vertical flow wetland setup to more effectively remove P in a New York research wetland and illustrated P removal increases with increased hydraulic residence (Figure 5). Optimal P removal was realized when hydraulic residence was greater than 40 hours (Brooks et al., 2000). Longer residence times may be required for removal of P to allow for compounds to precipitate and ions to adsorb to substrate particles. In Virginia subsurface flow wetlands designed to treat domestic wastewater, Huang et al. (2000) found that ammonium and Total Kjeldahl N (TKN) removal was exponentially more effective as residence time increased. The removal rates were irrespective of wetland configuration and input concentrations, suggesting residence time is a critical factor in nutrient removal efficiencies in any system.

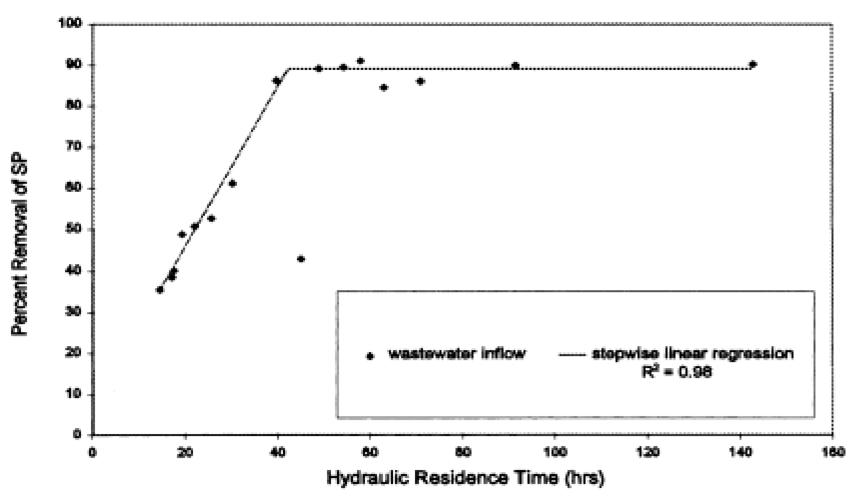

Figure 5: Hydraulic residence time compared to percent removal of P (Brooks et al., 2000)

Climate

P and N-species presence and concentration varies with relation to temporal and climatic regimes (Thorén et al., 2004; Passeport et al., 2009). Nitrate was found to be the dominant N-species found in constructed wetlands during spring and summer months in Sweden, but ammonium was more prevalent during the colder periods of the year (Thorén et al., 2004). In Figure 6, Lieyu Zhang et al. (2011) illustrated the seasonal variation on the removal of N from constructed wetlands in China. Clearly, their results indicate removal rate spikes during the summer months.

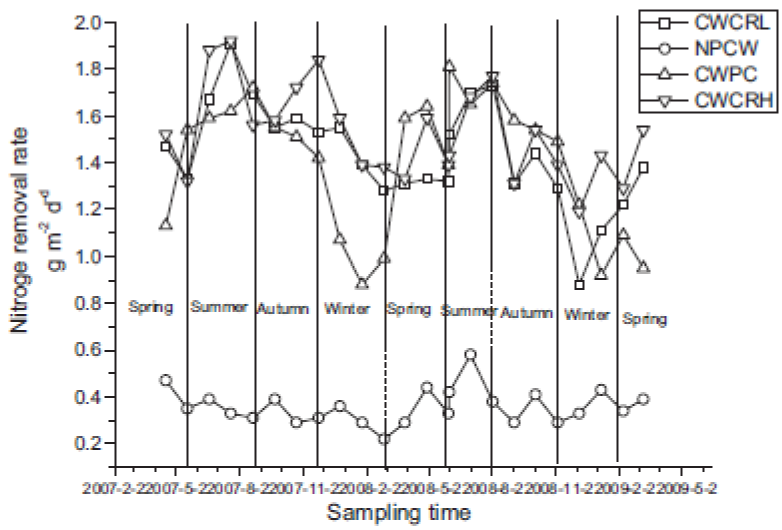

Figure 6: Nitrogen removal rate variation respective to season (Lieyu Zhang et al, 2011).

Within the same wetlands, they found that the rate of N species transformation was more rapid in warmer water temperatures (correlated with seasonal variation) (Figure 7). During warmer periods and within the growing season, microbial activity and plant assimilation occurs at higher rates. It is during these periods that the loading of N into wetland areas are lower as the fate of N is captured prior to entering the system or more readily within the system (Thorén et al., 2004).

This suggests there may be implications regarding removal efficacy within constructed systems in areas with colder climates. These temperature dependent fluctuations make design specifications difficult. As an interesting solution to this conundrum, Huang et al. (2000) were able to develop temperature dependent rate constants to predict nutrient (i.e., TKN) concentrations. Such constant predictions would allow the simplification of constructed system design and confidence in expected performance outcomes.

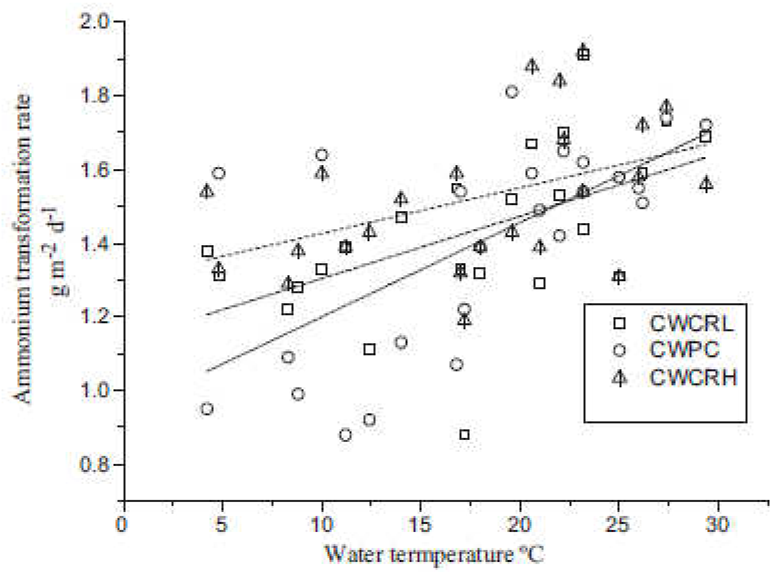

Figure 7: Rate of ammonium transformation rate correlated with water temperature (Lieyu Zhang et al, 2011).

Substrate

Variations in chemical properties of soil determine microbial, plant, and animal communities as well as the capacity of the soil to bind nutrients and contaminants (Gagnon., 1996; Bradshaw et al., 2005). Soil substrates within treatment systems provide a rooting matrix and nutrient source for vegetation establishment and habitat for microbial communities (Burchell et al., 2007). In turn, vegetation and microorganisms contribute to soil aggregate formation and organic C production. The substrate composition within bioretention and constructed wetlands will have a direct influence on pollutant removal efficiencies (Cui et al., 2008).

The selection of media will depend on the influent pollutants, concentration of pollutants, and desired outcome. For example, in watersheds with known excess P concentrations, the use of substrates with a low P-index, high cation exchange capacity (CEC), and binding compounds will provide enhanced absorption (Brooks et al., 2000; Hunt et al., 2006). Similarly, the use of soil with more organic carbon content will provide higher C:N and microbial N transformation in areas with N-species removal goals.

Whereas N removal is primarily related to nitrification and denitrification pathways, P removal is dominated by adsorption to clay colloids and binding to oxides and compounds (e.g., iron, aluminum, calcium). P can be precipitated from solution by iron and calcium or adsorbed to substrate exchange sites (Brooks et al., 2000). Since the fate of P and is not liberated from a treatment system, as is potential with N (e.g., volatilization, denitrification), removal efficiencies may be more difficult to predict. Stone et al. (2004) concluded that P removal isn’t optimally effective in constructed wetlands used to treat swine lagoon effluent. Hunt et al. (2006) found total P removal efficiencies in bioretention systems ranging from 65% to -240%. Lack of P removal efficiency is often attributed to the substrate chosen to be used within a treatment system (Brooks et al., 2000; Ballantine and Tanner, 2010). The dramatic increase in P effluent compared to influent in Hunt et al.’s (2006) research was associated with soil media type and P saturation of the substrate. Therefore selection of substrates for use in treatment systems that require P removal need to consist of soil with high cation exchange capacity and/or substances with tendencies to precipitate P. For example, Brooks et al. (2000) selected wollastonite (calcium metasilicate) to be used as a P removal substrate in a wastewater treatment wetland with good results (98% removal in 12 hours). This was further confirmed by studies performed by Hunt et al. (2006) where they found the material with a low P-index and high CEC is of utmost importance to effectively remove P. Additional information regarding substrate composition will be discussed later in more detail.

The properties of soil conditions for nutrient removal are not always ideal upon construction of a treatment system. Oftentimes, maturation of a wetland is necessary to develop biogeochemical removal functions. In part, this is due to sequestration of C into the soil profile that is available for microbial-induced nutrient flux. Aside from C amendments incorporated during construction of a system, soil C contributions are through plant roots and leaf litter, which may take time to develop. In a comparison of natural and constructed wetlands, Fennessy et al. (2008) presented that natural wetlands have significantly more organic carbon than constructed systems, 15.1% and 3.1% respectively. They also discovered that the vegetation within natural wetlands assimilated more nutrients than constructed site vegetation. Adding C to constructed systems may increase construction and maintenance costs but may provide enhanced denitrification rates (Burchell et al., 2007).

Care should be taken during construction of treatment systems to reduce impacts to native soil. It has been hypothesized that soil compaction and removal of layers of organic matter (e.g. A horizon) reduces the ability for a system to propagate vegetation, colonize microbial populations, and flux nutrients (Burchell et al., 2007; Fennessy et al., 2008). Experiments conducted by Cui et al. (2008) suggest that there was a negative correlation between soil compaction characteristics (bulk density, hydraulic conductivity, and total porosity) of substrates with P sorption. Reducing the physiochemical properties of the substrate during construction can result in a delay of achieving system equilibrium and nutrient removal benefits. Addition of organic matter to constructed system substrate has been shown to expedite biomass production and nutrient removal efficiencies in constructed systems (Burchell et al., 2007). Burchell et al. (2007) recommends locating sources for recycled substrate material that may be dredged or removed from construction sites or impacted wetlands elsewhere as OM rich substrate that will jump-start the system.

Additives specific to particular pollutant removal needs may also be required, such as calcium-rich byproducts (e.g., wollastonite) (Brooks et al., 2005). Soil amendments other than OM are usually incorporated to target the removal of P in constructed settings due to physiochemical pathways inherent in soil attenuation of P (Cui et al., 2008). Amendments are best utilized if sources can be located in close proximity to the proposed treatment facility to reduce importation costs. Ballantine and Tanner (2010) presented a comprehensive list of soil amendments and filter media that could be used to improve P removal efficiencies in constructed treatment wetlands (Table 1). Although these materials may not score similarly in different geographic areas, the table provides a good reference for potential amendments and filter media and illustrates the abundant choices available to increase substrate functionality. It is also recommended that different amendments are mixed to obtain optimal rates of P sorption and include both a high P-index and percolation rate to avoid export issues (Cui et al., 2008). Substrate and amendment choices should consider their ability to support and propagate macrophytes necessary for nutrient abatement (Calheiros et al., 2009).

Table 1: Potential soil amendment and filter media for use in constructed treatment of P (Ballentine and Tanner, 2010).

Material	Useful for	P removal	Availability	Likely cost	Reuse	Score
Allophane	Soil amendment	High	Medium	Low	Beneficial	9
Pumice soil	Soil amendment	Low	Medium	Low	Beneficial	7
Sand	Soil amendment	Low	Medium	Low	Neutral	6
Amended sand	Soil amendment	Medium	Medium	High	Difficult	4
Tephra (P)	Soil amendment	High	Medium	Low	Beneficial	9
Pumice	Filter	Medium	Medium	Medium	Neutral	5
Shale	Soil amendment	High	Low	Low	Beneficial	8
Shell-sand	Filter	Medium	Low	Low	Beneficial	7
Limestone	Both	Medium	High	Low	Beneficial	9
Serpentinite	Filter	Medium	Medium	Medium	Neutral	6
Wollastonite	Filter	Medium	Medium	Medium	Neutral	6
Zeolites	Filter	Medium	Medium	Medium	Beneficial	7
Phosphate rock	Filter	Medium (based on apatitie performance)	Low	High	Beneficial	5
Alum	Soil amendment	High	High	Medium	Difficult	7
Amended zeolites	Filter	High	Low	High	Difficult	4
Filtralite-P	Both	High	Low	High	Beneficial	8
LECA	Both	High	Low	High	Beneficial	8
Phoslock™	Both	High	Low	High	Beneficial	8
DWTRs	Soil amendment	High	Medium	Low	Difficult	7
Fly ashes	Soil amendment	High	Medium	Low	Difficult	7
Seashells	Filter	Medium	High	Low	Beneficial	7
Slag	Both	High	Medium	Medium	Beneficial	8
Fe-based materials	Soil amendment	High	Medium	Medium	Difficult	6
Tree bark	Filter	Low	High	Low	Useful	8
Subsoil	Substrate	Medium	High	Low	Beneficial	9

Score derived by addition as follows:
P-removal potential: 1 = low, 2 = medium, 3 = high
Availability in NZ: 1 = low, 2 = medium, 3 = high
Likely cost: 1 = high, 2 = medium, 3 = low
Reuse potential: −1 = difficult, 0 = neutral, 1 = beneficial

Dependency of P removal on substrate choice indicates the need for an increased areal treatment extent to reduce potential for P saturation with available substrate. Furthermore, a pretreatment of P laden water should be utilized for increased removal efficiency of P prior to discharge into a constructed wetland (Brooks et al., 2000; Stone et al., 2004). The combination of bioretention and constructed wetlands may be a more sustainable solution to remove P. Pre-filtering of P in a bioretention area may be beneficial in delaying or inhibiting P saturation of the receiving constructed wetland. Fink and Mitsch (2004) offer an alternative configuration where a bioswale is utilized as a polishing system of effluent discharged after treatment within a constructed wetland, not pretreatment. Although, this configuration may be more suited for N removal than that of P when considering P-saturation issues.

The soil-water interface is an active zone of nutrient transformation and removal pathways (Burchell et al., 2007). The removal of nutrients can be disrupted or enhanced in this zone by infaunal perturbation of treatment facility sediments. Bioturbation has been shown to both adversely and positively affect removal efficiencies and eutrophication (Angeler et al., 2001). Sediment remixing increases oxygen flux into substrate and modifies nitrification and denitrification rates. Macrofaunal sediment reworking and burrowing has been shown to flux nutrients between sediment and the overlying water column (Kristensen and Hansen, 1999; Aigars and Carman, 2001; Mermillod-Blondin et al., 2005). Bioirrigation nutrient flux can lead to elevated internal loads and also limit primary production (Angeler et al., 2001). Bioturbation increases the flux of reduced substances stimulating re-oxidation in oxygenated burrows to lead to reduced and liberated nutrients (Goňi-Urriza et al., 1999). Webb and Eyre (2004) found that bioturbation was responsible for reducing N concentration levels by 99%. Conversely, the reworking of sediment can release P from anoxic layers and contribute to export loading (Angeler et al., 2001). The role of bioturbation may be a consideration for the removal expectations of constructed wetlands, particularly when determining nutrient removal rates over time. Recolonization of defaunated or newly constructed systems has been shown to promote mineralization and accelerate nutrient flux (Hansen and Kristensen, 1997).The magnitude of bioturbation-related nutrient flux may increase as a wetland matures and substrate fauna communities develop.

Vegetation

Both bioretention and constructed wetland design must incorporate vegetation as a component in the sustainable functioning of each system (Tanner, 1996; Wong, 2006). The species composition and community structure varies between the two types of treatment centers. The persistence of water typical in constructed wetlands often precludes entry for maintenance or pedestrian recreational use; therefore, these treatment systems are often left in a naturalized vegetative state. In contrast, bioretention facilities are usually associated with urban and suburban landscapes where human interaction is more likely. In areas where aesthetics are a consideration the use of landscape grade vegetation is a viable option and maintenance is more easily conducted. Additionally, wetlands require the use of hydrophytic vegetation that can withstand extended periods of soil saturation and inundation. Facultative or upland species are a more sustainable option for bioretention facilities that drain rapidly. The effectiveness of one species over another to immobilize pollutants can vary widely and is a consideration when trying to attain maximal removal efficiencies. The following list, modified from Tanner (1996), is a general guideline when considering plant selection for a constructed treatment system:

- Climate conditions and plant adaptability – Use of species suited for growth in local climatic and soil conditions.
- Tolerance to target nutrients/pollutants and local antagonists – Use of species that will withstand high concentrations of pollutants carried by input water and tolerance to disease and pests.

- Hydrological requirements and tolerance – Use of species suitable for the anticipated hydrological regime (e.g., hydrophilic vs. hydrophobic).
- Assimilation capabilities – Use of species that are expected to uptake the pollutants targeted for removal within the system.
- Ecological suitability (e.g., native vs. invasive/exotic species) – Use of species that do not threaten the local ecological structure and are less susceptible to issues, such as disease and drought.
- Ease of installation and establishment – Use of rhizomes and bare-root plants to accelerate propagation opposed to seeding.
- Additionally, vegetation selection with depend on the overall project objectives, type of system, plant availability, maintenance expectations, aesthetic requirements, and system size.

Evidently there is some disagreement regarding the role of vegetation in the effectiveness of nutrient removal in constructed system (D. Zhang et al., 2009). Some researchers have found no reasonable correlation between plant uptake and nutrient removal efficiency (Saunders and Kaliff, 2001), but others have found that plants were significantly valuable in N and P removal (Liu et al., 2000; Tanner, 1996). Tanner (1996) found that eight emergent wetland plant species were responsible for the removal of over 90% of N and P from wastewater treatment wetlands. The contribution of plant assimilation to nutrient removal efficiency is related to temporal and climatic conditions and plant species composition.

Plant species composition and primary productivity will often determine nutrient removal efficiency of a wetland (Reeder, 2011). Furthermore, removal efficiency differences may not always be evident between species. Research conducted in New Zealand compared various wetland plant species efficiencies with results indicating removal of nutrients was relatively consistent, regardless of emergent plant species (Tanner, 1996). Research conducted by Huang et al. (2000) echoes this assertion, where they found no difference in ammonium and TKN removal between cattail (*Typha lattifolia*) and woolgrass (*Scirpus cyperinus*) plots. Reeder (2011) reported, at the Beaver Creek Wetlands Complex in Kentucky, that macrophytes within constructed emergent wetland habitats showed higher net primary productivity than submerged and open water systems. Comparisons regarding Tanner's (1996) and Reeder's (2011) results point to aggressive productivity found in submerged and emergent species and associated enhanced removal efficiencies in both experiments. This evidence suggests that constructed wetlands may be more functional if designed respective to submerged/emergent community persistence opposed to a canopy-related or woody stemmed structure.

Functions and nutrient removal efficiency shifts as a wetland matures (Campbell et al., 2002). Thorén et al. (2004) hypothesized that in early stages of constructed wetland inception, when plants are immature and natural senesce is still minimal, nutrient removal is elevated. This is attributed to plant uptake for use in biomass accumulation through growth and the rapid colonization of the sediments by microbes (Thorén et al., 2004). This is further corroborated by Tanner (1996) where a positive linear relationship was discovered between biomass quantity and N removal in wastewater wetlands (Figure 8).

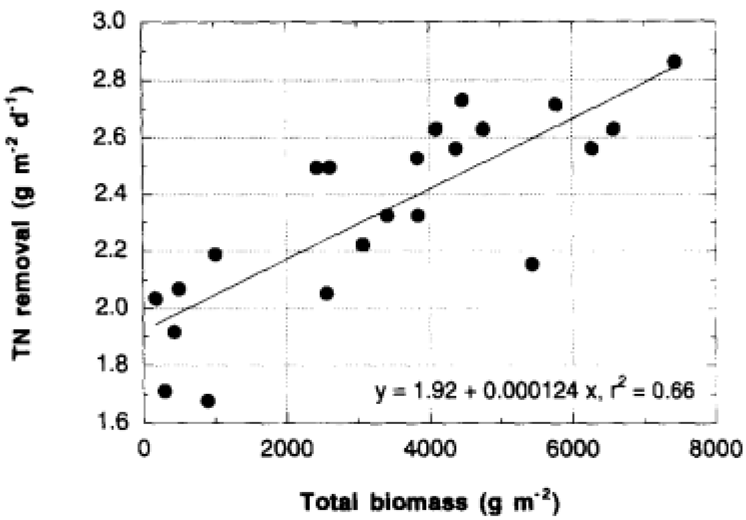

Figure 8: Positive linear relationship between total emergent species biomass and TN removal (Tanner, 1996)

As a wetland matures, its nutrient removal efficiency decreases as a result of a plant species limitation to uptake nutrients. This effect is compounded by processes such as mineralization from senesced plant material and plant exudates, where nutrients are released back into a system after assimilation (Thorén et al., 2004). However, mature wetlands may show increased rates of denitrification as they tend to accumulate organic matter (Fink and Mitsch, 2004). Vegetative cover is also expected to increase as a wetland matures, thereby detaining flow and hydrological residence time (Fink and Mitsch, 2004), which leads to implications regarding hydraulic loading rates and retention discussed earlier.

Vegetation removal capacities are centered on assimilation of nutrients for use in biomass production and biogeochemical processes associated with the root-zone. Plants uptake N and P for generation of plant tissue thereby making these nutrients unavailable for mobilization. Rhizosphere-related processes such as nitrification direct the fate of nutrients in close proximity to roots (Tanner, 1996). Oxygen leakage within the carbon-rich rhizosphere provides a favorable environment for aerobic microorganisms that are responsible for these processes (D. Zhang et al., 2009). The use of hydrophytic vegetation with higher amounts of aerenchyma tissue, such as *Typha* and *Phragmites*, will provide more oxygen to the root-zone (Fennessy et al., 2008; Dickopp et al., 2011). It is then further advisable to select vegetation known to possess extensive root systems to maximize the positive removal effects promoted by rhizospheres. Furthermore, senescence of aboveground biomass contributes to soil carbon additions that further stimulate soil faun activity. The rate of senesce will vary depending on vegetative cover density, species, and climate, which in turn will affect nutrient removal efficiency (D. Zhang, et al., 2009).

Microbial communities influence the rate of decomposition of leaf litter and associated soil conditions and nutrient pathways. Fennessy et al. (2008) suggested that the microbial communities in constructed wetlands may be much different than natural systems; therefore, contributing to conflicting information in comparison research between natural and constructed wetlands. Constructed systems may also have a less diverse microbial community due to limitations in the quantity and quality of N and C in constructed sediments (Fennessy et al. 2008). This assertion is further defined by Dong and Reddy (2010) who found diversity and enumeration of bacterial communities was directly correlated with nutrient and OM concentrations. Retention and proliferation of microbial communities may be more pronounced by manipulating plant communities. Calheiros et al. (2009) found that a positive relationship may exist between macrophyte density and diversity and bacterial communities within adjacent soil and rhizospheres of wetland plants. Rhizosphere benefits to microorganisms discussed previously assist in understanding the validity of this correlation.

It has been found that microbe population density and species distinction varies within a particular constructed wetland based on vegetation (Li et al., 2008). The benefits between plants and microbes are often symbiotic and can have an effect on treatment effectiveness. Fennessy et al. (2008) discovered the assimilation of N by vegetation within natural wetlands was considerably higher compared to constructed systems. They hypothesized the lack of carbon and nutrient sources within the constructed wetland reduced the microbial community structure and activity, thereby decreasing the nutrients available for plant uptake. This indicates the nutrients and carbon sources are limiting the productivity of the system through a chain reaction of insufficient constituents. Natural systems have had considerable more time to develop sufficient limiting components and a more efficient biological condition.

Maintenance

Maintenance of constructed wetlands and bioretention facilities may be necessary to continually achieve pollutant removal rates. As previously discussed, a maturing system may tend to become a source of the nutrient it was intended to abate. Therefore, maintenance or contingency actions may be required to ensure perpetual removal efficacy. Assuming hydrologic inputs do not change and were designed correctly, maintenance will be associated with substrate and vegetation. Removal of substrate and vegetation that has sequestered excess nutrients is the typical course of action.

Some suggest that maintenance of constructed wetlands, in the form of harvesting plant material, may be helpful in reducing N loading rates (Thorén et al., 2004). Other studies have concluded that vegetation harvesting has no effect on the nutrient removal efficiencies of a system (Wetzel, 2001). Since vegetation is responsible for assimilation of abundant quantities of nutrients the removal of this vegetation after the growing season may be warranted. Harvesting is intended to avoid release of pollutants back into the system during mineralization processes.

It should be noted that the removal of senesced plant material thereby removes available C sources utilized by the microbial community during respiration. The lack of C availability as an energy source for microorganisms can drastically reduce the denitrification and removal efficiency as depicted in Figure 9 (Song et al., 2010; Passeport et al., 2009). The presence of labile C permits the achievement of a favorable C:N for microbial respiration, which leads to removal of N from the system (Passeport et al., 2009; Lan Zhang et al., 2011). Where labile C sources are not readily available, such as bioretention areas that are heavily managed for aesthetics, it may be necessary to add C sources to the treatment facility in order to achieve maximal microbial activity and denitrification effects. Laboratory experimentation where C amendments such as newspaper, mulch, compost, and straw have been used for microbial energy sources, removal rates of N approached 100% (Kim et al., 2003). Kim et al. (2003) concluded that newspaper clippings were the most favorable C amendment. Although the addition of highly labile C sources would be relatively simple to implement during initial facility construction, the benefits of post-construction amelioration may be outweighed by cost of maintenance and periodic testing of C requirements. Lack of field studies related to the improved denitrification properties associated with amendments may also preclude designs with C amelioration management plans (Hunt et al., 2006).

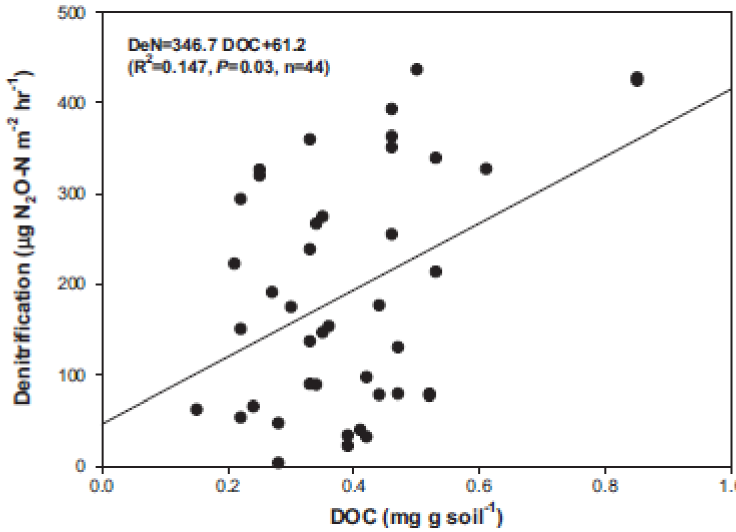

Figure 9: Linear regression of denitrification rates increasing in relation to dissolved organic carbon concentration (Song et al., 2010).

Although the removal of senesced and standing plant material after the growing season may reduce reintroduction of N in the system, there are economic, training, and consistency issues to consider. Entities that require the use of constructed wetlands to treat stormwater runoff typically do so to satisfy regulatory requirements and are less likely to commit to a long-term maintenance plan. If harvest maintenance is an applicable approach, it is recommended that harvest occur after senesce and prior to the next growing season to avoid undue disturbance. This is when cooler temperatures and photoperiod reduce microbial and plant activity and senesced biomass is mineralized, adding N to the system.

As a wetland matures, P removal efficiency tends to decrease as the site become saturated with removal capabilities and may eventually become a source of exported P (Ballantine and Tanner, 2010; Fink and Mitsch, 2004). Removal of P-saturated substrate may be required to avoid P export. This would require the mechanical removal of the substrate and appropriate disposal, which may be reuse as fertilizer (Ballantine and Tanner, 2010). The site would then require reintroduction of suitable substrate to “reactivate” the P removal properties of the system. This is potentially expensive procedure that should be considered in areas where P loading rates are a concern.

Because these systems are often incorporated into human-centric areas, maintenance may also need to include aesthetic grooming, damage repair from vandalism, and periodic monitoring and inspections (Somes et al., Furthermore, it should be determined at what frequency and to what intensity maintenance should occur. If the wetland is capable of removing enough N to satisfy goals on a three year cycle, maintenance should not occur more frequently. Baseline studies to determine success requirements are valuable in reducing unnecessary costs. Identifying a balance between constructed wetland maintenance requirements and project objectives will be integral to a cost-benefit analysis during the planning and design phases of a constructed wetland project.

Discussion and Conclusion

Use of bioretention facilities and constructed wetlands as treatment systems is a growing and effective method for attenuation of pollutants for point and non-point source loads. Selection of the appropriate treatment BMP will depend on the project’s goal and objectives. For treatment of stormwater runoff in urban and suburban environments the use of a bioretention facility is warranted. Since bioretention facilities have been shown to exhibit effective P removal efficiencies there use is also suited in areas where P loading is a concern. Conversely, in areas where nitrate-N removal is desired, the use of a constructed wetland system with persistent hydrology to maximize denitrification effects is a better choice. Oftentimes N and P are found together in nutrient lade water. In areas where the objective is to reduce both N and P concentrations the design should include a shallow fluxing wetland section close to inputs and a deeper water system with persisting anaerobic zones near facility exits (Fink and Mitsch, 2004). Dong and Reddy (2010) showed that N removal in constructed wetlands designed to treat swine effluent depended heavily on aerobic nitrification early in the treatment process and denitrification processes in following aerobic zone. An alternate solution may be to combine bioretention and constructed wetlands. Bioretention can be used as a pre- and/or post-treatment to constructed wetlands to capture the full spectrum of nutrients and increase system longevity.

One of the most important factors in treatment system success is the hydrologic approach. The hydraulic load rate, depth, and retention time will determine the biogeochemical process that will occur and the resultant nutrient efficiencies. For example, inundated systems with moderate load rates and extended retention time may be best favored for use in targeting denitrification processes to reduce nitrate-N concentrations. Conversely, in projects targeting ammonium or P removal, a more aerobic condition may desire a hydrologic regime with lower retention time, reduced depth, and higher flow rates. Climatic conditions and seasonality will play critical roles in determining the hydrologic fluctuations.

Substrate selection should include characteristics that promote adsorption, moisture retention, appropriate C:N, and suitability for biological propagation. Substrates can be amended to increase these capabilities and further enhance the sequestration properties of the treatment system. Replacement of substrates may be required when the adsorption capacity of the system is reached, increasing maintenance and management costs.

Incorporation of vegetation communities to enhance filtering mechanisms should focus on aesthetic in bioretention areas and primary production in constructed wetlands. Since bioretention relies more heavily on substrate removal mechanisms, the use of vigorous biomass producers is likely unnecessary. In constructed wetlands, the assimilation of nutrients and promotion of rhizosphere-inhabited microbes will benefit from the use of emergent herbaceous vegetation. Biomass reduction may be required under a maintenance plan to remove assimilated nutrients prior to natural senescence and reloading of nutrients to the system. In bioretention areas, maintenance will be more associated with mulching and grooming to enhance aesthetic value.

The degree of desired removal efficiency will dictate the size of the constructed system. Land area required for implementation of a treatment system is a common constraint, which is dictated by factors such as land cost and facility size required for effective treatment. When considering use of a treatment BMP in urban and suburban areas where land area is limited, it may be best to utilize bioretention systems. They require less land surface to implement and can be included as an aesthetic focal point. For constructed wetlands, considerably more area is required. Size of the system and required land should be considered to maximize retention capabilities without inducing export of trapped nutrients (Kohler et al., 2004). Consideration regarding placement of a treatment system should include landscape position to achieve the most sustainable configuration. To reduce excavation and potential pumping energy costs, it is advisable to locate treatment systems in low-lying areas where topographic relief can direct influent. The cost of land in low-lying areas may also be more affordable than desirable upland, buildable lots.

Constructed systems tend to be homogenous in design and lack the complexity found in nature (Fennessy et al., 2008). Focusing design on attempting to mimic natural wetland functions and allowing sufficient time to transpire will improve constructed system efficacy. It takes time for a wetland to mature to a point where all contributors to removal processes are in place. Because of this, it may take considerable time to realize the success or failure of a constructed system as shifting dynamics begin to reach equilibrium. Mitch and Wilson (1996) suggest that it may take 15 to 20 years or longer for the potential of a constructed system to be achieved. During the time leading up to an equalized state, it may be necessary to engage contingency actions when deficits in performance are discovered.

As a result of this review, it is clear that there is still a lack of knowledge regarding the mechanisms surrounding nutrient removal in constructed systems. Further research is required to understand the design characteristics that best suit the removal goals of a particular project. It is then important to identify the constants that can be used to predict removal efficiency rates with respect to climate, hydraulic load rate, retention time, substrate type, and pollutant concentration. Understanding these facets will further expand the successful use of bioretention and constructed wetlands. Although research to this point has shown high variability in removal efficiencies by bioretention and constructed wetland facilities, the considerations stated herein can be helpful in the design of these systems.

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