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Design and performance of denitrification bioreactors for agricultural drainage

by

Laura Elizabeth Christianson

A dissertation submitted to the graduate faculty

in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

Co-majors: Agricultural Engineering; Sustainable Agriculture

Program of Study Committee:
Alok Bhandari, Co-major Professor
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Ames, Iowa

2011

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CHAPTER 1 INTRODUCTION

A paper in preparation as a review article for the journal *Applied Engineering in Agriculture*

Introduction

Artificial subsurface agricultural drainage helps increase agricultural productivity by improving the timeliness of field operations and the workability of the soil in many locations around the world (Skaggs and van Schilfgaarde, 1999). However, the detrimental water quality impacts precipitated by such drainage systems are a concern in many of these locations. For example, in the US Midwest region, artificial agricultural drainage has been done for over 100 years (Dinnes et al., 2002), but over the past several decades, nitrogen loadings from drainage in this area have been causally implicated in one of the United States' largest water quality concerns: the hypoxic zone in the Gulf of Mexico (Rabalais et al., 1996; Turner and Rabalais, 1994; USGS, 2000a). In the summer of 2011, this hypoxic zone was larger than average (USEPA, 2011a) and in response to multiple years of such large zones (NOAA, 2009), the United States Environmental Protection Agency (USEPA) has called for a minimum of 45% reduction in total nitrogen loads in the Mississippi River (USEPA, 2007). Additionally, local water quality impairment stemming from agricultural drainage is beginning to be targeted by regulatory agencies around the world with programs like the Total Maximum Daily Load (TMDL, e.g. Cedar River, Iowa TMDL; IDNR, 2006), drainage permitting (Horizons Regional Council, 2007) and nitrogen trading schemes (Rutherford and Cox, 2009).

Denitrification bioreactors for agricultural drainage are one of the newest technologies being investigated for practical edge-of-field nitrate-nitrogen (NO_3^--N) reduction. Also known as woodchip bioreactors, denitrifying bioreactors, denitrification beds or biofilters, promising early results from these systems have led to increased attention over the past few years. A number of popular press pieces (e.g. Ag PhD, 2010; Caspers-Simmet, 2010; Willette, 2010), federal agency interest in design standards and research performance (USDA NIFA, 2011; USDA NRCS, 2009), and local commodity and watershed group activities (ISA, 2010; ISU Extension, 2006) are evidence of this interest across multiple sectors.

With a growing number of denitrification bioreactors being installed in drainage systems across the US, there is need for a comprehensive document which combines the latest available information on these systems. Though a review article on denitrification treatment was recently released (Schipper et al., 2010b), there are several critical issues that make denitrification treatment of drainage water distinct from treating groundwater, septic wastewater or other controlled wastewaters as discussed in this previous review. This paper provides a practice-oriented discussion of the design,

installation, performance, and monitoring of bioreactors as specifically related to the unique characteristics of agricultural drainage. Because the quality of agricultural drainage waters is not regulated in the US, any such treatment of these waters will be voluntary, and thus must remain practical. This additional dimension of farm-scale practicality is a theme which necessarily runs throughout this review.

Literature Review

Agricultural Drainage

The history of artificial, subsurface tile drainage in the US Midwest began in the late 1800's (Dinnes et al., 2002), with artificial drainage now implemented on more than fifteen million hectares in this region (Sugg, 2007). It is thought that the ability to drain the Midwest "prairie pothole" region in combination with the increased use of inexpensive N fertilizer precipitated the intense increase in agriculture in this region which has led to a large positive impact upon the U.S. economy (Dinnes et al., 2002). Unfortunately, NO_3^--N loadings in Midwestern drainage can be very high (e.g. 81 kg N/ha in Kaspar et al. (2007) and 88 kg N/ha in Lawlor et al. (2008)); it is not unusual for NO_3^--N concentrations in drainage to exceed the USEPA drinking water standard (10 mg NO_3^--N/L)(USEPA, 2011b). Typical NO_3^--N loadings in this region are on the order of 25-35 kg N/ha with typical flow-weighted concentrations varying from 10 mg NO_3^--N/L to over 25 mg NO_3^--N/L (Jaynes et al., 1999; Kalita et al., 2006; Lawlor et al., 2011 Accepted).

The drainage volume and N load can vary significantly in US tile drainage systems as these are affected by a number of factors including soil type, cropping rotation, management decisions, tillage, weather patterns, and drainage system characteristics (Kladivko et al., 2004; Patni et al., 1996). Clearly, the effects of precipitation on drainage and NO_3^- losses are important with 25% to 41% or 7% to 22% of precipitation exiting a field as drainage in Illinois and southern Minnesota, respectively (Gast et al., 1978; Gentry et al., 2000). These values also highlight the spatial differences in drainage hydrology; frozen soils during the winter in the northern Midwest preclude drainage during this period, whereas in the southern Midwest, winter drainage is common. Annual variation in precipitation can have bearing with wet years (e.g. the flooding of 1993 in the Midwest) leading to very high drainage NO_3^- losses (Kanwar, 2006). This importance of variation between years was reiterated by Gentry et al. (2000) who noted a poor crop yield in the previous year might increase leaching of NO_3^- from the stored N pools in the soil. Within a given year, there are seasonal considerations with potentially low NO_3^- concentrations in winter drainage due to snow melt (Patni et

al., 1996) and high drainage volumes in spring generally making this season the most critical period for N loading and management (Kladivko et al., 2004; Mirek, 1999; Patni et al., 1996).

More broadly, drainage systems vary globally with tile drainage predominant in the US but "mole and tile" or "mole and pipe" drainage more common for impermeable clay plastic soils in locations such as New Zealand, Australia, and the United Kingdom (Bowler, 1980; Hopkins, 2002). In these drainage systems, the mole channels are unlined tunnels installed at shallow depths (40 to 70 cm) and narrow spacings (2 to 3.5 m) using a cylindrical foot and expander to help compact the tunnel wall (Bowler, 1980). Drainage from such systems tends to be extremely precipitation driven with dry periods between drainage events and can have distinct NO_3^- loss trends (Bowler, 1980). In New Zealand for example, NO_3^- concentrations typically decline over the winter drainage season with the highest concentrations occurring within the first 100 to 150 mm of drainage (Houlbrooke et al., 2004; Monaghan et al., 2002). This variability between drainage systems combined with the multitude of agricultural management decisions and environmental factors means that addressing NO_3^- in drainage will require a variety of approaches.

Options for Improving Drainage Water Quality

There are a number of agricultural practices and technologies that reduce the amount of N in drainage, reduce the amount of drainage, or treat N edge-of-field. Several proposed options include increased use of wetlands, better crop rotations and cover crops, and improved fertilizer application rate and timing (Dinnes et al., 2002). There are also a variety of constructed practices like controlled drainage and, now, denitrification bioreactors that show promise (Appelboom and Fouss, 2006). It is likely that a combination of two or more such methods and management strategies will be necessary at many sites to meet water quality goals (Dinnes et al., 2002).

Some standard conservation practices will be ineffective for treating drainage or will have other limitations. For example, vegetated filter strips are not suitable for subsurface drainage water treatment because they rely primarily upon infiltration (Cooke et al., 2001) and cannot treat flow that is short-circuited through by drainage pipes (Kovacic et al., 2000). Jaynes et al. (2004) reported that phytoremediation and cover crops may have disadvantages including land out of production or yield losses. Fertilizer timing and rate modifications can have a positive water quality impact, but in some cases, even applying at below the economically optimum fertilizer rate produces drainage NO_3^- concentrations that don't meet water quality goals (Jaynes et al., 2001). While optimization of N fertilizer rate and timing can play a role in water quality improvement (Lawlor et al., 2008; Randall

and Mulla, 2001), it is unlikely that water quality goals will be met with this option alone (Randall and Sawyer, 2008).

Controlled drainage (alternatively, water table management) was developed specifically to help address drainage water quality concerns via decreased drainage volume (Dinnes et al., 2002; Gilliam et al., 1979). Regardless of potentially high N load reductions (mean reduction 41% in a review by Christianson et al., (Under review), the major limitation of this approach is that it can only most effectively be used on slopes less than 0.5% to 1% due to the number of control structures required (Dinnes et al., 2002; Frankenberger et al., 2006). Wetlands are another option that have high N removal effectiveness and, unlike some other water quality practices, also offer a number of beneficial ecosystem services including wildlife habitat and flood regulation (Iovanna et al., 2008). There has been success recently with constructed wetlands in Iowa (IDALS, 2009), but their initial cost makes broad-scale implementation difficult.

There are several processes currently used in the waste water industry that could also be considered for drainage treatment. These include attached growth bioreactors, packed or fluidized bed bioreactors, ion exchange, biological and chemical denitrification, reverse osmosis, lagoons, and passive bioreactors with biofilm (Cooke et al., 2001; Jaynes et al., 2008). For agricultural field-scale applications, biological denitrification in passive reactors is the most practical in terms of expense and maintenance (Cooke et al., 2001; Jaynes et al., 2008). Importantly, because agricultural drainage water quality is not currently regulated in the US, the practicality of an agricultural treatment system must always be a major focus.

Nitrogen Cycle and Denitrification

Modification of the global N cycle over the past century through increased usage of N fertilizer has led to vastly increased food production but also major worldwide environmental impacts (Diaz and Rosenburg, 2008; Gruber and Galloway, 2008). From the 1960's to the 1990's in the United States, N fertilizer use increased by 2.4 kg/ha/yr (Dinnes et al., 2002), and today's N inputs into cropped agricultural systems remain substantial (USDA ERS, 2011). These changes in agriculture over the past century, including both increased N inputs and subsurface drainage, have decreased N cycling efficiency (i.e. N use efficiency) compared to the once existing natural system (Dinnes et al., 2002). While undoubtedly these changes have impacted the amount of N in surface and ground waters, it also bears to mention that increased inorganic N applications are not the sole cause of Midwestern drainage water N loads. Mineralization of soil N is an important process which can eventually contribute to N leaching indicating that part of these losses are due to the lack of

synchrony between the time of nitrate availability in the soil and period of plant uptake (Dinnes et al., 2002).

Denitrification, the microbially mediated reduction of NO_3^- to N_2 (Eq. 1.1), is one of the most important possible fates of NO_3^- in the soil (Tiedje, 1994). Artificial drainage modifies the nitrogen cycle as well as the hydrologic cycle in agricultural systems; the relatively rapid transport of drainage water in tile drains decreases the time for natural processes like denitrification to occur (Kellman, 2005). Moreover, denitrification in soils can be carbon limited especially at deeper depths, significantly reducing the ability to denitrify soil solution before it becomes drainage (Moorman et al., 2010).

$$5C + 4NO_3^- + 2H_2O = 2N_2 + 4HCO_3^- + CO_2 \qquad \text{Equation 1.1}$$

In order for denitrification to proceed, the requirements are (1) N oxides (e.g. NO_3^-, NO_2^-, NO, N_2O; the electron acceptors) (Eq. 1.2), (2) denitrifying bacteria, (3) carbon source (electron donor), and (4) suitable dissolved oxygen (DO) conditions (Korom, 1992). Under saturated conditions, bacteria utilize oxygen to process (oxidize) the available carbon. When oxygen concentrations become limiting, facultative anaerobes become active using NO_3^- as electron acceptors in their respiration electron transport chain. This limiting DO level varies amongst the numerous denitrifying organisms (Korom, 1992), though DO concentrations as low as 0.2 mg/L are able to inhibit denitrification from reaching maximum rates (Metcalf and Eddy, 2003). Denitrifying bacteria are a very diverse group of mostly facultative anaerobes, and though there are both autotrophic and heterotrophic denitrifiers, most are heterotrophic (Korom, 1992). Organic matter is nature's most common source of electrons for reactions and it yields the most energy for these heterotrophs (Korom, 1992). Autotrophic denitrifiers utilize Mn^{2+}, Fe^{2+}, or sulfides as an electron donor rather than carbon (Korom, 1992).

$$NO_3^- \xrightarrow{nitrate\ reductase} NO_2^- \xrightarrow{nitrite\ reductase} NO \xrightarrow{nitric\ oxide\ reductase} N_2O \xrightarrow{nitrous\ oxide\ reductase} N_2$$

Equation 1.2

After nearly complete reduction of NO_3^- and with further decreases in reducing conditions, obligate anaerobes become active and use other electron acceptors such as sulfate (SO_4^{2-}), manganese (Mn (IV)), iron (Fe(III)), and methane (CH_4) (Korom, 1992). The order these reactions proceed is based on the amount of free energy released with denitrification, for example, releasing more energy than sulfate reduction (Metcalf and Eddy, 2003).

The end products of denitrification include dinitrogen gas (N_2), carbon dioxide (CO_2), and bicarbonate (HCO_3^-) (Eq 1.1). The HCO_3^- is of interest because this release of alkalinity increases the solution pH (Korom, 1992; Metcalf and Eddy, 2003). The main product of interest is usually the gaseous phase nitrogen. While the main nitrogenous end product, N_2, is stable due to its molecular triple bonds, denitrification can also produce nitrous oxide (N_2O) (Eq. 1.2), a potentially harmful greenhouse gas (Korom, 1992). The environmental conditions of low pH, low temperature, high solution DO and low carbon to nitrogen ratio (C:N) may shift the final N_2O:N_2 denitrification production ratio towards N_2O (Chapin III et al., 2002). Additionally, the microbiology of the bacteria may be important; denitrifiers which lack the nitrous oxide reductase yet exhibit expression of the other denitrification genes may have reduced denitrification N_2O emissions.

In addition to heterotrophic denitrification, dissimilatory reduction of nitrate to ammonium (DRNA) is also relevant here. This process is similar to denitrification in that they are both reduction reactions involving NO_3^- and are both anaerobic processes (Korom, 1992). However, denitrification leads to nitrogen loss from the aqueous phase while with DRNA, the nitrogen remains in the system. DNRA may occur in anaerobic environments where there is an abundance of carbon relative to NO_3^- (i.e. NO_3^- limited conditions) (Tiedje, 1994).

Enhanced denitrification treatment with solid carbon sources

Relatively recent developments in the field of water remediation have led to advancements with solid carbon source, enhanced denitrification permeable reactive barriers, a novel approach for the treatment of waters containing high concentrations of NO_3^-. The "enhancement" is provided by the added carbon which both encourages aerobic respiration to reduce solution DO so denitrification can proceed and offers a carbon source for denitrifiers (Schipper et al., 2005). This technology has been used for nearly two decades to treat NO_3^- in groundwater, septic effluent and hydroponic waste with findings recently reviewed by Schipper et al. (2010b).

In the first published work in this field, three 200L barrels were filled with mixtures of organic materials and buried in a stream bank 100 m from a tile drainage outlet (Blowes et al., 1994). Influent NO_3^- concentrations were reduced from 2-6 mg NO_3^--N/L to less than 0.02 mg NO_3^--N/L thus validating the potential of organic media to be used to enhance NO_3^- removal. Similar work soon followed with the investigation of passive treatment of septic wastewaters (Robertson and Cherry, 1995); four of these systems had N reductions between 58% to 91% over six years (Robertson et al., 2000). Based on this work, the University of Waterloo trademarked Nitrex™, a reactive flow through barrier for passive, low maintenance septic treatment (Robertson et al., 2005a). Four Nitrex™

systems (9 m^3 to 360 m^3, filled with bark, sawdust and woodchips) were installed in southern Ontario, and with hydraulic retention times of 1 to 10 d, achieved a five year average of 96% removal (Robertson et al., 2005a).

Shortly after Blowes et al. (1994) initial work in Canada, field-scale enhanced denitrification studies began in New Zealand with the installation of a groundwater denitrification wall in 1996 (Schipper and Vojvodic-Vukovic, 1998). Some of the latest research from this group helped identify optimal denitrification fill material (Cameron and Schipper, 2010) and provided insight on treatment of multiple types of waters (Schipper et al., 2010a) and processes within denitrification beds (Warneke et al., 2011a; Warneke et al., 2011b).

Drainage denitrification bioreactors

Though much early work with enhanced denitrification systems focused on groundwater or septic water treatment, the use of enhanced denitrification for reduction of nitrate in drainage waters is now being investigated. Table 1.1 provides a review of drainage denitrification bioreactor performance at multiple scales; additional treatment systems are included for context. In terms of drainage treatment, bioreactors are often compared to wetlands because both provide edge-of-field N load reduction. While both options provide high percent mass load reduction (40% average, greater than 90% at times for both (Christianson et al., Under review), areal NO_3^--N removal rates for bioreactors can be at least an order of magnitude higher than wetlands (Robertson and Merkley, 2009; Van Driel et al., 2006b).

Table 1.1 Review of denitrification treatment for agricultural drainage

Source	Site	Volume (m^3)	Influent NO_3^--N Conc.	Retention Time	Percent Reduction[a]	Nitrate-N Removal Rate	Notes
Field Scale Drainage Treatment Studies							
Blowes et al., 1994	Ontario, Canada	0.2 (barrels)	2 to 6 mg/L	1-6 d	Nearly 100% concentration	NA	Partially buried in a stream bank
Wildman, 2001	South of Chatsworth, IL (#1)	27.2	Approx. 4 to 16 mg/L	NA	Nearly 100% concentration	NA	4.0 ha treatment area
“”	South of Chatsworth, IL (#2)	“”	Approx. 1 to 18 mg/L	NA	Nearly 100% concentration	NA	5.3 ha treatment area
van Driel et al., 2006b	Ontario, Canada; lateral flow	17.2	11.8 mg/L (Mean)	9 h (during tracer test)	32.5% concentration	12 kg N/yr; 2.5 g $N/m^2/d$	Fine and coarse wood media
Jaynes et al., 2008	Central IA	38.9	19.1 to 25.3 mg/L (control plot)	NA	40% - 65% load	0.62 g $N/m^3/d$	Flow-through woodchip wall between crop rows
Moorman et al., 2010	“”	“”	20 to 25 mg/L	24 h required	50%-60% conc. (i.e. to ≤10mg/L)	23.6 mg N/ kg wood/d	Retention time conclusion based on field data
Chun et al., 2010	Decatur, IL (west)	55.8	269.9 g NO_3^--N slug	4.4 h	47% load	NA	2.0 ha treatment area

Table 1.1 Review of denitrification treatment for agricultural drainage (continued)

Verma et al., 2010	“”	“”	Approx. 5 to >20 mg/L	NA	81% - 98% load	NA	“”
Woli et al., 2010	East-Central Illinois (De Land, IL)	76.9	2.8 to 18.9 mg/L	26 min to 2.8 h	23% - 50% load	6.4 g $N/m^3/d$	14 ha treatment area
Verma et al., 2010	“”	“”	Approx. 3 to 16 mg/L	NA	42 % - 48% load	NA	“”
Verma et al., 2010	Decatur, IL (east)	NA	Approx. 4 to 15 mg/L	NA	54% load	NA	6.5 ha treatment area
Ranaivoson et al., 2010	Claremont, MN	NA	11 to 28 mg/L	32 h for 50% conc. reduction	47% - 18% load	NA	52% load reduction in snowmelt; 10.5 ha treatment area
Ranaivoson et al., 2010	Dundas, MN	NA	7 to 14 mg/L	NA	45% - 35% load	NA	7% load reduction in snowmelt
Christianson, 2011	Central IA; Pekin	18	1.2 to 7.8 mg/L (annual mean)	NA	22% - 74% load	0.38-3.78 g $N/m^3/d$	1.2 ha treatment area
Christianson, 2011	Northeast IA, NERF	128	9.0 to 11.3 mg/L (annual mean)	NA	11% - 13% load	0.86-1.56 g $N/m^3/d$	Trapezoidal cross section; 6.9 ha treatment area
Christianson, 2011	Central IA, Greene Co.	127	7.4 to 12.8 mg/L (annual mean)	NA	27% - 33% load	0.41-7.76 g $N/m^3/d$	19 ha treatment area
Christianson, 2011	Central IA, Ham. Co.	102	7.03 to 13.11 mg/L(annual mean)	NA	49% - 57% load	0.42-5.02 g $N/m^3/d$	20.2 ha treatment area
Laboratory and Pilot Scale Drainage Treatment Studies							
Christianson et al., 2011b	Central IA; pilot scale	0.71	10.1 mg/L (Mean)	4 to 8 h	30%-70% concentration	3.8-5.6 g $N/m^3/d$; 1.5-3.4 g $N/m^2/d$	Mixed hardwood chips; different design geometries
Christianson et al., 2011c	Palmerston North, New Zealand; pilot scale	0.53	7.7 to 35.6 (Event means)	1.5 to >15 h	14% - 37% load	2.1 - 6.7 g $N/m^3/d$	*Pinus radiata* chips
Chun et al., 2009	IL; lab column	0.30	10.4 to 33.7 mg/L	2.6 -12.0 h	10%–40% concentration	NA	Three parameter estimation, first order reaction
“”	“”	“”	10.4 mg/L and 25.7 mg/L	15.6 and 19.2 h	100% concentration	NA	
Greenan et al. 2009	Central IA; lab column	0.01	50 mg/L	9.8, 3.7, 2.8, and 2.1 d	100, 64, 52, and 30% load (respective to retentions)	11-15 mg N/kg woodchip/d	
Cooke et al., 2001	IL; lab column	0.001	25 mg/L	8 h	Nearly 100% concentration	NA	Woodchips at 25 ℃
Doheny, 2002	“”	“”	25 mg/L	10 h	60% concentration (i.e. to below 10 mg/L)	NA	Woodchips at 10 ℃
Selected Denitrification Studies (Non-Drainage Specific)							
Schipper and Vojvodic-Vukovic 1998	Cambridge, New Zealand	78.8	5 to 16 mg/L	NA	NA	3.6 g $N/m^3/d$ (max)	Sawdust wall for groundwater denitrification rate
Schipper et al. 2005	“”	“”	NA	5 d	NA	1.4 g $N/m^3/d$	Sawdust wall for groundwater
Robertson and Merkley, 2009	Ontario, Canada	50.0	4.8 mg/L (mean)	NA	78% concentration	220 mg $N/m^2/h$	In-stream reactor
Robertson et al., 2000	Ontario, Canada;	1.9	4.8 mg/L (mean)	3-7 h	58% concentration	5 - 30 mg N/L/d	In ground plywood framed

Table 1.1 Review of denitrification treatment for agricultural drainage (continued)

	North Campus						reactor
Van Driel et al. 2006a	Ontario, Canada; Woodstock reactor	0.73	11.5 mg/L (mean)	1 d	67% concentration	4.5 mg/L/d; 0.7 g $N/m^2/d$	Fine wood media, containerized groundwater treatment
Xue et al. 1999	IL, USA	0.6 and 0.8 ha surface	No detect to 10 mg/L	7 d	19 - 59% load	0.05-0.28 g $N/m^2/d$	Constructed drainage wetland denitrification rate

[a] concentration or load reduction noted

Work from Canada provided one of the first peer-reviewed studies of enhanced denitrification to directly treat drainage water with Van Driel et al. (2006b) investigating a bioreactor consisting of alternating layers of fine and coarse woody material. In the US, Cooke et al. (2001) were the first to explore enhanced denitrification treatments for tile drainage in Illinois. Early work from this group explored carbon media (Cooke et al., 2001), additions of gravel to reduce compaction (Wildman, 2001), and retention time requirements for different media at a range of temperatures (Doheny, 2002). Their most recent field-scale performance results indicate bioreactors can reduce annual NO_3^- loads by 23% to 98% (Verma et al., 2010; Woli et al., 2010). These positive results have led to a number of similar investigations in other tile drained areas of the US.

Initial drainage bioreactor installations in Iowa occurred in 2002 near Pekin, Iowa and in 2006 near Independence, Iowa (Bhandari and Kult, 2010; ISU Extension, 2006). Lately, the Environmental Programs and Services division of the Iowa Soybean Association has been very active in this area by facilitating funding and overseeing installation and management of at least six bioreactors (ISA, 2010). Government officials and programming in Iowa have been involved through the development of an NRCS interim design standard and cost-sharing for denitrifying bioreactors, the first such available funding for enhanced denitrification of tile drainage in the country (Iowa NRCS, 2010). Several laboratory- and pilot-scale studies also hail from Iowa; these have investigated carbon media selection and properties, flow rate and retention time impacts, and design geometry (Christianson et al., 2010a; Christianson et al., 2011d; Christianson et al., 2010b; Christianson et al., 2011b; Greenan et al., 2006; Greenan et al., 2009). Field studies from Iowa and Minnesota have documented performance, longevity, N_2O emissions, and removal of compounds other than NO_3^- (Christianson, 2011; Jaynes et al., 2008; Moorman et al., 2010; Ranaivoson et al., 2010).

Denitrification bioreactor performance factors

Retention time and hydraulics

Bioreactor flow rates combined with the design factors media porosity and bioreactor flow volume dictate the retention time in the reactor. Very low retention times may not be sufficient to reduce the amount of influent drainage DO to a level which would allow denitrification to proceed, whereas very high retention times would provide excellent NO_3^- removal, but also the potential for oxidation reduction (ORP) conditions indicative of undesirable processes like sulfate reduction (Blowes et al., 1994; Robertson and Cherry, 1995; Robertson and Merkley, 2009; Van Driel et al., 2006b) and mercury methylation (Hudson and Cooke, 2011). Much initial work with denitrification systems investigated slow-flowing ground waters or septic effluent; to use the high retention times from these studies (i.e. several days) as the design criteria for drainage treatment would result in an impractically large bioreactor considering these systems are intended to fit in edge-of-field areas to minimize land removed from production.

For drainage denitrification systems, higher retention times generally correlate with higher NO_3^- removal (Table 1.1). For example, Chun et al., (2009) reported NO_3^--N concentration reductions of 10% to 40% at retention times of generally less than five h with 100% removal at retention times of 15.6 and 19.2 h. Greenan et al., (2009) corroborated this, though at a longer time scale, with retention times ranging from 2.1 d to 9.8 d resulting in removal efficiencies of 30% to 100%, respectively. At the field scale, retention time has also been correlated with nitrate removal in Iowa and Illinois (Christianson, 2011; Woli et al., 2010).

In Midwestern bioreactors, the use of inflow and outflow control structures allows closer management of retention times. The inflow structure (i.e. the diversion structure, Chun et al., 2010) routes water into the bioreactor but also allows water to be transmitted via a by-pass line at high flow events (Figure 1.1). The outflow structure (i.e. the capacity control structure, Chun et al., 2010) allows the control of retention time and is thus the structure requiring the most in-field management (Figure 1.1). This capacity control structure can be lowered at low flows (e.g. late summer) to prevent the retention time from becoming too high and it can be raised at higher flow periods (e.g. spring) to maintain a sufficient retention time. Lower cost alternatives to control structures such as moveable pipes have been used in other denitrification systems to control the flow rate, head and/or and retention time (Robertson and Merkley, 2009; Van Driel et al., 2006a).

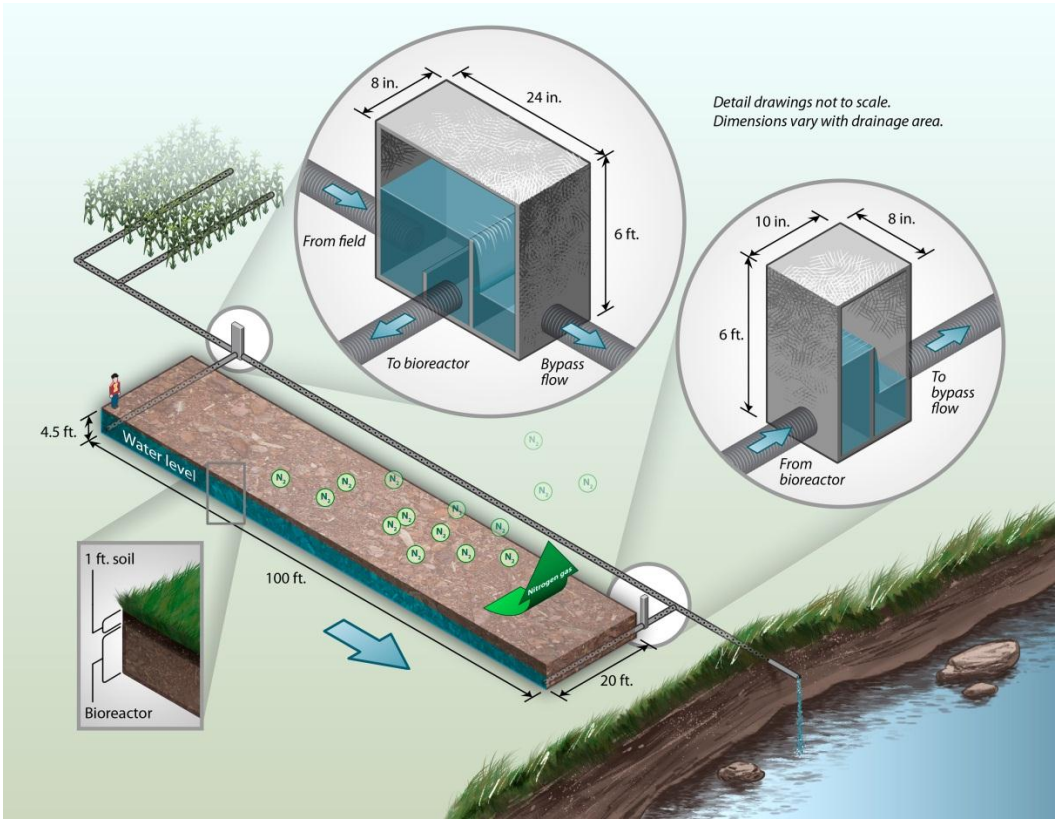

Figure 1.1 Schematic of denitrification bioreactor for agricultural drainage

Hydrologically, many drainage systems experience very low flows or dry periods during an active drainage season. Fortunately, bioreactor start-up once flow resumes after dry periods has not been problematic (Van Driel et al., 2006b). Woli et al. (2010) noted that N removal for several high flow events (i.e. low retention time events) was unexpectedly high likely due to dry periods which immediately preceded each of these events. In general, a drainage event hydrograph advancing through a bioreactor will cause decreased retention times and decreased N removal performance (Christianson et al., 2011b; Christianson et al., 2011c). Additionally, bioreactors experiencing fluctuating flow rates may have decreased performance compared with more steady-state bioreactors even when N removal is compared at the same retention time (Christianson et al., 2011c).

Reaction kinetics

Past analysis of enhanced denitrification systems has not reached a consistent consensus about NO_3^- removal reactor kinetics. Mass removal rates graphed versus inflow NO_3^- concentrations illustrates such kinetics. Zero order reactions occur at a constant rate regardless of reactant

availability; first order reactions occur at an increasing rate proportional to increased concentrations of reactant (Metcalf and Eddy, 2003). Chun et al. (2009) reported NO_3^- removal most closely fit first order removal parameters in laboratory experiments using inflow concentrations of less than 35 mg NO_3^--N/L common in agricultural drainage. Similarly, Leverenz et al. (2010) found that first order removal best fit their pilot-scale woodchip/wetland system, though they noted that "…while most field-scale systems are well approximated assuming zero order reaction kinetics, at low nitrate concentrations and at reduced temperatures, first-order kinetics may provide a better fit." Gibert et al. (2008) reported zero order NO_3^- removal for a batch test using 32 mg NO_3^--N/L and Van Driel et al. (2006b) assumed zero order kinetics for their field reactor receiving a maximum of 20 mg NO_3^--N/L. Robertson (2010) recently documented increasing NO_3^- concentrations from 3.1 to 49 mg NO_3^--N/L in lab experiments yielded a zero order reaction which corroborated their previous field-scale assumption (Robertson et al., 2000). A review by Schipper et al. (2010b) reported the design of these systems could functionally use zero order kinetics; however, earlier work by this group showed increased daily average denitrification rate strongly correlated with increased daily average NO_3^- concentrations of less than 20 mg NO_3^--N/L (Schipper et al., 2005).

Temperature

Drainage water entering a bioreactor may have temperatures that vary seasonally with early spring temperatures just above freezing and late summer temperatures at greater than 15°C (Christianson, 2011). As a biologically mediated transformation, denitrification in a bioreactor is influenced by the drainage water temperature though NO_3^- removal has been documented at water temperatures as low as 2°C to 4°C (Robertson and Merkley, 2009). Not surprisingly, many studies show increased NO_3^- removal at higher temperatures (Cameron and Schipper, 2010; Diaz et al., 2003; Volokita et al., 1996). Reported and calculated values of Q_{10}, or the factor by which the reaction rate increases for every 10°C rise in temperature, for these systems has ranged from less than 1 to nearly 3 with most values around 2 (approximately ± 0.5) (Cameron and Schipper, 2010; Robertson and Merkley, 2009; Van Driel et al., 2006b; Warneke et al., 2011a). In other temperature investigations, Cooke et al. (2001) used the Van't Hast-Arrhenius Law to show increased retention times were required at lower temperatures.

Maximum drainage temperatures in summer will precipitate enhanced bioreactor NO_3^- removal, meaning that annually bioreactor treatment is not optimized as the highest drainage N loads occur in the spring (Mirek, 1999; Patni et al., 1996). The effect of temperature on bioreactor performance is known to be significant (Christianson, 2011), but with better understanding of

operational parameters like seasonal retention time management, it is possible this sensitivity to temperature can be reduced. Additionally, NO_3^- concentrations in early season drainage may be relatively lower due to snow melt (Patni et al., 1996), potentially also complicating reaction kinetics. At low temperatures, it is recommended to manage for a longer retention time (Robertson et al., 2005a; Volokita et al., 1996); such control structure management would likely be done in the spring in the Midwest regardless to address the higher spring flow rates.

Microbiology

Little peer-reviewed work has been done on drainage denitrification bioreactor microbiology. Because denitrifiers are abundant in the environment, no inoculation has been required for these systems to date (Schipper et al., 2010b) other than the addition of soil, typically in small amounts (i.e. one kg from Blowes et al. (1994) or 1 L from Christianson et al. (2011c)). However, slow bioreactor start-up after an early spring installation has been attributed to the slow growing microbial community (Wildman, 2001).

It is thought that these denitrifiers are the primary denitrification vehicle but fungi may also provide an important enhancement due to its ability to release soluble carbon substrates (Appleford et al., 2008). Appleford et al. (2008) reported that denitrifiers were present on both the woodchip surface and in the bioreactor solution. Denitrification sites may not be limited to the chip surface though, as Robertson et al. (2000) found dark coloration extended several mm into the wood particles indicating that water infused into the wood may also be denitrified (Robertson et al., 2005b). Andrus et al. (2010) used DNA techniques to document that microbial communities varied within a bioreactor with denitrifiers commonly present throughout, and Moorman et al. (2010) reported that woody media walls supported higher levels of denitrifiers than the surrounding soil. Most recently, Warneke et al. (2011b) documented the bacterial community in a small-scale bioreactor containing woody media contained a higher percentage of denitrifiers than the community in a maize cob bioreactor indicating there was a potential for more carbon to be utilized by non-denitrifiers in the maize reactor.

Bacteria other than denitrifiers have been documented at bioreactor sites mainly by the presence of biofilms (Chun et al., 2009). These biofilms may cause clogging in the lines or control structures and flushing (via stop log control, if possible) or agitation may be the best management option (Christianson et al., 2011c; Van Driel et al., 2006b; Wildman, 2001). Conversely, there may be problems with denitrifier wash-out at high flow rates (Volokita et al., 1996), though this has never been documented in the field.

Denitrification bioreactor design

One of the largest design and performance challenges of drainage denitrification systems is the variable, and oftentimes unknown, flow rates inherent to drainage systems (Christianson et al., 2009; Woli et al., 2010). A peak flow rate could be estimated for a given drainage system by multiplying a drainage coefficient by the drainage area (e.g. 1.3 cm/d coefficient for a 16 ha site yields 24 L/s) or by using a pipe-full flow equation (e.g. Manning's equation), but drainage systems vary rarely operate at this maximum flow rate. Flow rates within a given year range from zero to this maximum (or above, as this is theoretical) with low and higher flow periods interspersed depending upon precipitation patterns.

A recent design method by Christianson et al. (2011a) attempted to account for flow rate and retention time by estimating a peak flow rate for the drainage system and sizing the bioreactor to treat a percentage of that maximum at a chosen retention time. This downsizing of the peak estimated flow rate is in agreement with reports that designing a bioreactor to treat the peak drainage flow rate may not be economical (Van Driel et al., 2006b). In Iowa, a very similar design method is used by the USDA NRCS to design bioreactors that are seeking cost-share through the Environmental Quality Incentives Program (EQIP) (Iowa NRCS, 2010). An alternative design concept from Illinois consists of correlating bioreactor surface area (i.e. aerial footprint, L x W) and treatment area on an efficacy curve; for example, approximately 9.3 m^2 of bioreactor surface area would be required for every 1.4 ha of drainage area (100 ft^2 per 3.5 ac) in order to achieve load reductions of 60% (Verma et al., 2010). A design table from Wildman (2001) allows estimation of a required bioreactor volume based on drainage area and drainage coefficient; unfortunately, the drainage area and coefficient are not known for many drainage systems. Another method from the Midwest has used the rough estimate of approximately 3 m of bioreactor length for every 0.4 ha of drainage. Finally, the stoichiometry of the denitrification reaction (Eq. 1.1) can be used to develop a volume of carbon required, but this theoretical method may be prone to error as many other microbial reactions will also utilize the carbon (Wildman, 2001).

In addition to different design methods, alternative configurations for drainage denitrification systems have been investigated. Jaynes et al. (2008) used a hybrid approach of denitrification walls on the sides of a tile line as a passive, flow-through technology. In-stream bioreactors have also been installed in drainage ditches with a system by Robertson and Merkley (2009) consisting of 40 m^3 woodchips plus a gravel infiltration gallery, a downstream silt impedance layer and a downstream berm. Different bioreactor design geometries have been explored though there may be no significant

benefit of different shaped cross-sections at least at the pilot scale (Christianson et al., 2010b). Among researchers and collaborators, there have been discussions of including baffles within a bioreactor or designing bioreactors in series or parallel (Cooke et al., 2001) to maximize treatment. While these ideas are certainly interesting in the research realm, all such ideas must eventually be tempered under the umbrella of farm-scale practicality. The use of a denitrification bioreactor as part of a "suite of solutions" for drainage is also an idea worth consideration (Christianson and Tyndall, 2011); bioreactors can easily be paired with wetlands (Robertson and Merkley, 2009), controlled drainage (Woli et al., 2010), and other in-field conservation practices for improved water quality.

Installation Considerations

Several issues surrounding the installation of a drainage bioreactor include pre-installation site evaluations, component availability, and construction details (Table 1.2).

Table 1.2 General factors to be considered for denitrification bioreactor installation

Pre-Design		Materials Availability	Construction
Drainage Characteristics	**Site Conditions**		
Drainage area	Available space for the bioreactor	Inflow control structure (3 chamber)	Uniform and consistent filling of media
Tile locations	Soil type	Outflow control structure (2 chamber)	Use of a liner
Tile size	Proximity to sensitive or public water bodies	Suitable fill media	Mounding soil cover
Tile slope	Equipment traffic-ability	Non-perforated pipe near the structures	Reseeding with appropriate seed mixture
Drainage coefficient		Construction labor and equipment	
Number of surface intakes (minimize sediment)		Labor for annual maintenance	
		Cover seed	

Installation generally consists of positioning the control structures, excavating and filling the trench, laying geo-fabric over the fill, mounding the soil cover, and re-seeding the site (Sutphin and Kult, 2010). Woli et al. (2010) recommended using a bioreactor liner after documenting a lack of outflow from one of their unlined bioreactors; Doheny (2002) also suggested the use of a liner for sandy areas and many installations to date have been lined (ISA, 2011; Van Driel et al., 2006b). This highlights the importance of site evaluations which carefully consider potential designs for sites with highly permeable soils. A mounded soil cover is sometimes used to help prevent subsidence as the woodchips can settle (Schipper and Vojvodic-Vukovic, 1998). In addition to subsidence concerns, a soil cover may be beneficial for mitigating N_2O emitted through the bioreactor surface. Christianson et al. (In preparation-a) found N_2O emissions from the soil cover of pilot bioreactors were lower than

emissions directly from the surface of the woodchips. Similar, but non-significant, results were documented at a non-soil-covered and a soil-coverd bioreactor in Illinois (Woli et al., 2010).

Carbon Media

The type of carbon fill is one of the most important considerations of denitrification systems because media properties affect many vital factors ranging from retention time to longevity to start-up flushing. Robertson et al. (2005a) noted the selection of denitrification fill material should be based upon cost, porosity, C:N ratio, and longevity. These requirements mean that a wide variety of materials may be most practical in different locations with tested materials including corn cobs, corn stalks, wood media (multiple sizes and species), wheat and barley straw, and pine and almond shells (Cameron and Schipper, 2010; Christianson et al., In preparation-b; Diaz et al., 2003; Greenan et al., 2006; Hashemi et al., 2010; Soares and Abeliovich, 1998). In general, woody media is the preferred fill material due to cost, conductivity, longevity, and C:N (Schipper et al., 2010b).

The chemical properties of the media can most notably affect longevity and organic flushing. While woody media is the recommended material, there can be a wide variety of C:Ns between tree species with lower C:N materials generally not recommended due to flushing losses or potential mass degradation (Christianson et al., In preparation-b; Christianson et al., 2011d; Gibert et al., 2008). Some authors have discussed the use of hardwood versus softwood, but this terminology may be misleading. For example, two species used successfully in denitrification studies are oak, a hardwood (Greenan et al., 2006; Jaynes et al., 2008) and pine, a softwood (Cameron and Schipper, 2010; Schipper and Vojvodic-Vukovic, 1998), which both have a C:N in the range of several hundred (Greenan et al., 2006; McLaughlan and Al-Mashaqbeh, 2009).

In addition to chemical properties, the physical properties of the fill material (i.e. porosity, particle size, and hydraulic conductivity) are also important and can change over time. Porosities of woody chipped media typically range from 0.6-0.86 (Christianson et al., 2010a; Chun et al., 2009; Ima and Mann, 2007; Robertson, 2010) with *in situ* values reported at 0.65 to 0.79 (Chun et al., 2010; Van Driel et al., 2006b; Woli et al., 2010). Increased moisture content (Ima and Mann, 2007) and packing density (Christianson et al., 2010a) both decrease woodchip porosity.

There can be a large range in particle sizes and shapes for the term "chip". Commonly, chipped material described by Christianson et al. (2010a) had 50% of particles which fell between 13 to 25 mm and Chun et al. (2009) and Woli et al. (2010) used chips of which had 66% and 62%, respectively, fall in the 6 to 25 mm range. Several studies reported no consistent, significant differences in NO_3^- removal for coarse versus fine or ground materials (Greenan et al., 2006; Van

Driel et al., 2006b) and have recommended coarse materials for preferable flow properties (Van Driel et al., 2006b). Additionally, at higher flow rates, fine materials may be washed out thus modifying porosity and hydraulic conductivity (Chun et al., 2009). The addition of gravel to woodchip media may help reduce compaction-related porosity reduction (Wildman, 2001), but it may be difficult to obtain a homogenous mixture at the field-scale.

Considering the relatively high flow rates a drainage denitrification bioreactor may experience, the hydraulic conductivity of the media is one of its most important physical parameters. Even in groundwater treatment this parameter is important as Schipper et al. (2004) reported incorrect estimation of conductivity led to preferential flow around a denitrification wall. Average conductivities for wood material have ranged from 0.35 cm/sec (sawdust) to 11.6 cm/sec (61 mm chips) (Cameron and Schipper, 2010) with the Christianson et al. (2011a) design method based on an average of 9.5 cm/sec though this user input can be changed (Christianson et al., 2010a). *In situ* values range from 1.2 cm/sec to 11 cm/sec (Robertson et al., 2005a; Van Driel et al., 2006b). Over time, the conductivities can decrease with possible explanations of biofilm formation (Chun et al., 2009; Robertson and Merkley, 2009) or consolidation; reactors containing larger particles may experience relatively higher reductions in conductivity compared to reactors with smaller particles (Cameron and Schipper, 2010).

Longevity

Bioreactor longevity depends upon multiple factors including the type of carbon source, flow characteristics, and the consistency and level of saturation. Blowes et al. (2000) noted the lives of these systems are finite and will also depend upon mass of reactive material, the reaction rate, and physical changes in the barrier (porosity and permeability). Further factors affecting longevity include other microbial processes like sulfate reduction (Blowes et al., 2000) and, to a small extent, dissolved organic carbon leaching (Robertson and Cherry, 1995).

Most of the longevity estimates to date have been for groundwater treatment systems or tile drainage walls. Based upon stoichiometry, half-lives, or carbon losses, these estimates approximate lives greater than several decades (i.e. 20, 37, 66 or 72 yr from Robertson and Cherry (1995), Moorman et al. (2010), Long et al. (2011) and Blowes et al. (1994), respectively), though denitrification may be carbon limited by then. Several of reports have shown there is very little carbon deterioration of consistently saturated wood media in the first years of operation (e.g. less than 13% C loss in saturated chips over nine yr by Moorman et al. (2010); negligible C loss over five yr after initial losses by Schipper and Vojvodic-Vukovic (2001); little C deterioration after four to six yr

by Robertson et al. (2000)). At this point, the longevity of drainage denitrification systems (and denitrification systems in general) is not exactly known because seemingly none have failed due to carbon exhaustion (Schipper et al., 2010b). However, the longevity of denitrification bioreactors is very important as these systems will likely not be economically feasible if the carbon source has to be frequently replaced (Robertson et al., 2000).

Cost

The total cost of denitrification bioreactors in Iowa has ranged from $4,400 to $11,800 to treat a range of drainage areas (12 to over 40 ha) (Sutphin and Kult, 2010). Other reported costs have been on the order of $2000 (Canadian dollars) and $3200 from Van Driel et al. (2006b) and UMN Extension and MN Department of Ag. (2011), respectively. Schipper et al. (2010b) provided the first cost efficiency calculation of a denitrification system at $2.39 per kg N to $15.17 per kg N. This was higher than a newer cost report of $1.44 per kg N ± $0.92 from Christianson et al. (Under review)(including government cost-share). In this recent analysis, Christianson et al. (Under review) found that modifications of N fertilizer application had the potential to provide a cost savings to the producer and that constructed practices (bioreactors, wetlands, and controlled drainage) all cost less than $2.00 per kg N removed.

Concerns and limitations

Several major concerns about denitrification systems include start-up issues, greenhouse gas production, and mercury methylation (Schipper et al., 2010b). Because this is still a relatively new technology for treatment of drainage, many of these concerns are still being investigated.

Organic flushing

The flushing of organics upon woody reactor start-up has been noted in many studies with flushing parameters of water quality concern including TOC/DOC, BOD, NH_4^+, and TKN (Cameron and Schipper, 2010; Gibert et al., 2008; McLaughlan and Al-Mashaqbeh, 2009; Schipper and Vojvodic-Vukovic, 1998). Initially dark or tea colored effluent means early concentrations for several of these parameters can be in the hundreds of milligrams per liter (Schipper et al., 2010b), though these concentrations stabilize at lower rates over time (Robertson et al., 2005a). McLaughlan and Al-Mashaqbeh (2009) suggested the release of DOC was “multi-phasic”; releases may occur in the short term as well as long term as new pieces are dissolved or degraded. To reduce this effect, laboratory-scale studies have used pre-flushed materials (Diaz et al., 2003), but at the field-scale this

may be logistically difficult (Schipper et al., 2010b). In an applied agricultural sense, organic flushing may never be eliminated from denitrification bioreactors, but can be minimized through selection of more optimal material (Christianson et al., In preparation-b) or starting-up under high flow conditions when by-pass flow is also occurring (ISA, 2011; Schipper et al., 2010b). Site selection is also important as careful consideration should be given to proposed bioreactors outleting to sensitive or public waters.

Nitrous oxide

As a natural by-product of denitrification (Eq. 1.2), nitrous oxide may be released from a denitrification bioreactor either from the surface or in the dissolved form in the liquid effluent. Of the environmental conditions that may shift the N_2O:N_2 denitrification production ratio towards N_2O (i.e. low pH, temperature, or C:N and high DO, Chapin III et al., 2002), high DO may be of special concern for drainage bioreactors considering their inherently fluctuating flow rates and possible flow depths. Christianson et al. (In preparation-a) investigated lab-scale bioreactor N_2O emissions under fluctuating flow conditions and found small spikes of N_2O were released when water levels dropped in the reactors, but overall the N_2O released was never more than 1% of the influent NO_3^-. This corroborated multiple field-scale reports that total N_2O emitted is less than 4.5% of the N removed (Elgood et al., 2010; Moorman et al., 2010; Warneke et al., 2011a; Woli et al., 2010).

As an N_2O mitigation technique, Elgood et al. (2010) suggested designing systems for complete NO_3^- removal, but this may exacerbate sulfate reduction and mercury methylation (see next section). The use of a soil cover may help mitigate N_2O surface emissions (Christianson et al., In preparation-a). In terms of the total nitrogen balance over a watershed, Moorman et al. (2010) noted that if NO_3^- in drainage is treated less efficiently downstream, more N_2O may be released than if it is treated in a bioreactor.

Sulfate reduction and mercury methylation

The process of sulfate (SO_4^{2-}) reduction has been documented in many denitrification systems at low flows when NO_3^- has been removed nearly completely and often at high temperatures (Blowes et al., 1994; Robertson and Cherry, 1995; Robertson and Merkley, 2009; Van Driel et al., 2006b). Thermodynamically, the use of SO_4^{2-} as an electron acceptor is not as favorable as NO_3^-, but when NO_3^- is reduced completely in a bioreactor, SO_4^{2-} reducing organisms can out-compete denitrifiers for carbon and convert naturally present SO_4^{2-} to hydrogen sulfide gas (H_2S). Sulfate reduction occurs at oxidation reduction potentials (ORPs) less than ORPs for denitrification when there is an excess of

reducing capacity (Blowes et al., 1994). This process is of concern because (1) its represents a loss of carbon for the denitrifiers (2) the production of hydrogen sulfide can be a noxious gas (though bioreactors are not in confined spaces), and (3) this process is closely linked to the methylation of mercury.

Common forms of mercury include elemental mercury, inorganic mercury, and methyl mercury (CH_3Hg) (Krabbenhoft and Rickert, 2009; USGS, 2000b), with methyl mercury forming from inorganic mercury via the biological processes of sulfate reducing bacteria (Eckley and Hintelmann, 2006; Krabbenhoft and Rickert, 2009). In most locations, atmospheric deposition is the biggest source of mercury (Krabbenhoft and Rickert, 2009) and watersheds with wetlands have been identified as important sources (net producers) of methyl mercury (Rudd, 1995). Human intake of mercury occurs typically by ingestion of bio-accumulated methyl mercury in fish or inhalation of elemental mercury (Krabbenhoft and Rickert, 2009).

There has been little work investigating mercury methylation in enhanced denitrification systems with the only direct evidence of this process reported by Hudson and Cooke (2011). To reduce this concern, bioreactors should be designed and managed to minimize sulfate reduction by perhaps retaining very low concentrations of NO_3^- in the effluent (Robertson and Merkley, 2009); if hydrogen sulfide (i.e. a rotten egg smell) is noted around the outflow control structure, the stop logs should be lowered to allow water to flow unimpeded through the reactor. Moreover, as methyl mercury can be detoxified if exposed to sunlight (especially UV light) (USGS, 2000b), day lighting outflow to the surface may be a good design option. Because this is potentially a very inflammatory issue, there is much interest in evaluating the potential of denitrification bioreactors for mercury methylation and this is an area of ongoing and future research for several groups.

Monitoring methods

As denitrification bioreactors for the treatment of agricultural drainage continue to move from the research to the demonstration phase, one of the most important considerations is the availability of practical, field-scale monitoring methods. Many researchers have used techniques such as denitrifying enzyme activity (DEA), stable isotopes (^{15}N), and gas monitoring to better understand the denitrification process and estimate the nitrogen balance in these systems (Elgood et al., 2010; Greenan et al., 2006; Long et al., 2011; Moorman et al., 2010; Schipper and Vojvodic-Vukovic, 1998; Warneke et al., 2011a). While these methods provide interesting and valuable research data, it is unlikely such methods will be used to monitor farmer-managed bioreactors, and thus a description of simpler methods is useful. For the simplest representation of drainage bioreactor function,

comparison of inflow and outflow NO_3^- concentrations based on grab sampling is the most basic level of monitoring recommended. Although this method is easiest, without supporting evidence provided by some of the relatively straightforward monitoring techniques described below, very little is known about the bioreactor and many questions are left in terms of performance.

Sampling

Grab sampling from the inflow and outflow structures is the most rudimentary level of monitoring recommended for these systems. Water samples can easily be collected with a sampling rod (i.e. a stick with a sample collector attached to the end) at the overflow point of the stop logs in both structures. For research purposes, Rodrigue (2004) investigated the required frequency of bioreactor sampling and recommended sampling every 4 d though this may be more intensive than is practical at demonstration sites. A number of researchers have sampled weekly to every other month for common parameters (e.g. NO_3^-), while also having some samples analyzed less frequently for other compounds (e.g. BOD, TKN, NH_4^+, DOC, SO_4^{2-}) that are of research interest but may not directly pertain to NO_3^- removal performance (Blowes et al., 1994; Robertson and Merkley, 2009; Van Driel et al., 2006b). Quality assurance and control requirements for other substances like volatile organic compounds or mercury necessitate special methods which may be less appropriate for demonstration site monitoring (Hudson and Cooke, 2011; Robertson and Merkley, 2009). Regarding timing of sample collection, Van Driel et al. (2006b) did not collect samples within 48 h of a rainfall event to avoid diluted samples, and similarly, Woli et al. (2010) did not collect samples during two high drain flow events under the assumption no NO_3^- removal would occur.

In order to capture higher resolution data, an autosampler (e.g. Teledyne Isco 6712 Portable Sampler) can be used to collect samples from a structure over the course of a storm event or during a tracer test (see below). Another variation on grab sampling is that sampling can be tied to flow measurement to obtain flow proportional samples; Jaynes et al. (2008) used this method to obtain weekly composited samples from collection sumps where flow volumes were also recorded versus time.

Flow measurement

While sampling for NO_3^- concentrations provides some insight on NO_3^- removal, the ability to relate concentration data to flow volumes allows a more complete performance description in terms of NO_3^- loads. The most elementary flow monitoring method utilizes a container of known volume and a stop watch and can be done at bioreactor sites that outlet directly to a surface water body.

Unfortunately, this method can be prone to error and variability, though several authors have published results with this method (Robertson and Merkley, 2009; Van Driel et al., 2006a; Van Driel et al., 2006b).

The next least expensive method is the use of pressure transducers (and optional data loggers) to record water depth in the control structures (e.g. Solinst Model 3001 Levelogger Junior, Global Water Instrumentation, Inc. WL16 Water Level Loggers). Limitations here are that the transducers give no indication of water movement meaning standing water or backwards flow in the structure is problematic for flow calculations. Chun and Cooke (2008) developed weir calibration equations for AgriDrain control structures that are commonly used in bioreactor designs. The installation of a v-notch weir in the structure can give increased accuracy for flow calculations especially at low flow depths (Christianson, 2011; Woli et al., 2010). Other more expensive flow monitoring methods include the use of doppler-based velocity meters (e.g. Teledyne Isco 2150 Area Velocity Module, MACE Series 3 FloPro) or digital or mechanical totalizing flow meters with data loggers (Jaynes et al., 2008). Other non-drainage denitrification treatments have used flow monitoring equipment such as mechanical water meters, inline sonic flow meters, and impellor water meters (Schipper et al., 2010a; Warneke et al., 2011a).

In situ measurements

Additional information provided by measurement of parameters such as DO, temperature, pH and ORP is relatively easy to obtain with measurement probes; the inflow and outflow structures provide ideal locations to deploy such probes to below the water level either permanently or for a spot reading during a site visit. Temperature and pH meters and probes are especially common laboratory equipment (e.g. WTW 3300i pH field meter) and provide interesting information as temperature impacts the microbiology of denitrification and pH is typically increased by this process (Warneke et al., 2011a). Media bags are another useful *in situ* research tool for investigating longevity and carbon dynamics (Christianson et al., 2011d; Moorman et al., 2010).

As an anoxic process, DO measurements can indicate if the conditions are present for denitrification to proceed. At several sites, DO has been shown to be reduced to below 0.5 mg DO/L within approximately 25% of the length from the inlet (Christianson et al., 2011b; Van Driel et al., 2006b; Warneke et al., 2011a). In smaller scale and laboratory studies, DO is difficult to control as the influent water may be easily disturbed (Chun et al., 2010; Schipper et al., 2010b).

The use of an ORP probe (AKA redox or oxidation reduction probe; e.g. WTW SenTix ORP Electrode Probe) provides slightly more insight than DO measurements into conditions conducive to

denitrification (Blowes et al., 1994; Christianson et al., 2011b; Van Driel et al., 2006b). Because the use of different electron acceptors (i.e. oxygen, nitrate, sulfate, etc.) varies based on the strength of the reducing conditions, ORP measurements have been used as supporting data for occurrence of these various reactions. This parameter may be reported as an ORP, which is often relative to a Ag/AgCl electrode, or as an Eh, which is the voltage reading relative to a standard hydrogen electrode; the offset between the two depends upon the reference electrode used but is usually around 200mV (YSI Environmental, 2001). The range most suitable for denitrification is +50 to -50mV (ORP) with the Eh upper limit of approximately 350mV (Gilliam et al., 1979; YSI Environmental, 2008).

Tracer testing

Tracer tests are commonly used in reactor engineering to investigate hydraulic performance and residence characteristics. In plug flow reactors, non-ideal hydraulic conditions include short-circuiting, where a certain volume of flow arrives at the outlet of the reactor early, or dead zones, where a certain volume of flow becomes trapped in the reactor. These conditions can be caused by poor mixing and poor design (Metcalf and Eddy, 2003).

In denitrification systems, bromide or chloride are typically used as conservative tracer compounds to better study hydraulic properties and flow characteristics (Christianson et al., 2011b; Christianson et al., 2011c; Schipper et al., 2005; Schipper et al., 2004; Van Driel et al., 2006b). In drainage bioreactors, it is suggested that by-pass flow be avoided during the tracer test (Chun et al., 2010). Tracer testing from Chun et al. (2010) was used to more closely model hydraulic parameters while Christianson et al. (2011b) and Christianson et al. (2011c) did tracer tests as part of pilot scale testing to determine if theoretical retention times differed from *in situ* tracer residence times. Similar tracer tests were done at the field scale to also determine tracer residence time and evaluate design methodology criteria; these tests indicated plug flow characteristics, but contrary to pilot testing, showed tracer residence times were lower than theoretical retention times (Christianson, In preparation-c).

Wells and piezometers

The installation of wells or piezometers in a bioreactor is useful for sampling to determine where NO_3^- removal or other processes are occurring and provides ideal *in situ* locations for probe measurements (e.g. temperature, electrical conductivity, pH, Eh, DO) (Christianson, In preparation-c; Van Driel et al., 2006b; Warneke et al., 2011a). Samples are usually collected from the piezometers

via a pump or syringe with well evacuation or purging recommend prior to collection (Van Driel et al., 2006b). Pressure transducers can likewise be fitted in the wells to determine the head difference across the reactor and to document drainage hydrographs moving through (Christianson, In preparation-c; Chun et al., 2010). Depth to water can also be manually measured in such wells with the use of a measuring tape (e.g. Solinst Model 101 or 102 Water Level indicators) (Christianson et al., 2011b). Installation of "bundles" of piezometers with each individual piezometer screened at a different depth allows measurement and sampling of the Z axis of the reactor as well as along the 2D surface (Van Driel et al., 2006a; Van Driel et al., 2006b).

Conclusions

Enhanced denitrification systems to reduce NO_3^- loadings from agricultural drainage systems are a promising new technology. However, this new water quality option is not without limitations or additional research needs. As this technology begins to move from the research to demonstration phase, more field scale bioreactor data are urgently needed to evaluate design methods, quantify potential deleterious effects, and develop better management methods for optimized performance. It is hoped this practice-oriented document can help landowners and professionals in the field better understand, manage, and monitor their denitrification bioreactors for agricultural drainage.

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Dissertation Organization

The overall goal of this dissertation is to investigate the design, performance, and cost-efficiency of denitrification bioreactors for agricultural drainage. This Ph. D. work is the culmination of 3.5 yr of work both in Iowa, USA and in Palmerston North, New Zealand. The four main chapters of this dissertation are each articles which are undergoing various stages of the peer review process. In short, pilot-scale investigations of bioreactor performance (Chapters 2 and 3) influenced the design and field-scale evaluation of four bioreactors in Iowa (Chapter 4), and as interest in denitrification bioreactors grew over this Ph. D. period, a comprehensive cost comparison of bioreactors and other water quality improvement options became necessary (Chapter 5).

Chapter one is a comprehensive literature review coving the design, installation, performance, and monitoring of agricultural drainage bioreactors. It was written with a practice-oriented focus so it could easily be condensed and submitted as a review article for the journal Applied Engineering in Agriculture. As denitrification systems move from the research to the demonstration phase in the US Midwest, this document will be helpful to those working in the field to better understand, design and monitor these systems.

Chapter two describes pilot-scale work with denitrification bioreactors of three different design geometries from Iowa, USA. This work investigating design hydraulics and retention times has been published in the Journal of Environmental Engineering with coauthors of Alok Bhandari, chair of Civil Engineering, Kansas State University and Matthew Helmers, Associate Professor of Agricultural and Biosystems Engineering, Iowa State University.

Chapter three is a continuation of pilot-scale work but with new applications in New Zealand. Here denitrification treatment of New Zealand's mole and tile drainage was investigated under the premise that containment of drainage waters prior to bioreactor treatment helps optimize nitrate removal. This work has been published in the journal of Agricultural Water Management with coauthors James Hanly, Research Officer, Fertilizer and Lime Research Centre, Massey University, Palmerston North, NZ, and Mike Hedley, Professor and Group Leader, Soil and Earth Sciences, Massey University, Private Bag 11 222, Palmerston North, NZ.

A field-scale evaluation of four bioreactors in Iowa is described in chapter four. This work was done in collaboration with the Iowa Soybean Association (ISA); two of the four bioreactors described were installed and monitored by the Environmental Programs and Services division of the ISA. This work is in preparation for the Journal of Environmental Quality. Coauthors include Matthew Helmers, Associate Professor of Agricultural and Biosystems Engineering, Iowa State University, Alok Bhandari, chair of Civil Engineering, Kansas State University, Roger Wolf, Iowa Soybean Association, Keegan Kult, Iowa Soybean Association, and Todd Sutphin, Iowa Soybean Association.

The fifth chapter attempts to put denitrification bioreactors in context of other drainage water quality practices through a comprehensive cost evaluation. Costs for seven water quality options were itemized and used to develop comparison parameters of cost efficiency in terms of $ per kg N removed. This work is in preparation to be submitted to the Journal of Environmental Quality with coauthors of John Tyndall, Assistant Professor in Natural Resources Ecology and Management, Iowa State University and Matthew Helmers, Associate Professor of Agricultural and Biosystems Engineering, Iowa State University.

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CHAPTER 2 PILOT-SCALE EVALUATION OF DENITRIFICATION DRAINAGE BIOREACTORS: REACTOR GEOMETRY AND PERFORMANCE

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Abstract

Denitrification drainage bioreactors are emerging as an innovative practice to address water quality concerns stemming from nitrate leaching from drained agricultural lands. Although installation of these systems has begun in farms in the Midwestern United States, the understanding of their design and in-field performance remains deficient. This study utilized a set of pilot-scale drainage bioreactors to evaluate the impact of bioreactor geometry on reactor hydraulic properties and to determine nitrate removal under steady state conditions and during a simulated storm event. Bioreactors with different cross-sectional geometries but similar depths and total volumes were evaluated. Percent reduction of the influent nitrate mass was linearly correlated to the theoretical hydraulic retention time (HRT) with 30 to 70% NO_3^--N removals observed within the four to eight h of retention time suggested for field installations. Tracer tests revealed that in-situ HRTs were substantially greater (i.e. at least 1.5 times as large) than theoretical HRTs.

Keywords: denitrification; nitrate; drainage; water treatment; bioreactor

Introduction

The annual occurrence of a large hypoxic zone in the northern Gulf of Mexico has been identified by the U.S. Environmental Protection Agency (EPA) as a key water quality issue related to upland practices in the United States. The expanse of this zone was the largest on record in 2002 measuring 22,900 km^2 and the second largest in 2008 measuring 20,500 km^2 (LUMCON, 2010). Nitrate loadings in the Mississippi River have been identified as a key cause of this water quality impairment and these loadings stem in large part from agricultural drainage in the US Midwest (USGS, 2000). The EPA Science Advisory Board's 2008 Gulf Hypoxia Action Plan has outlined a national strategy to improve water quality in the Mississippi River Basin and restore water quality in the Northern Gulf of Mexico (USEPA, 2007). The report calls for a minimum 45% reduction in the total nitrogen load to the Gulf from the Mississippi River.

Throughout the upper Mississippi and Ohio River basins poorly drained soils are artificially drained to improve overall crop productions (Zucker and Brown, 1998; Lawlor et al., 2008). The subsurface drainage systems, however, create a short-circuit in the natural hydrologic pathway, resulting in faster transport of water and soluble contaminants, such as nitrate, from farms to surface waters (Kellman, 2005). The concentration of nitrate-N in agricultural drainage often exceeds the drinking water standard of 10 mg NO_3^--N /L with annual loadings from farms over 66 kg NO_3^--N/ha (Jaynes et al., 1999; Kalita et al., 2006). Regardless of improved in-field management strategies, nitrate concentrations in drainage waters can still exceed the drinking water standard because the timing of precipitation and natural mineralization of soil organic matter can also affect nitrate leaching from the drained soils (Randall and Goss, 2001; Kladivko et al., 2004).

Denitrification drainage bioreactors have attracted recent interest among farmers and watershed protection groups in the upper Midwest for their potential to reduce nitrate in drainage. In the past, similar systems have been used to treat nitrate in groundwater and in septic system effluent (Robertson and Cherry, 1995; Schipper and Vojvodic-Vukovic, 2000). This technology has good potential for application to treat nitrate-containing subsurface drainage from agricultural fields before the water flows into a surface stream.

Since drainage bioreactors are targeted for use by farm operators, their design criteria have to include cost-effectiveness and minimal operation and maintenance. The reactors are, therefore, designed as passive systems consisting of a simple excavation filled with an electron donating carbonaceous material, commonly woodchips, through which drainage is routed. Reductions in nitrate-N concentrations in such reactors can be moderate to high (50 to 100%) with daily removal rates ranging from 0.6 to 2.3 g NO_3^--N/m3 of total reactor volume (Van Driel et al., 2006; Appleford et al., 2008; Jaynes et al., 2008). These rates can be an order of magnitude higher than those typically associated with wetlands (Van Driel et al., 2006).

Drainage bioreactors are installed to intercept the flow in drainage mains but allow excess flows to bypass during high flow events. A denitrification bioreactor design balances the ability to treat a significant percentage of the drainage flow with the ability to maintain sufficient retention time. Bioreactors in Iowa are being designed to accommodate up to 20% of the peak flow rate in the drainage main (estimated from pipe diameter and slope) for a retention time of four to eight h. The depth of reactor below ground surface is typically set by the depth of the drainage main (typically 1.2 to 1.5 m). The length of the reactor is controlled by the desired retention time while its width is a function of the anticipated peak flow rate through the system.

The theoretical hydraulic retention time (HRT, τ) of the bioreactor is calculated as:

$$\tau = \frac{\rho V}{Q} \qquad \text{Equation 2.1}$$

where V is the active reactor volume, ρ is the porosity of the packing media, and Q is the volumetric flow rate through the reactor. A retention time too short results in little nitrate removal because dissolved oxygen (DO) may remain too high for denitrification and residence time may be too small for nitrate utilization by denitrifying organisms. Alternately, a retention time too long will result in complete nitrate removal followed by other undesirable reactions such as the production hydrogen sulfide or methane (Blowes et al., 1994). A flow control structure located near the downstream end of the bioreactor allows farm operators to manually adjust the hydraulic retention time in the reactor. The active reactor volume can be adjusted by raising stop logs in the flow control structure for high precipitation periods during the growing season (spring and fall), and removing the stop logs during low precipitation periods (summer).

An interim design standard for denitrifying drainage bioreactors developed recently by the Natural Resources Conservation Service allows sizing of bioreactors for retention times that are sufficient to achieve a substantial reduction in nitrate-nitrogen concentration (NRCS, 2009). However, no reactor geometries (i.e. length to width ratio or cross sectional shape) are suggested. Furthermore, aside from preliminary data from this study (Christianson et al., 2010b), there are no reported scientific studies on appropriate bioreactor geometries to produce optimum nitrate removal under field conditions. Anecdotal field data show that long, narrow reactors and wider, more rectangular reactors are both able to remove nitrate but no comparison of removal efficiencies has been done. Theoretically, long, narrow channel designs provide the closest to ideal plug flow. Trapezoidal cross-sections have the potential to provide improved retention time control due to relatively higher flow velocities at low flow depths. The work described in this paper was conducted to investigate reactor hydraulic properties and nitrate removal under near-field conditions with different design geometries. Specific goals included understanding the differences between theoretical and actual HRT, evaluating dispersion for different design geometries, and assessment of nitrate removal potential in pilot-scale woodchip bioreactors.

Methods and Materials

Design Characteristics of The Pilot-Scale Bioreactors

Pilot-scale reactors with identical volumes (0.71 m^3) and depths (0.6 m), and three cross-sectional geometries – channel, rectangular and trapezoidal – were constructed with plywood and

installed at Iowa State University's Agricultural Engineering and Agronomy research farm near Ames, Iowa as described in Christianson et al. (2010b). The dimensions of these reactors are described detailed in Figure 2.1. The reactors were 1:10 of field-scale based on surface footprint. The inflow and outflow pipes (5 cm and 10 cm PVC, respectively) were placed in the bottom center of the bioreactors. The plywood boxes were lined with polyethylene tarpaulin and packed with woodchips. The chips were a mixture of various local hard wood species and similar to those used in field bioreactor installations. Particle size analysis showed a mean particle size of 1.1 cm, an effective size (D10, 10% by mass of woodchips was smaller than this size) of 7 mm, and a uniformity coefficient (D60/D10) of 2 (Christianson et al., 2010a). The layer of woodchips in the pilot reactors was covered with a lightweight geofabric and approximately 7 cm of topsoil. Control valves allowed manipulation of the flow rates and outflow was measured with Neptune™ T-10 water meters. Water depth within the reactor was set using a downstream flow control structure consisting of an upturned PVC elbow at the reactor outlet. Five to twelve PVC monitoring wells (2.5 cm diameter) were placed at predetermined locations within the reactors to monitor flow depth and redox conditions within the reactor. Flow depth data were used to calculate the active reactor volumes at water depths set by the downstream flow control structure. Feed water was obtained from a 4000 L underground reservoir connected to a 30 cm diameter county main drainage line that drained fields planted with corn and soybean. The rain volume passing through the reactors during the testing period was less than 0.6% of the total flow volume and was thus considered negligible.

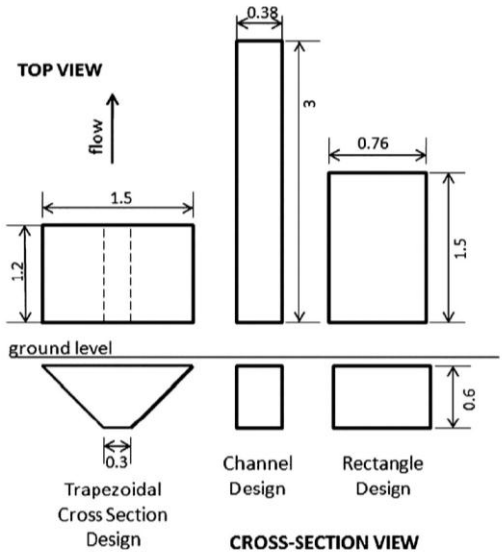

Figure 2.1 Plan and cross sectional views of pilot-scale denitrification bioreactors installed near Ames, Iowa; All dimensions are in meters.

Bromide Tracer Testing

Bromide tracer tests were conducted to determine the flow characteristics and in situ HRTs of each bioreactor. The test consisted of chasing a slug of 4.4 to 57 g of potassium bromide with four bioreactor pore volumes of drainage water at predetermined flow rates and flow depths. The channel reactor was evaluated at three flow depths with a single flow rate (3.8 L/min) and also at three different flow rates with a relatively constant flow depth (average of 21 cm). In-situ HRT was also determined for the three reactor cross-sectional geometries at a flow depth of 28 cm though the observed flow depth was 24 cm in the channel reactor due to leakage at the outlet connection. For the theoretical HRT calculation, a porosity value of 0.67 was used for the woodchip media based on laboratory determination (Christianson et al., 2010a) which compared well with an effective porosity of 0.7 reported by van Driel et al (2006). Water samples were analyzed for bromide and nitrate colorimetrically with a Lachat™ Quick-Chem 8000 automated analyzer (Standard Methods, 1998). Where possible, the Morrill Dispersion Index (MDI) was calculated to provide a qualitative indicator of dispersion. MDI is defined as:

$$\mathrm{MDI} = \frac{P_{90}}{P_{10}} \qquad \text{Equation 2.2}$$

where P_{90} and P_{10} are the 90th and 10th percentile values, respectively, from a log-probability plot of the time versus cumulative percentage of tracer recovered in the effluent (Metcalf and Eddy, 2003). While an MDI of 1.0 indicates ideal plug flow, the US Environmental Protection Agency considers reactors with MDIs less than 2.0 to have "effective" plug flow (Metcalf and Eddy, 2003). Tracer tests also allowed comparison of the in-situ tracer residence time with the theoretical HRT. The in-situ HRT was calculated as:

$$\bar{t}_{\Delta c} \approx \frac{\sum t_i C_i \Delta t_i}{\sum C_i \Delta t_i} \qquad \text{Equation 2.3}$$

where $t_{\Delta c}$ (bar) is the mean in situ HRT of the reactor calculated using discrete time steps; t_i is the retention time corresponding to the ith sample, C_i is the bromide concentration in the ith sample, and Δt_i is the time increment about C_i (Metcalf and Eddy 2003).

Nitrate Removal over A Range Of Theoretical Hydraulic Retention Times

To study nitrate removal at various retention times, theoretical HRTs of the pilot bioreactors were gradually increased from 1.5 to more than 10 h over two months with equilibration periods of at least one week at individual retention times and sampling events at least twice a week. The flow

depths were maintained at 28 cm for the rectangular and trapezoidal cross-section bioreactors and at 20 cm for the channel design; the flow rates were calculated based on the desired retention times. To maintain identical retention time in all reactors, a lower flow rate was required for the trapezoidal cross-section design due to its smaller plan area at low flow depths. Influent, effluent and well samples were analyzed for NO_3^-N in the laboratory while portable meters with specific probes were used to measure oxidation-reduction potential (ORP), DO, pH, and depth to water in the wells (WTW® 3300i pH/mV Field Meter and WTW® SenTix ORP Electrode Probe, Fisher Scientific Accumet® AP74; Solinst® water level meter Model 101). Influent samples were collected from the underground reservoir and effluent samples were collected from the outlet of the downstream control structure.

Simulated Storm Event Effects on Nitrate Removal

Simulated removal of nitrate in the bioreactors during a storm event was explored using a hydrograph test modeled after a typical Iowa drainage hydrograph. During the 3.2-day simulated event, flow rates were sharply increased and then consistently decreased to allow for theoretical HRTs of less than 2 h up to nearly 15 h. The trapezoidal cross-section design was not studied here as its smaller cross-sectional area resulted in a different required flow rate compared to the channel and rectangle designs to achieve approximately the same retention times for the hydrograph.

Impact of Influent Nitrate Concentration On Bioreactor Performance

The impact of influent nitrate-N concentration on reactor performance was studied by injecting a concentrated solution of potassium nitrate just upstream of the inlets of the channel and trapezoid cross-sectional reactors. Due to the orientation of the reactor injection ports in the field, only these two reactors were used for this test. Homogeneous mixing of the solution with flow from the drainage reservoir was assumed. The resulting influent nitrate-N concentrations were determined by mass balance. All tests were conducted during summer and fall of 2009 when drainage water temperatures ranged from 10.5 to 15.4˚C and DO concentrations in the drainage water ranged from 6.8 to 8.0 mg/L.

Results & Discussion

Bromide Tracer Testing

Because retention time is a primary design parameter affecting reactor performance, it is vital to evaluate actual in situ HRTs for the bioreactors and compare these with the theoretical HRT values

used in design calculations. The tracer test data allowed calculation of in-situ HRT and MDI values which facilitated comparison of the three reactor geometries.

Tracer tests revealed that mean in-situ HRT values for the bioreactors were substantially greater than the theoretical HRTs (Table 2.1). This discrepancy was likely due to nonideal conditions in the pilot reactors introduced by hydraulics within the woodchip media and at connections near the inlet and outlet of the reactor. While theoretical and in-situ HRTs for the channel reactor were similar at the lowest flow depth, their values diverged with increasing flow depth. This was a consequence of the design and location of the distributor and collector pipes within the reactor. Since these pipes were located at the bottom of the reactor, at higher flow depths (greater active volume), flow was more likely to be nonideal due to dead zones near the inlet and outlet of the structure. A larger volume of standing water in the downstream flow control structure at higher flow depths may also have affected the in-situ HRT values. On-site examination of the channel reactor revealed that while operating at a desired flow depth of 28 cm, the stand-pipe in the effluent control structure had settled by about 4 inches resulting in an actual flow depth of only 24 cm.

Table 2.1 Bromide tracer test results for pilot scale bioreactors

Design	Flow Rate (L/min)	Design Flow Depth (cm)	Percent Tracer Recovered	Calculated Retention Time (hr)	Tracer Mean Residence Time (hr)	Morrill Dispersion Index
Channel	3.8	2.5	85%	0.09	0.14	2.3
Channel	3.8	15	95%	0.52	1.1	3.3
Channel	3.8	24	64%	0.82	1.8	4.3
Channel	1.9	21	37%	1.4	ND	3.5
Channel	0.95	19	17%	2.6	ND	3.8
Rectangle	3.8	28	50%	0.95	ND	6.0
Trapezoid	3.8	28	72%	0.58	1.0	5.3

ND: Not determined because tracer concentration did not reduce to background levels during test period.

When the tracer tests on the channel reactor were repeated at the lower flow rates of 1.9 and 0.95 L/min, the effluent tracer concentration did not reduce to background levels even after five pore volumes of flow. This resulted in lower mass recoveries and an inability to determine the in-situ HRT using Eq. 2.3. Tracer mass recovery was also low for the test conducted on the rectangle reactor where dispersion was excessive due to the relatively large active cross-sectional area of flow. The slate of tests was not rerun as it was possible to calculate the tracer residence time for other tests which allowed the necessary comparisons to be made.

The divergence in theoretical and actual HRT values has a significant impact on the design of field-scale drainage bioreactors because if in-situ residence times are consistently greater than

theoretical retention times due to the presence of dead zones, bioreactor sizes would need to be reduced to prevent development of sulfate-reducing or methanogenic conditions. Unpublished data from field installations indicate that reactors designed using theoretical HRTs consistently perform better than expected i.e., remove more nitrate-N than predicted (ACWA, 2009). This may not always be desirable as complete removal of nitrate in the bioreactor may lead to anaerobic conditions causing production of hydrogen sulfide and methane. Field-scale bioreactor dimensions, specifically the reactor length, are based on providing a sufficient HRT. Thus, if tracer tests indicate in-situ residence times that are consistently and substantially greater than the design HRT, the latter can be reduced thereby reducing the bioreactor size and cost.

MDI values from the pilot-reactor tracer tests indicated that the channel reactor behaved as a plug-flow reactor at the smallest flow depth and the highest flow rate evaluated. Dispersion in the channel reactor increased as flow depth (and active reactor volume) increased. When comparing the three reactor geometries at similar flow rates and depth, greatest dispersion was observed for the rectangular cross section (MDI = 6.0) followed by the trapezoidal cross-section (MDI = 5.3) and the channel cross section (MDI = 4.3).

Others have also noted dispersion in bioreactor tracer tests. Chun et al. (2010) conducted tracer studies on a 6.1 m X 6.1 m field-scale woodchip bioreactor with a rectangular cross-sectional geometry and reported longitudinal and transverse dispersivities of 10.2 cm and 1.13 cm, respectively. When nitrate was introduced as a pulse load, the bioreactor removed 47% of the nitrate mass in 4.4 h. Van Driel et al. (2006) conducted tracer tests on a full-scale woodchip bioreactor and reported an in-situ HRT of 9 h at a flow rate of 11.2 L/min with little dispersion from the coarse to the fine media layers.

Nitrate Removal over a Range of Theoretical Hydraulic Retention Times

Denitrification was assumed to be the major sink of the influent NO_3^--N in the pilot bioreactors. Sufficient time was given for the bioreactors to stabilize nitrate removal at each retention time (Figure 2.2); however, at the smallest flow rates, it was difficult to manually control the valves to produce an exact flow hence the variability at the highest retention times. Nitrate-N concentrations in the influent sourced from the underground reservoir averaged 10.1 mg/L during the study period (Figure 2.2). Though HRT and age both increased over the test, the relatively clear breaks in Figure 2.2 a and b correlate indicating changes in effluent concentration were related to changes in retention time rather than age. Moreover, the bromide tracer tests were performed before any nitrate removal experiments in order to ensure sufficient time for inoculation. During these tests, dissolved oxygen

was rapidly removed and was absent in samples collected from wells beyond the first 20% of bioreactor length even at retention times < 4 h. Redox potential dropped to 0 ± 100 mV in the first 60 to 70% of the reactor length from the inlet at theoretical HRTs > 4 h.

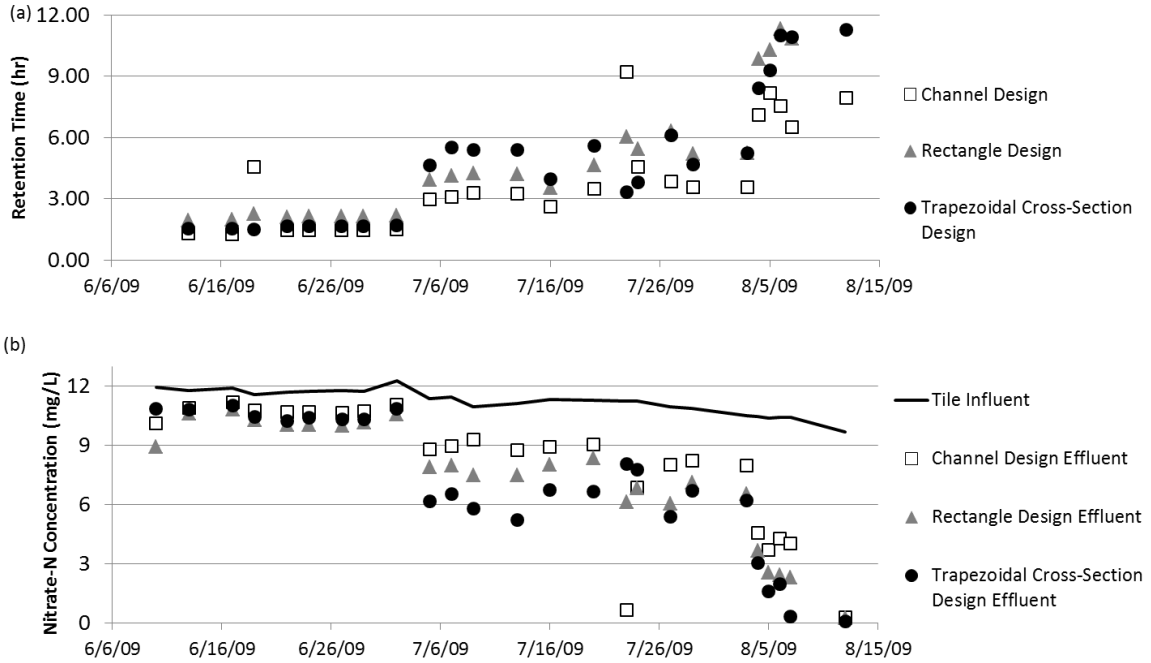

Figure 2.2 Retention times (a) and influent and effluent NO_3^--N concentrations (b) for three pilot-scale bioreactor design geometries

The mass reductions percentages of influent NO_3^--N were observed to be linearly correlated to the theoretical HRT values (Figure 2.3). The theoretical HRTs were calculated using the active reactor volume determined from water level measurements in the wells and the volumetric flow rate through the reactor. Note that the in-situ HRT values determined from tracer tests were approximately two times the theoretical HRT for the three different reactor geometries (Table 2.1). This is validated by the data shown in Figure 2.3 where nitrate removal was correlated to the theoretical HRT irrespective of the bioreactor design geometry. Theoretical HRTs of over 10 h were necessary for > 90% NO_3^--N removals. Approximately 30 to 70% NO_3^--N removal was observed between the 4 to 8 h of retention time suggested in the interim practice standard (NRCS, 2009). The three designs consistently produced removals of 2 to 4 g NO_3^--N/d at retention times ranging from 1.3 to 11.3 h (data not shown). At high retention times, low inflows resulted in low loading rates and thus nearly 100% mass reductions; at lower retention times, higher loads entered and lower percent removals occurred. Christianson et al. (2010b) provides a detailed analysis of the design geometry effects upon performance.

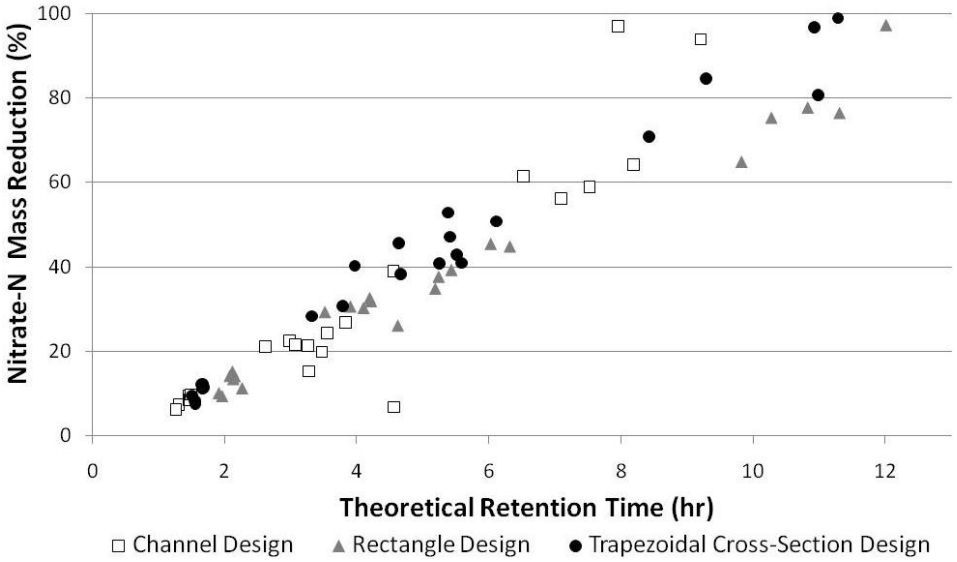

Figure 2.3 Percent mass NO_3^--N reduction at various theoretical hydraulic retention times for three bioreactor design geometries

Simulated Storm Event Effects on Nitrate Removal

When a hydrograph flow was simulated through the reactors, retention times changed rapidly affecting nitrate removal in the reactors (Figure 2.4). Minimum retention times at peak flow were 1.9 and 1.6 h for the channel and rectangle designs, respectively, and occurred at 0.66 d after initiation of the hydrograph test. The influent nitrate-N concentration ranged from 7.1 to 9.2 mg NO_3^--N/L during the hydrograph test. As the simulated drainage event proceeded, the channel and rectangular cross-sectional designs showed similar reductions in nitrate-N mass loads. However, when percent removals were normalized by retention time (% mass reduction per hour of HRT), the rectangular design showed consistently higher removals compared with the channel design. Though observed mass loading reductions for the two designs were similar (Figure 2.4a), small differences in reactor flow rate and flow depth produced slightly different theoretical HRTs, producing the differences noted in Figure 2.4b.

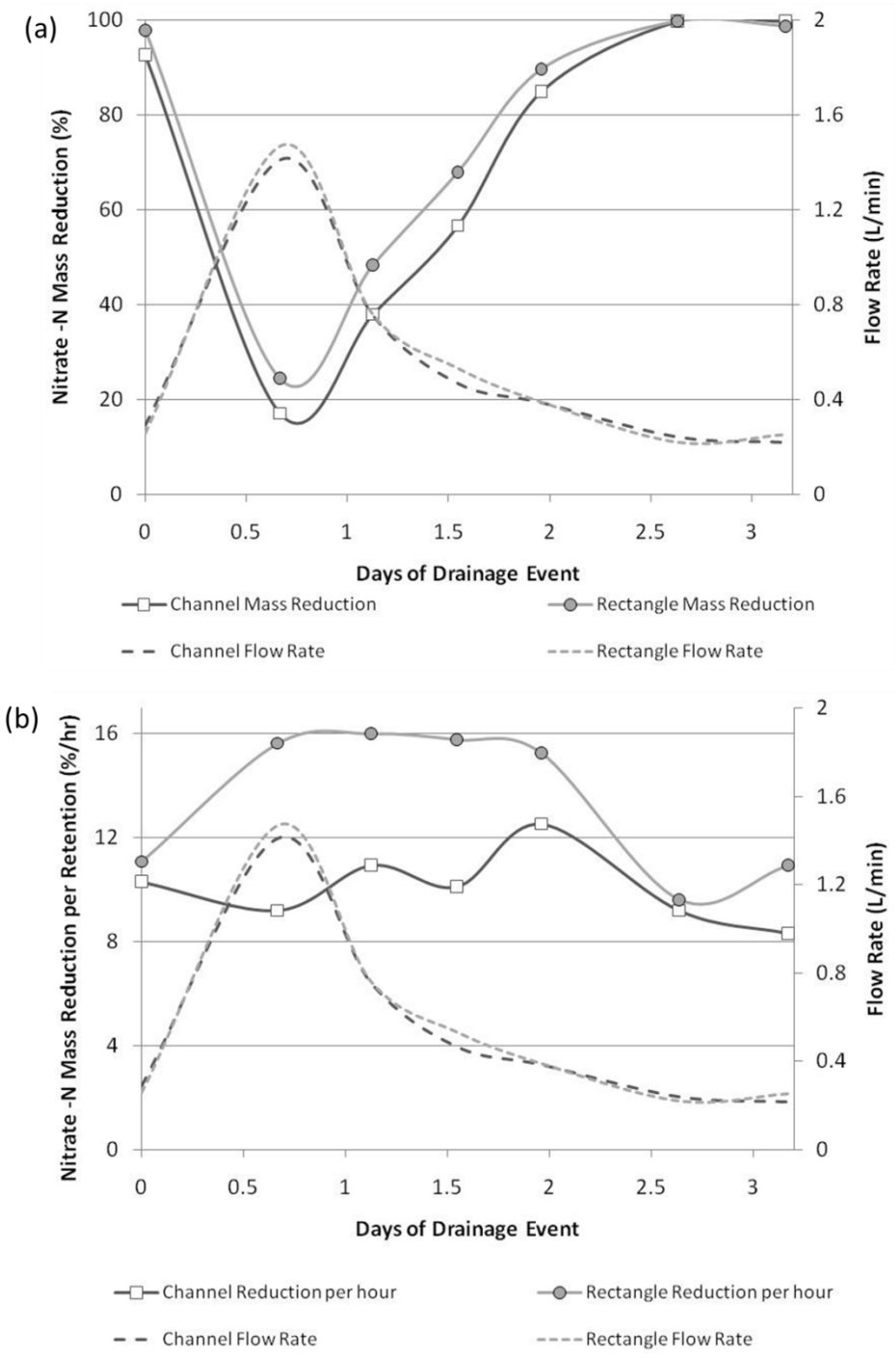

Figure 2.4 Impact of simulated hydrograph on NO_3^--N removal in channel and rectangular pilot reactors:(a) flow rates and percent mass reductions, and (b) flow rates and percent mass reductions normalized by retention time

Impact of Influent Nitrate Concentration on Bioreactor Performance

Increasing the influent nitrate loading by increasing the influent NO_3^--N concentration to approximately 25 mg/L had an adverse impact on the percent mass reduction of NO_3^--N (Figure 2.5) as tested in the channel and trapezoidal cross-section designs. It may appear that this behavior was contrary to that observed by Chun et al. (2009) who reported that denitrification reactors typically exhibit first order decay of nitrate. However, we also noted that during the highest NO_3^--N tests in our study, the drainage water temperature dropped noticeably from 13.6˚C to 10.5˚C. It is possible that the temperature drop affected denitrification as many researchers have documented reduced denitrification rates at lower temperatures (Volokita et al. 1996; Robertson et al. 2000) though denitrification has been reported to occur even at temperatures as low as 2 to 4˚C (Robertson and Merkley, 2009). Robertson et al. (2005) documented their highest nitrate removal rates under the highest influent concentrations at both high and low temperatures though they modeled denitrification using a zero-order reaction. Diaz et al. (2003) also reported a consistent trend of increased nitrate removal with increasing temperature except when the influent concentration was higher at a lower temperature. In the latter case, the first order reaction effect may have been masked by the change in temperature as was in our study.

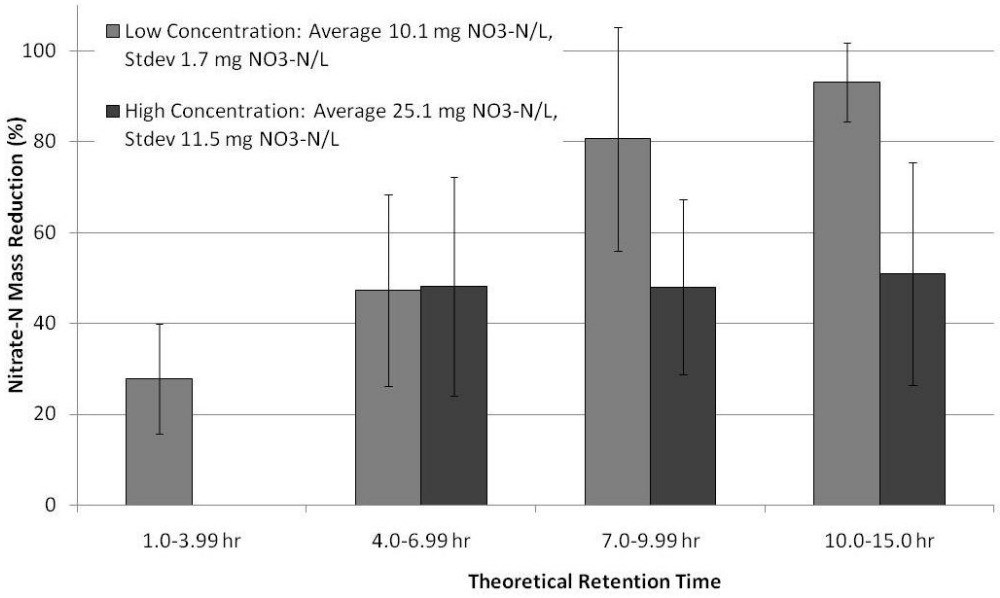

Figure 2.5 Average percent reduction of NO_3^--N mass for two influent concentrations at four ranges of theoretical retention times. Error bars indicate one standard deviation; No data was collected for high concentration at the lowest retention time.

Based on the entire woodchip volume in the pilot bioreactors, the channel, rectangle, and trapezoidal designs averaged removal rates of 3.8, 5.6, and 4.1 g NO_3^--N/m^3/d, respectively (standard deviations 0.92, 1.1, and 0.87, respectively) over the duration of the study. These rates are an order of

magnitude higher than those reported by Jaynes et al. (2008) (0.62 g NO_3^--N/m^3/d) for denitrification wall reactors installed in Iowa. Based on the surface footprint of the bioreactors, which is common for comparison with wetlands, the pilot bioreactors had average nitrogen removal rates of 2.3, 3.4, and 1.5 g NO_3^--N/m^2/d for the channel, rectangle, and trapezoidal designs, respectively (standard deviations 0.56, 0.65, and 0.32, respectively). These rates are consistent with van Driel et al.'s (2006) findings of 2.5 g NO_3^--N/m^2/d for a lateral-flow wood media reactor. Xue et al (1999) documented substantially lower daily removal of 0.05 to 0.28 g NO_3^--N/m^2 for a constructed wetland treating drainage water.

Conclusions

Denitrification drainage bioreactors have the potential to significantly reduce nitrate-loading from drained agricultural fields to surface waters and thus play a key role in mitigating the hypoxia in the Northern Gulf of Mexico. In this work, 1:10 scale pilot reactors were installed and evaluated to understand nitrate removal from woodchip denitrification systems under field conditions and the impacts of design geometry on bioreactor hydraulics. In-situ HRTs were nearly twice as large as theoretical HRTs, likely due to nonideal flow resulting from the design and placement of the flow distributor and collector pipes within the reactor. Comparison of field-scale bioreactor HRTs with calculated HRTs for those systems will help determine if this difference was a factor of these pilot-scale systems. Increased hydraulic retention time improved bioreactor performance with the current design criteria of 4 to 8 h of retention providing between 30 to 70% mass reduction regardless of cross-sectional shape. MDI values indicated that the channel cross-sectional design produced the least dispersion, especially at high flows and low flow depths. Influent nitrate concentration and possibly to a lesser degree temperature also appeared to play a major role in the performance of these systems. It is recommended that future work focus on evaluating the hydraulic characteristics of field scale systems and developing improvements in the design and location of the flow collector and distributor pipes within the bioreactors.

Acknowledgements

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CHAPTER 3 OPTIMIZED DENITRIFICATION BIOREACTOR TREATMENT THROUGH SIMULATED DRAINAGE CONTAINMENT

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Abstract.

In the design of wood-based, enhanced-denitrification bioreactors to treat nitrate in agricultural drainage, the consideration of the highly variable flow rates and nitrate concentrations inherent to many drainage systems is important. For optimized mitigation of these nitrate loads, it may be best to contain drainage water prior to treatment in order to facilitate longer, more constant retention times rather than to allow cycles of flushing and dry periods in the denitrification bioreactor. Simulated containment prior to bioreactor treatment compared to passing drainage directly through a bioreactor was investigated with the use of six pilot-scale denitrification bioreactors constructed with plywood and filled with *Pinus radiata* woodchips at Massey University No. 4 Dairy Farm (Palmerston North, New Zealand). Initial bromide tracer tests were followed with a series of five simulated drainage events each at successively declining inflow nitrate concentrations. During each drainage event, three pilot bioreactors received a simulated hydrograph lasting 1.5 d (Non-Containment treatment) and three pilot bioreactors received the same total drainage volume treated over 4 d at a constant flow rate (i.e. constant retention time; Containment treatment). Results showed significantly different total mass removal efficiencies of 14.0% vs. 36.9% and significantly different removal rates of 2.1vs. 6.7 g N m^{-3} d^{-1} for the Non-Containment and Containment treatments, respectively, which indicated that treating drainage at constant retention times provided more optimized nitrate removal. While this work was done to evaluate treatment under New Zealand drainage conditions, it also provides valuable information for optimizing agricultural drainage denitrification bioreactor performance in general.

Keywords. Nitrate, denitrification bioreactor, agricultural drainage

Introduction

The implementation of agricultural drainage worldwide has allowed increased agricultural intensification and productivity (Ritzema et al., 2006), but these gains have not been without environmental impact. Nitrate (NO_3^-) losses from agricultural drainage have been documented in many regions (Mohammed et al., 1987; Randall and Goss, 2001; Singh et al., 2002; Noory and

Liaghat, 2009) and regulatory bodies are increasingly trying to address the resulting decline in water quality (European Commission, 1991; Horizons Regional Council, 2007; USEPA, 2007). One of the newest, on-farm approaches for mitigating NO_3^- loadings from agricultural drainage is the use of enhanced denitrification. Drainage waters high in NO_3^- are routed through denitrification bioreactors where NO_3^- transformation is enhanced by an additional carbon source and the maintenance of saturated conditions (Schipper et al., 2010a).

Wood-based denitrification bioreactors for reducing NO_3^- in agricultural drainage have shown promise in American Midwest drainage systems (Jaynes et al., 2008; Chun et al., 2010; Woli et al., 2010), and it is thought this mitigation strategy may also be effective in other locations. In New Zealand, the average annual drainage NO_3^- losses under grazed dairy pastures are approximately 25-30 kg N ha^{-1}, which is similar to loadings from row cropped areas in the US Midwest (Ledgard et al., 1999; Randall and Goss, 2001; Monaghan et al., 2002). A major difference between these two drainage systems is that while Midwestern drainage typically has relatively consistent NO_3^- concentrations over a drainage season at a given site, in New Zealand drainage systems there is a significant trend of declining NO_3^- concentrations over the season with the highest concentrations typically occurring within the first 100-150 mm of drainage (Monaghan et al., 2002; Houlbrooke et al., 2004). Hydrologically, New Zealand's mole and pipe drainage systems have high peak flows stimulated by storm events (pulsed flow) with significant periods of no flow between events (Bowler, 1980).

Uncontrolled and infrequent pulsed drainage flow rates present a challenge for bioreactor treatment as these fluctuating flow rates result in fluctuating bioreactor retention times. Low bioreactor retention times occurring at peak flow rates may result in dissolved oxygen (DO) concentrations that are too high for NO_3^- to be reduced by denitrifiers. Indeed, past work has documented decreased bioreactor NO_3^- removal at higher flow rates (Woli et al., 2010; Christianson et al., 2011). In addition, short duration, intensive flows present design issues because designing a system for 100% of the peak flow rate requires an impractically large bioreactor volume. Currently in the Midwest, bioreactors are designed using a design flow rate that is only a portion of the peak flow rate, meaning that not all of the total annual volume receives bioreactor treatment (Christianson et al., 2009; USDA-NRCS, 2009).

In New Zealand, drainage water NO_3^- mitigation could focus on capturing and treating early season drainage water when NO_3^- concentrations are the highest (i.e. first 100 - 150 mm of drainage). In order to achieve this, temporary diversions or impoundment facilities may be constructed in paddock gullies to retain drainage water between drainage events. The controlled, slower discharge

of this impounded drainage into a denitrification bioreactor would allow treatment at a longer and more consistent retention time. This two stage containment/ treatment system would allow more effective treatment of nearly all the early season drainage volume by maintaining a sufficient retention time. A two stage design is a major departure from current denitrification bioreactor design in the US Midwest. Pre-treatment containment of drainage could provide at least two related benefits including: (1) stabilization of flow rate variability to allow treatment at a longer, more constant bioreactor retention time, and (2) treatment of all the critical early-season drainage containing the highest NO_3^- concentrations. Though past work documented declining nitrate removal during a simulated hydrograph (Christianson et al., 2011), there has been no direct treatment comparison of uncontrolled rapid drainage discharge with controlled, slower discharge from containment systems.

The objective of this work was to compare bioreactor NO_3^- removal occurring during steady retention times (i.e. simulated drainage containment) with removal occurring during flow rate-varying drainage events. It was hypothesized that the steady retention times would provide improved NO_3^- removal over the course of the simulated drainage season compared to non-containment. Moreover, this work assessed the feasibility of denitrification bioreactors for New Zealand drainage systems by simulating declining NO_3^- concentrations over the drainage season, using realistically scaled local drainage hydrograph events, and operating under in situ temperatures.

Methods

Six pilot-scale bioreactors (2.0 m x 0.31 m x 0.85 m) were constructed with plywood in two sets of three, which were installed in June 2010 at Massey University No. 4 Dairy Farm near Palmerston North, New Zealand (Figure 3.1). The site receives an average annual rainfall of 980 mm and has a low average monthly soil temperature in July of 7°C. The inside surface of each bioreactor was painted with exterior house paint and a Non-toxic silicone sealant (Ecoshield™), and all seams were sealed with silicone caulk to prevent leakage.

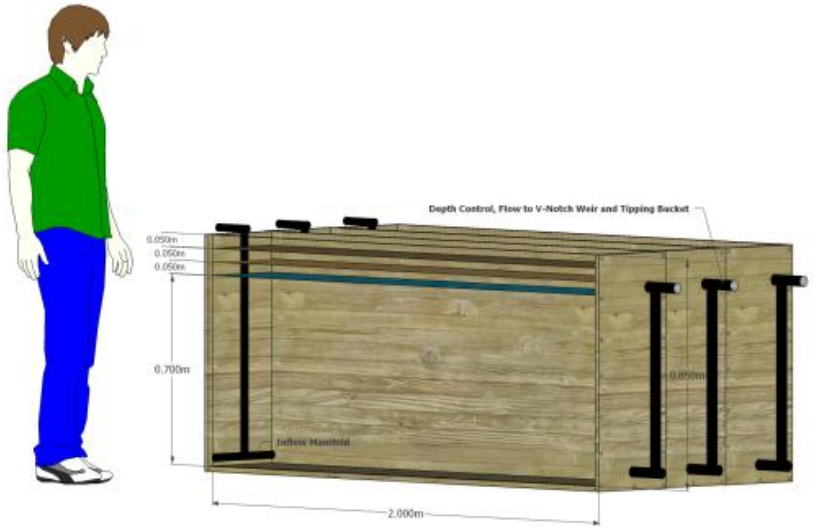

Figure 3.1 Schematic of three of the six pilot bioreactors with flow from left to right and location of outflow monitoring noted; Image does not reflect the painted interior or caulk sealed seams.

The bioreactors were filled with pine chips made in May 2010 from 1 yr old *Pinus radiata* prunings at the No. 4 Dairy Farm. The woodchip size distribution by dry weight was: >2.2 cm: 14%, 1.1-2.2 cm: 30%, 0.8–1.1 cm: 24%, and <0.8 cm: 32% with an estimated porosity of 60% and bulk density (dry weight) of 190 kg m^{-3}. Porosity was determined using methods described in Christianson et al. (2010) where one liter jars were packed with woodchips and then filled with water. After 24 h (i.e. after the woodchips had absorbed some of the initial volume), the water was replenished and this final volume was used to determine porosity. The bioreactors were filled to a depth of 75 cm with woodchips and approximately a 5 cm depth of soil was used to cap the chips. The soil, a Tokomaru Silt Loam, was taken from a grazed long-term (> 10 yr) pasture at the No. 4 Dairy Farm. One liter of this soil was also scattered among the woodchips during filling to inoculate the system with native denitrifiers, although no inoculation of other similar systems has been necessary to date (Schipper et al., 2010a).

Outflows from the pilot-scale bioreactors were measured with v-notch weirs and water depth loggers (4 bioreactors; NIWA Hydrologger 2001) or tipping buckets with loggers (2 bioreactors; Odyssey Data Logger). Flow rates were also manually verified with a graduated cylinder and stopwatch. During the trials, one of the v-notch weirs malfunctioned, and manual flow measurements were used instead of logged data for this single replicate. Flow data were logged every ten minutes and were then reduced by calculating thirty minute average flow rates to be used in the statistical analysis. Two monitoring wells were installed at each end of the bioreactors to document water

depth and solution dissolved oxygen (DO) (YSI Model 55). Water temperature was continuously logged every hour (Thermochron iButton® DS1921Z, Dallas Semiconductor) in two of the six outlet wells (within 15 cm from outlet) and in the constant head feed tank. During the testing period (1 July to 1 August, 2010), rain at Dairy Farm No. 4 was 45 mm (less than 1% of the water balance for each bioreactor).

Water in a runoff/drainage pond at the No. 4 Dairy Farm was pumped to a 5000 L mixing tank, where it was dosed with fertilizer grade potassium nitrate to mimic nitrate concentrations in agricultural drainage. Water in this supply tank was gravity fed to a constant head tank controlled by a float valve. Each bioreactor received water from this constant head tank through a 15 mm alkathene pipe with flow rates manually controlled by a ball valve. The inflow pipe (15 mm alkathene) extended to the bottom of the bioreactor where a diffuser manifold tee was attached. The outflow side of the bioreactor had an opening approximately 2.5 cm from the bottom of the bioreactor to which a head-controlling stand pipe (25 mm alkathene, 70 cm height) was attached. The depth of water in each bioreactor was set at 70 cm resulting in a saturated volume of 0.434 m^3 (woodchip volume 0.465 m^3). The retention time calculation was based on the entire woodchip volume (to reflect the entire investment) multiplied by the woodchip porosity and divided by the flow rates from the loggers.

Tracer Test

A bromide tracer test was conducted to determine the in situ residence times and dispersion indices for the reactors. A one liter slug containing 28 g NaBr was injected into each pilot bioreactor upstream of the inlet and at least 15 outflow samples were spaced over time to capture at least four pore volumes. A pore volume was defined as the volume equal to the total saturated volume (0.434 m3) multiplied by the woodchip porosity (60%). During these tests, potassium nitrate was used to dose the inflow pond water to achieve a concentration of 36.5 mg NO_3^--N L^{-1}. Tracer tests were run at four different retention times (i.e. 4.4, 7.7, 10.7, and 15.7 h of retention), two of which were duplicated (4.4 and 15.7 h). Outflow samples were analyzed for bromide, nitrate and sulfate with ion chromatography (Lachat 5000), though there were no significant differences between inflow and outflow sulfate values (inflow 15.3±0.3 mg L^{-1}, outflow 15.2±1.9 mg L^{-1}). The tracer residence time and Morrill Dispersion Index (MDI) were calculated using methods from Metcalf and Eddy (2003).

Drainage Event Simulations

Two treatments (triplicate), referred to as Containment and Non-Containment, were used. The Containment treatment simulated impoundment of a drainage event with subsequent slow release

of impounded water to allow treatment at a longer, more constant retention time, whereas, the Non-Containment treatment bioreactors received drainage at flow rates typical of drainage events at the site with no prior storage simulated. Drainage flow rate data from 910 m^2 research plots at the No. 4 Dairy (2009, unpublished data) were analyzed to determine the average length of time between drainage events and the typical duration and flow rates of individual drainage events in order to design the two treatments. The flow rate for the Non-Containment treatment was modeled after the descending limb of a drainage hydrograph using:

$$Y = Ae^{-bx} \qquad \text{Equation 3.1}$$

where Y was the flow rate at any given time (L min^{-1}), A was the peak flow rate (L min^{-1}, average peak rate from 2009 data: 16.6 L min^{-1}), b was a decay coefficient which was solved for, and x was time (minutes). Using the Microsoft Excel solver function, the total simulated event volume was set to the 2009 average drainage event volume (7304 L), and b, the decay coefficient, was solved for automatically (0.0022).

For the Containment treatment, a retention time of 14 h was chosen to provide a high NO_3^- removal potential based on reported retention times from previous agricultural drainage bioreactor research (Van Driel et al., 2006; Chun et al., 2009; Chun et al., 2010; Christianson et al., 2011). With this retention time, a porosity of 0.60 and a woodchip volume (0.465m^3), a resulting flow rate was calculated (approximately 0.33 L min^{-1} per bioreactor). This flow rate multiplied by the average length of time between 2009 events (approximately 4 d) gave 1920 L as the total volume to be treated in a single Containment treatment event. Using the average event volume from the 2009 data (7304 L per 910 m^2 plot), the simulated Non-Containment events were proportionally sized (1920 L / 7304 L = 0.26). The 2009 data average peak flow rate (16.6 L min^{-1}) was downsized using this factor of 0.26 and the resulting scaled peak flow rate (4.36 L min^{-1}) was used to develop the final Non-Containment hydrograph model:

$$Y = 4.36e^{-0.0022x} \qquad \text{Equation 3.2}$$

For ease of field implementation, this equation was simplified into a three stage hydrograph. The retention time and stage length of the three stages were iteratively calculated so as to (1) be similar to the modeled hydrograph (Equation 3.2) and to (2) ensure the total volumes of each treatment were the

same (approximately 1920 L). The three stages consisted of flow rates of 3.5-4 L min^{-1} for four h (stage one), 2 L min^{-1} for four h (stage two), and 0.3 L min^{-1} for 28 h (stage three). These three flow rates equated to retention times of approximately 1.5, 2.5, and 14 h for the stages, respectively. For each drainage event, the Containment treatment ran continuously at 14 h retention time for four d, whereas, the Non-Containment treatment ran for 1.5 d (approximate 2009 drainage event length) after which the flow to these bioreactors was turned off for 2.5 d without lowering the water-table height in the bioreactor.

The amount of supplemental KNO_3 added to the 5000 L supply tank was reduced over the series of simulated drainage events to mimic the typical decline in NO_3^- concentrations in mole and pipe drainage over a drainage season from grazed pastoral land in New Zealand (Monaghan et al., 2002; Houlbrooke et al., 2004). The four d drainage event test was repeated five times at subsequently lower inflow nitrate concentrations (35.6, 27.5, 12.8, 11.2, 7.68 mg NO_3^--N L^{-1}) over twenty d. During the drainage events, water samples were taken at the end of each hydrograph event stage (i.e. at time = 0, 4, 8, 24 and 36 h) with additional samples taken at 24 h intervals for the Containment treatment. Inflow and outflow samples were analyzed for NO_3^--N with a Technicon Auto Analyser II using methods from Kamphake et al. (1967). Least significant difference student t-tests (SAS® Software) were used to analyze for statistically significant differences between treatments (α=0.05).

Results

Tracer results

Normalized bromide tracer data (Figure 3.2a) showed that the tests, which were run with a range of retention times (Table 3.1), described similar breakthrough curves for each reactor (tracer elution complete within 4 pore volumes) and allowed sufficient time for NO_3^- removal to initiate (Figure 3.2b and Table 3.1). The bromide concentrations were normalized with respect to the maximum outflow bromide concentration for each test (maximum concentrations ranged from 20 mg Br-/L to 66 mg Br-/L). The bromide tests indicated no major leakage between the bioreactors as the curve shape and timing was relatively consistent for all treatments.

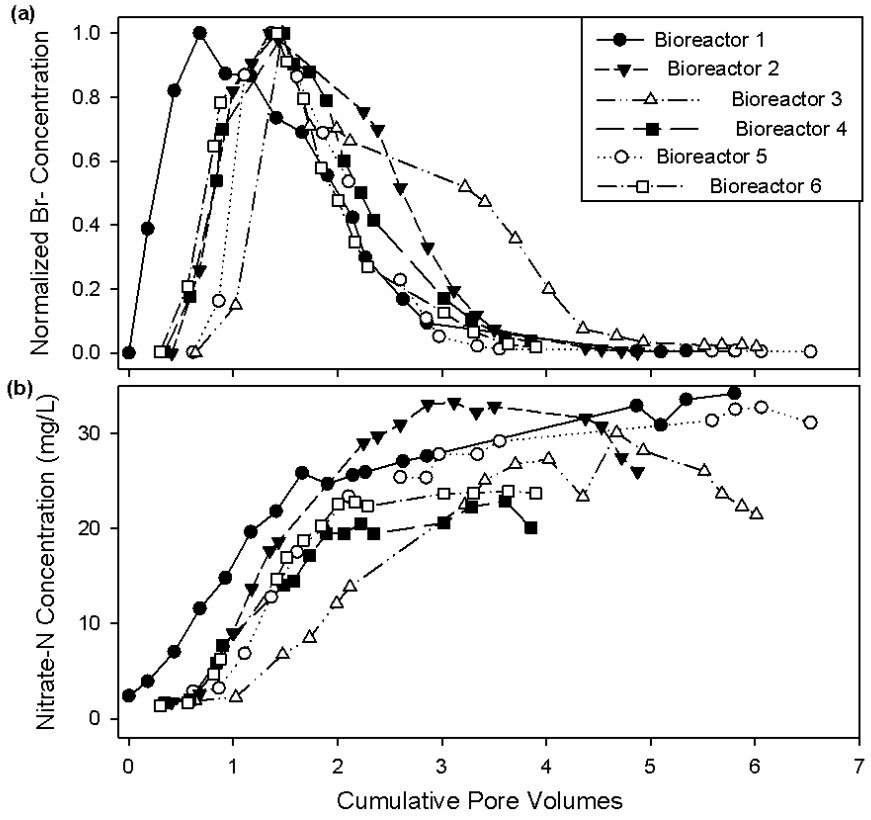

Figure 3.2 Six pilot bioreactors' tracer test normalized bromide (a) and nitrate concentrations (b) in outflow; similar symbol shapes denote similar retention times. Br- concentrations normalized with respect to maximum outflow concentration for each test.

Table 3.1 Bromide tracer test results parameters for six pilot denitrification bioreactors; * not accurate due to low tracer recovery

	Bioreactor 1	Bioreactor 2	Bioreactor 3	Bioreactor 4	Bioreactor 5	Bioreactor 6
Mean Retention Time based on flow data (h)	4.5	10.7	7.7	15.9	4.3	15.5
Tracer Test Cumulative Pore Volumes	5.8	4.9	6.0	3.9	6.5	3.9
Tracer Test Length (h)	25	43.5	37	71	27.5	72.0
Tracer Recovery (%)	101%	83%	57%	40%	97%	46%
Morrill Dispersion Index (MDI)	7.6	3.2	3.1	2.7	2.2	2.9
Mean Residence Time based on tracer data (h)	5.2	14.1	13.4	21.2*	4.1	19.6*
Percent difference in calculated retention vs. residence	14%	24%	42%	25%	-5%	21%
NO_3^--N concentration of final sample	34.2	26.0	21.5	20.1	31.1	23.7
Percent NO_3^--N concentration reduction of final sample	6.3%	28.8%	41.2%	45.0%	14.8%	35.2%

Tracer recovery was high for bioreactors 1, 2, and 5 (Table 3.1). Bioreactors 4 and 6 had tracer recoveries too low to accurately calculate mean tracer residence time, though these values are nevertheless reported. The low tracer recoveries corresponded with the highest retention times and

longest duration tested (approximately 72 h). In general, the tracer residence time was greater than the retention time calculated using flow data. Except for bioreactor 5, the tracer residence time ranged from 14 to 42% greater than the calculated retention time. Differences in these two values were most notable for bioreactors 2 and 3 whose retention times of 10.7 and 7.7 h yielded residence times of 14.1 and 13.4 h, respectively.

Because denitrification bioreactors are designed to be plug flow reactors, the MDI is an important indicator of reactor hydraulic performance. Dispersion in the bioreactors was evaluated using the MDI where an MDI value of 1 signified theoretically ideal plug flow reactor conditions (Metcalf and Eddy, 2003). The MDIs for bioreactors 2-6 were between 2.2 and 3.2 indicating similar hydraulic characteristics among boxes and overall plug flow conditions. The high MDI in bioreactor 1 (7.6) was a consequence of the relatively early tracer peak.

During the tracer testing period, solution DO sampled in wells near the inlet side of the bioreactors ranged from 4.27 to 3.02 mg L^{-1} and in wells near the outlet ranged from 0.73 to 0.24 mg L^{-1}, which indicated that conditions suitable for denitrification were present. This was further corroborated with NO_3^- data from the tracer tests samples, which showed NO_3^- removal occurring towards the end of the test (Figure 3.2b). The bioreactors with the shortest retention times (bioreactors 1 and 5) had higher final outflow NO_3^- concentrations compared to the bioreactors with the longest retention times (bioreactors 4 and 6) at 34.2 and 31.1 vs. 20.1 and 23.7 mg NO_3^--N L^{-1}, respectively.

Drainage Event Simulation

The flow measurements for the treatment replicates matched closely. The total cumulative volumes of the treatment means were within 3.5% of each other at 8909 vs. 9201 L for the Non-Containment and Containment treatments, respectively (Figure 3.3a and b). The volume means of the individual events matched the experimental design treatment volume specification of 1920 L per event reasonably well with an average of 1810 L per event (standard deviation 144 L). The volume means were not significantly different for all treatments except for the maximum volume of 2027 L from Event #2 Containment treatment and the minimum of 1605L from Event #5 Non-Containment treatment. Leakage from bottom seams of the bioreactor plywood boxes was quantified to be an average of 16% of flow through the bioreactors.

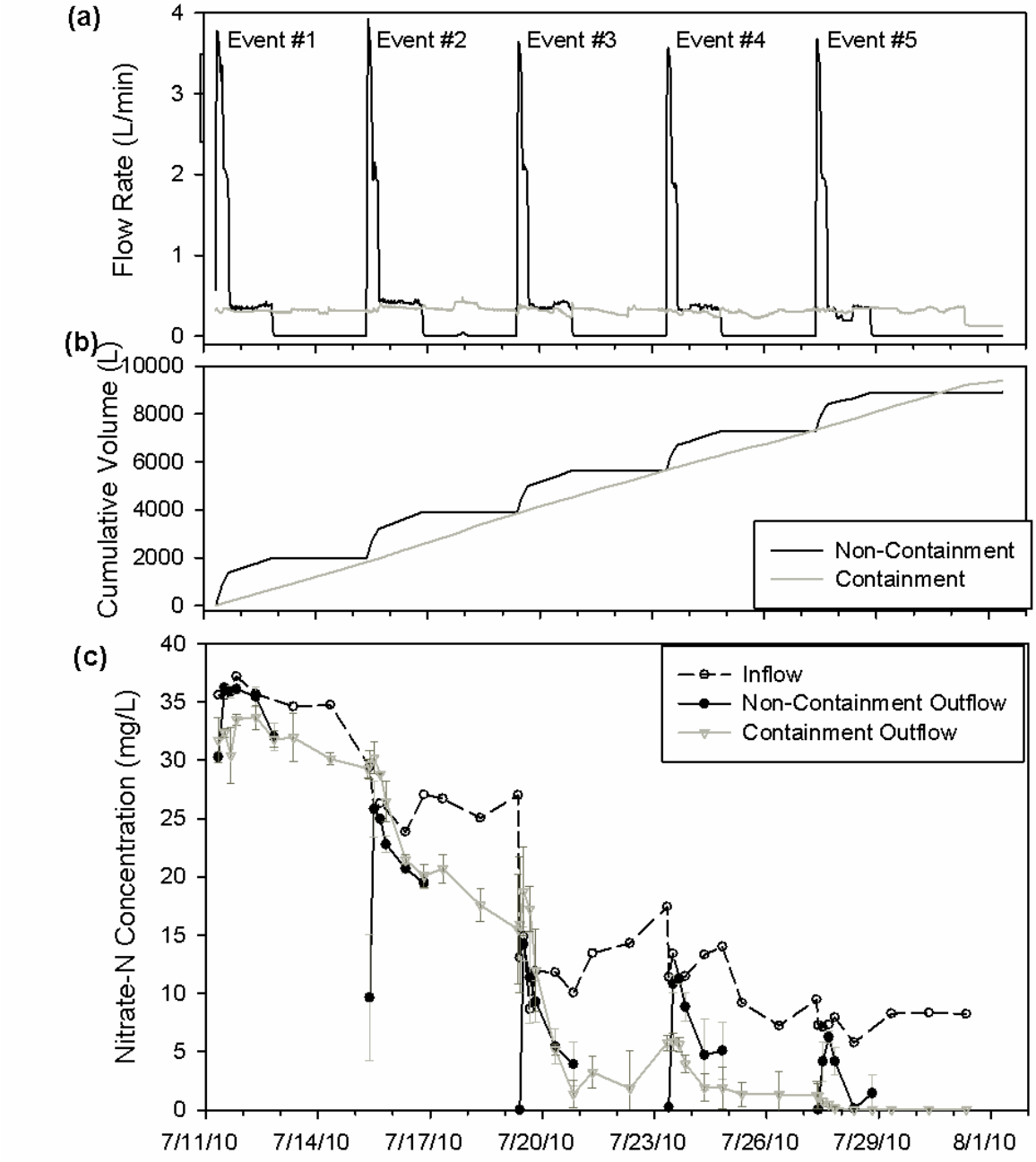

Figure 3.3 Pilot denitrification bioreactor mean flow rates (a), cumulative volumes (b) and inflow and outflow NO_3^--N concentrations with error bars representing ±one standard deviation (c) by treatment for five simulated drainage events (#1 – #5)

The initial pond water NO_3^- concentrations ranged from 0.35 to 3.9 mg NO_3^--N L^{-1}, and after KNO3 dosing the bioreactor inflow ranged from 37.2 to 5.8 mg NO_3^--N L^{-1} in Events #1-#5,

respectively (Figure 3.3c, Table 3.2). Spikes in NO_3^- concentrations caused by sampling near the time of NO_3^- dosing were not included in the analysis as they were not representative of the equilibrated NO_3^- concentrations in the constant head tank. At the beginning of each 1.5 d Non-Containment Event, water contained in Non-Containment bioreactors had very low concentrations as it had been retained in the bioreactors for 2.5 d (i.e. since the end of the previous event) (Figure 3c). As the flow rate peaked on the rising hydrograph, the outflow NO_3^- concentrations generally increased rapidly to within a standard deviation of the inflow concentration. This was most likely because the high flow rates (and associated low retentions time of 1.5 and 2.5 h) were too rapid for DO to be completely removed from the water, thus, limiting denitrification, though DO was not measured to verify this. Nitrate removal increased after approximately one pore volume at the final stage of the Non-Containment treatment (14 h retention), although these bioreactors only ran at this retention time for approximately two pore volumes (28 h total for this stage).

Table 3.2 Mean NO_3^--N mass into and out of bioreactors with standard deviation in parenthesis and mass removal efficiency and removal rate by treatment for each drainage event and overall; means with the same letter or symbol (* or †) are not significantly different (α=0.05)

	Average Inflow Conc. (mg NO_3^--N L^{-1})	NO_3^--N Mass In (g/bioreactor)		NO_3^--N Mass Out (g/bioreactor)		NO_3^--N Removal Efficiency (%)		Removal Rate (g N m^{-3} d^{-1})	
		Non - Containment	Containment	Non – Containment	Containment	Non – Containment	Containment	Non – Containment	Containment
Event 1	35.57	71.9 (13.5)	62.1 (4.4)	70.6 (13.1)	56.2 (3.6)	1.81g	9.46fg	0.71d	3.17b
Event 2	27.50	53.8 (5.5)	53.1 (2.9)	43.8 (6.2)	38.4 (2.3)	19.0ef	27.6de	3.11bc	7.69a
Event 3	12.84	19.8 (3.5)	25.0 (2.0)	16.8 (4.7)	9.5 (1.4)	14.3f	62.3c	1.46cd	8.49a
Event 4	11.20	21.3 (3.4)	17.2 (2.3)	15.2 (3.9)	3.4 (1.9)	29.7de	80.7b	3.34bc	7.40a
Event 5	7.68	10.1 (1.6)	13.3 (1.3)	6.4 (2.2)	0.2 (0.1)	38.0d	98.7a	2.02bcd	7.05a
Total		177*	171*	153*	108†	14.0†	36.9*	2.1†	6.7*

At the beginning of each event, the Containment treatment bioreactors had noticeably higher outflow concentrations than the inflow concentrations which were a result of the higher inflow nitrate concentration used in the previous event (Figure 3.3c). The length of these high outflow concentrations coincides well with this treatment's 14 h retention time; after this initial pore volume, the Containment bioreactor outflow concentrations were significantly lower than the inflow based on their standard deviations.

The Containment treatment emitted significantly less NO_3^- mass than it received for four of the five events (Events #2 - #5), whereas, the Non-Containment treatment had significantly less NO_3^- mass emitted for only one event (Event #2) (Table 3.2). Inflow NO_3^- mass differed between the two treatments in Event #1 with significantly more NO_3^- received by the Non-Containment treatment. In

this event, removal was low for both treatments with each having statistically similar amounts for received versus emitted NO_3^-. The total NO_3^- mass loading received for all five events was 177 and 171 g NO_3^--N per bioreactor and the total emitted was 153 and 108 g NO_3^--N per bioreactor for the Non-Containment and Containment treatments, respectively (Table 3.2 and Figure 3.4). There was no significant difference between the total inflow masses for both treatments and the outflow mass from the Non-Containment treatment.

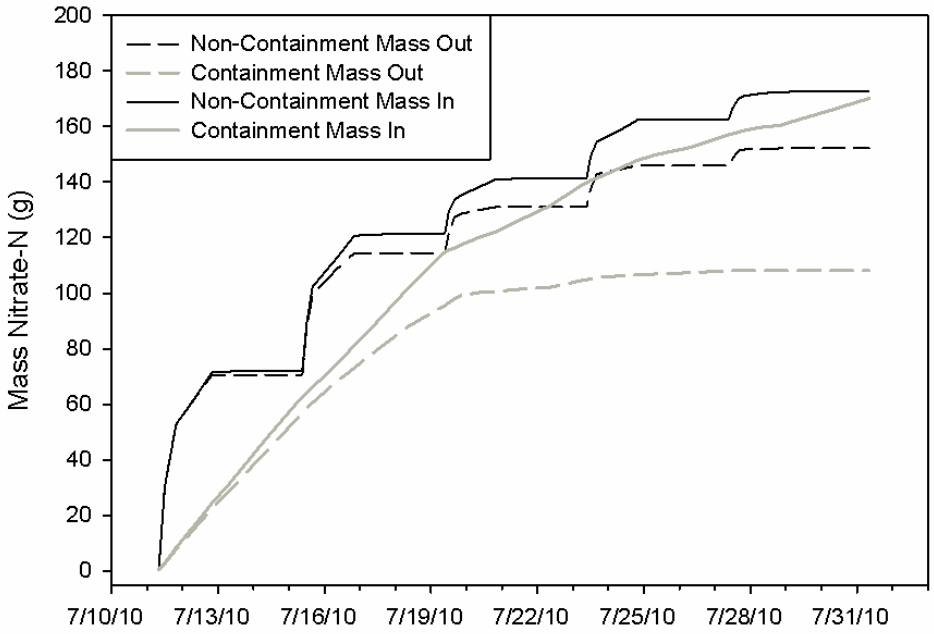

Figure 3.4 Mean cumulative inflow and outflow NO_3^--N mass for five drainage events by treatment (Containment and Non-Containment) for six pilot bioreactors

Mean removal efficiencies increased for both treatments as lower inflow concentrations were subsequently used (Table 3.2). These efficiencies ranged from 1.81% for the highest inflow concentration (Event #1) with the Non-Containment treatment to 98.7% for the lowest inflow concentration (Event #5) with the Containment treatment. In Events #3 - #5, the Containment treatment had significantly higher removal efficiencies than the Non-Containment treatment. This was echoed with the significantly greater total removal efficiency of 36.9% versus 14.0% for the Containment and Non-Containment treatments, respectively.

It was thought that containment simulation would allow optimized treatment of the high NO_3^--N concentrations in early season drainage. However, Event #1 which was designed to simulate early season peak concentrations had the lowest removal efficiencies at 1.9% and 9.9% for the Non-

Containment and Containment treatments, respectively, with no significant difference between treatments. There was a strong trend for higher removal efficiency as the inflow concentration decreased.

The removal rates by event were calculated from the total amount of NO_3^- removed during an event divided by the reactor volume and event length (four d). The total removal rate was calculated using the total mass removed during all five events divided by reactor volume and 20 d. For each event the removal rates for the Containment treatment were significantly higher than the Non-Containment (Table 3.2), and the total NO_3^- removal rate was also significantly higher for the Containment treatment at 6.7 compared to 2.1 g N m^{-3} d^{-1} for the Non-Containment treatment.

Retention Time Effects

There was a correlation between retention time and percent concentration reduction measured in drainage events for the Non-Containment treatment and during the preliminary tracer tests with higher removals at higher retention times (Figure 3.5). Data in Figure 3.5 show the Non-Containment treatment means for four of the five sampling events (i.e. excluding first sample point for each drainage event), Containment data after NO_3^- removal had stabilized (i.e. approximately 12 h from each drainage event's start), and the tracer data from the final sample for each bioreactor (at least 3.9 pore volumes). For Events #3 - #5, the Containment treatment NO_3^- percent concentration reduction was clearly higher than the Non-Containment's at high retention times of 12-15 h.

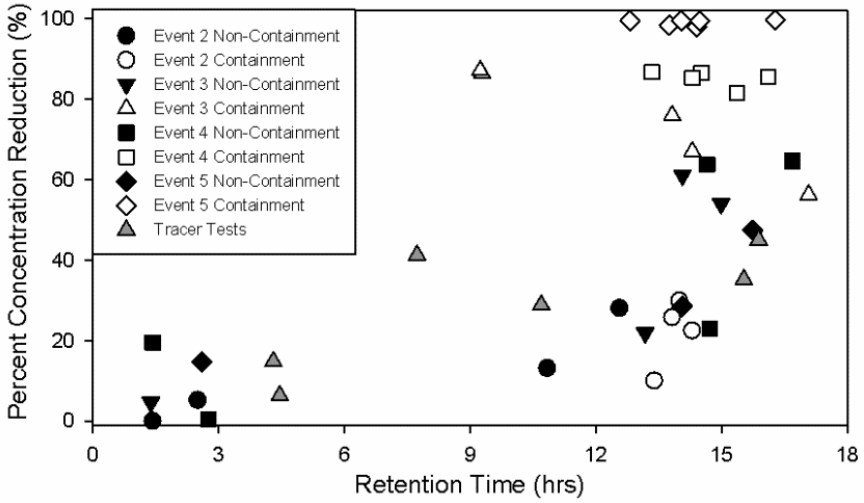

Figure 3.5 Percent concentration reduction by treatment for four drainage events and tracer tests versus retention time. Non-Containment points are mean values from four of the five sampling events (i.e. excluding first sample for each drainage event), Containment points are mean values from samples beginning 12 h from each drainage event's start, and Tracer points represent each bioreactor's final sample during the tracer tests

Temperature and Scale Effects

The temperature of the inflow solution measured in the constant head tank varied between 4.8 and 12.9°C (mean 8.8 °C) and there was no significant correlation between temperature and NO_3^- removal. However, the temperatures of solutions within the bioreactors of both treatments were less variable than the inflow solution with the temperature of the Containment treatment outflow mirroring the diurnal fluctuations of the constant head tank more so than the Non-Containment treatment. There was an observable lag in diurnal temperature fluctuation between the inflow solution and the Containment treatment solution corresponding to lags of about 12 h (almost one pore volume) for temperature peaks and about 4 h for temperature lows. At 7.7°C, the average temperature of the Event #1 was lower (not significant) than the average temperature for the tracer test (8.2°C) or for the other events which had average temperatures ranging from 8.7°C to 9.3°C. However, it is doubtful this small temperature difference played a role in the lower removal efficiency for this Event #1.

Discussion

Pilot-scale denitrification bioreactors in New Zealand showed that treating simulated drainage water at constant retention times provided higher removal efficiency and total mass removal compared to uncontrolled drainage events being treated directly. With significantly different total removals of 14.0% versus 36.9% and removal rates of 2.1 versus 6.7 g N m^{-3} d^{-1} for the Non-Containment and Containment treatments, respectively, there were clear performance differences. Removal rates were similar to other denitrification bioreactors with Woli et al. (2010) reporting 6.4 g N m^{-3} d^{-1} for a full size reactor in the U.S. Midwest. In general, removal rates for wood-based denitrification systems are less than 10 g N m^{-3} d^{-1} (Schipper et al., 2010b), though Schipper et al. (2010a) reported a range of 2-22 g N m^{-3} d^{-1} for a variety of reactors.

It was thought simulated containment would allow optimized treatment for early season drainage containing the highest NO_3^- concentrations. However in these experiments, Event #1, with the highest inflow NO_3^- concentration, provided the least NO_3^- removal. This trend may have been influenced by experimental design and slow start-up conditions though the tracer tests performed at a similarly high inflow concentration immediately prior to the first drainage event had high removal efficiencies (45.0 and 35.2%). Temperature may also have been a factor, but it is unlikely the slightly lower temperatures during these first events would have been significant enough to affect removal. Other work investigating temperature effects on denitrification has used a far greater temperature

range. Cameron and Schipper (2010) documented higher removal rates at 23°C compared with at 14°C, though NO_3^- reduction in denitrification systems has been documented in the field at temperatures as low as 2°C degrees (Robertson et al., 2000).

Regarding retention time as a design parameter, approximately 15 h of retention were required for 50% NO_3^- reduction from the Non-Containment treatment which was nearly twice what was required for similar reductions shown by Christianson et al. (2011). This difference may be partially due to the lack of steady state conditions for the Non-Containment treatment. In the Non-Containment bioreactors, NO_3^- removal may not have stabilized, thus, yielding lower percent removals. This suggests that fluctuating flow rates and retention times may result in lower percent removals even at the same retention time compared with a steady-state reactor.

Restriction of oxygen availability (O_2), as indicated by DO, is one of the most important requirements to allow denitrification to proceed (Korom, 1992). Small-scale denitrification research systems such as laboratory columns and also, potentially, pilot-scale reactors are especially susceptible to the impacts of inflow DO compared with field-scale bioreactors (Schipper et al., 2010). In wastewater treatment systems, DO above 0.2 mg L^{-1} has been reported to inhibit denitrification; DO concentrations of 0.50 mg L^{-1} can result in denitrification rates 17% of the maximum rate (Metcalf and Eddy, 2003). This indicates some denitrification was possible at DOs in the range reported here during the tracer tests (0.73 to 0.24 mg L^{-1}), though not at the maximum possible rate. Korom (1992) reported the specific DO concentration resulting in a facultative denitrifier's change in electron acceptor from O2 to NO_3^- will vary based on the specific organism and will usually be much less than their cited allowable DO concentration maximums of 2.0 mg L^{-1} (groundwater) and 6.9 mg L^{-1} (waste water). Although the DO concentration declines documented during the tracer tests here do not directly allow quantification of denitrification, denitrification was likely as NO_3^- removal was also documented. Moreover, though DO was only measured during the tracer experiment and not during the drainage events, NO_3^- removal via denitrification was the most likely explanation for the reduction in NO_3^- concentration and mass during the drainage events.

The preliminary tracer experiments indicated plug flow characteristics in the pilot bioreactors (i.e. MDIs close to 1) and also that the tracer residence time was greater than the theoretical retention time. From a design standpoint, the difference between these two values could be important as Christianson et al. (2011) noted that if in situ tracer residence times are consistently higher than theoretical retention times, design retention times could be reduced to minimize bioreactor volume and expense.

In an applied sense, the Containment treatment bioreactors provided nearly 37% NO_3^- removal over the simulated drainage season (five events with decreasing inflow nitrate concentrations) for a drainage treatment area of 236m^2 (original 910 m^2 drainage plot sized by 26%) resulting in removal of 2.7 kg NO_3^--N ha^{-1}. A given event flow volume of 1920L for this 236 m^2 area equated to a drainage depth of 8 mm meaning the series of five events treated a total of 40 mm of drainage. If this trial had been extended to simulate treatment of closer to 100mm of critical early season drainage, it is likely there would have been more events with moderate NO_3^- inflow concentrations (approximately 15 mg N L^{-1}). This means it is possible more events would have been treated at removal efficiencies greater than 50% which reduces the importance of low removal efficiencies from the first two events. Based on the theoretical study treatment area (236 m^2), the required bioreactor volume to treat a hectare of drainage area would be 19.7 m^3 ha^{-1}, though this may be a factor of this being a pilot scale study. The bioreactor surface area to treatment area ratio (0.63 m^2/236 m^2) was in range of other bioreactors in the US Midwest which tend to be about 0.1% or less of the treatment area (Christianson et al., 2009).

Conclusions

Pilot-scale bioreactors were used to treat simulated New Zealand dairy farm early season drainage consisting of five drainage events with decreasing NO_3^- concentrations. The major conclusion was that containing drainage events and providing a controlled flow through a denitrification bioreactor provides more optimized NO_3^- removal than treating drainage hydrographs directly from the field. The NO_3^- removal differences between treatments were clear with the Containment treatment providing higher removal efficiency (14.0% versus 36.9%) and higher removal rates (2.1 versus 6.7 g N m^{-3} d^{-1}) compared to uncontrolled drainage events being treated directly (i.e. Non-Containment treatment). Also, increased removal efficiency was correlated with increased retention time and fluctuating flow rates may result in lower removal efficiency even at high retention times. Lastly, field scaling of results supported calculations made in the US with regards to percentage of drainage area required for the surface footprint of a denitrification bioreactor.

In New Zealand, drainage containment prior to bioreactor treatment will allow optimized NO_3^- removal at longer and more stable retention times. This type of "pre-treatment" would be most beneficial for drainage systems characterized by flashy, pulse-flow drainage events. However, because this work simply simulated drainage containment, it did not account for changes in the physical or chemical properties of the drainage water while being contained. Also, the additional cost of creating impoundment facilities in addition to the denitrification treatment system itself must be

considered. Although this work indicates that denitrification bioreactors are a promising NO_3^- mitigation option for drainage in New Zealand, long-term and full-scale agricultural studies are needed. Field-scale evaluations of denitrification bioreactors not only in New Zealand, but in a variety of agricultural locations are necessary before this can be a viable strategy to significantly mitigate NO_3^- losses to surface water bodies.

Acknowledgements

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Dissertation Addendum

New Zealand's moderate climate means that dairy cattle are often grazed year-round (especially on the North Island) and mole drainage in these agricultural systems typically occurs during winter (e.g. June – August) (Monaghan et al. 2002). The majority of drainage N loss from New Zealand's dairy pasture systems is in the nitrate form with such losses on the order of 25-30 kg N/ha though estimates of losses from individual urine spots are as high as 1000kg N/ha (Monaghan et al., 2002; Houlbrooke et al. 2004).

A containment system of the type proposed here would likely consist of a temporary dam or diversion installed in a natural topographic basin. Here, the drainage event volume of 1920 L resulted in a drainage depth of 8 mm for this scaled experiment. Sizing the containment for this event depth would require $80m^3$ storage for one ha of drainage area. Assuming an average depth of 3m in a theoretical ravine impoundment yielded an impoundment water surface area of 26.7 m^2/ha drainage area.

The R^2 measure of correlation between retention time and percent concentration reduction for all data points in Figure 3.5 was approximately 0.34 ($y = 4.055x + 2.167$); however, individually, a linear regression for the Non-Containment data points had an R^2 of 0.64 and for the tracer tests yielded an R^2 of 0.61. Because conservation of mass was assumed for water in the bioreactors (i.e. inflow volume equaled outflow volume), this trend for percent concentration reduction in Figure 3.5 was the same for percent mass reduction. Comparison of this data with Figure 2.3 in Chapter 2 (percent mass reduction = 8.381* retention time – 3.013) showed the work in Iowa had a steeper linear regression slope indicating greater removal at a given retention time. One factor that may have affected this difference was temperature; water temperatures in the Chapter 2 Iowa study ranged from

10.5°C to 15.4°C, while inflow temperatures were 4.8°C 12.9°C in the New Zealand pilot studies in Chapter 3.

Steady state bioreactor conditions were indicated at greater than one to two pore volumes; such conditions would likely be shown by a consistent reduction in nitrate concentration holding other factors such as temperature and influent concentration constant. Here, when the Non-Containment treatment was operated at its highest retention time during most events, effluent concentrations after one to two pore volumes were similar to the Containment treatment effluent concentrations. If designing a similar experiment to specifically achieve steady state nitrate removal, no less than three pore volumes (preferably at least five) should be used before changing conditions.

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CHAPTER 4 PERFORMANCE EVALUATION OF FOUR FIELD-SCALE AGRICULTURAL DRAINAGE DENITRIFICATION BIOREACTORS IN IOWA

A paper to be submitted to the *Journal of Environmental Quality*

Laura Christianson, Alok Bhandari, Matthew Helmers, Keegan Kult, Todd Sutphin, and Roger Wolf

Abstract

Recently, interest in denitrification bioreactors to reduce the amount of nitrate in agricultural drainage has led to increased installations across the US Midwest. Despite this recent attention, there are few peer-reviewed, field-scale comparative performance studies investigating the effectiveness of these denitrification bioreactors. The object of this work was to analyze nitrate removal performance from four existing bioreactors in Iowa with particular attention paid to potential performance-affecting factors including: retention time, influent nitrate concentration, temperature, flow rate, age, length to width ratio, and cross-sectional shape. Based on a minimum of two years of water quality data from each of these four bioreactors, annual flow-weighted nitrate-nitrogen concentration reductions ranged from 12% to 75% with a mean of 43% from all sites in all years. Bioreactor and total (including bypass flow) nitrate-nitrogen load reductions ranged from 12% to 76% (mean 45%) and 12% to 57% (mean 32%), respectively. Statistical modeling showed temperature and influent nitrate concentration were the most important factors affecting percent bioreactor nitrate load reduction and nitrate removal rate, respectively. Modeling also indicated load reductions within the bioreactor were significantly impacted by retention time at three of the four reactors; a retention time effect on nitrate concentration reduction was especially evident during and after elevated drainage flow events at one of the sites. More field-scale performance data from bioreactors of different designs and from multiple locations around the Midwest are necessary to further enhance understanding of nitrate removal in these systems and their potential to positively impact water quality.

Introduction

Local water quality problems in the US Midwest combined with concerns about the hypoxic zone in the Gulf of Mexico (IDNR, 2006; McMullen, 2001; Turner and Rabalais, 1994; USEPA, 2007) require new approaches to improve agricultural drainage water quality. Nitrate-nitrogen (NO_3^--N) loadings, one of the main contaminants of concern in agricultural drainage, can be reduced using a number of in-field and edge-of-field approaches. However, in light of the United States

Environmental Protection Agency's (USEPA) call for a 45% reduction in nitrogen in the Mississippi river, a combination of multiple approaches will be necessary (Dinnes et al., 2002; USEPA, 2007).

Denitrification bioreactors are a new remediation technology to reduce the amount of nitrate (NO_3^-) in agricultural drainage that have preliminarily been successful in agricultural systems. The provision of additional carbon and maintenance of saturated conditions facilitates this "enhanced denitrification" process. In the US Midwest, a handful of bioreactors have been installed in recent years, and lately, interest in these systems has grown as evidenced by increased publicity in mass media (e.g. Ag PhD, 2010; Caspers-Simmet, 2010) and in scientific literature (e.g. special issue of *Ecological Engineering* on managed denitrification in 2010, Strock et al., 2010).

Despite this interest in denitrification systems for NO_3^- removal, there are few peer-reviewed, field-scale performance studies investigating the effectiveness of agricultural drainage denitrification bioreactors. One of most comprehensive local studies is from Jaynes et al. (2008) who showed a denitrification system removed 55% of the NO_3^- load in agricultural drainage averaged over five years; however, this work utilized a different design from most current drainage denitrification bioreactors. According to a review by Schipper et al. (2010b) who defined terminology for these denitrification systems, Jaynes et al.'s (2008) site could be termed "denitrification walls" rather than "denitrification beds" (or bioreactors) which could be important as bioreactors may have higher removal rates than walls (Schipper et al., 2010b). Newer work by Woli et al. (2010) and Verma et al. (2010) from Illinois showed annual bioreactor load reductions of 23% to 98%. This work leaves additional performance questions as two of the bioreactors under investigation had significantly different performance with no outflow documented from one of the reactors. Other field-scale work investigated hydraulic modeling (Chun et al., 2010) or provided performance data from early designs that differed from more current control-structure based bioreactors (Van Driel et al., 2006). More continuous performance data from a number of denitrification bioreactors is needed for a more robust understanding of the potential contribution of these systems to water quality efforts.

Randall and Goss (2001) described controllable and uncontrollable factors for NO_3^- leaching from drainage, and it is thought there are similar controllable and uncontrollable factors affecting bioreactor performance. A primary "controllable" design factor is bioreactor length to width ratio (L: W). Bioreactors in the Midwest have tended to be long and narrow (i.e. high L: W) with the exception of bioreactors described in Chun et al. (2010) which was square (L: W = 1). In one design model, the retention time was highly dependent upon the length of the bioreactor meaning that many resulting designs had L: Ws of around 10 (length at least 30 m) (Christianson et al., 2011a). There has been little discussion in the literature about the effect of this ratio on performance. Another controllable

design parameter is cross-sectional shape. Christianson et al. (2010) found there was no difference in NO_3^- removal between a trapezoidal cross-section versus a rectangular cross-section in pilot scale denitrification bioreactor experiments. Though several trapezoidal cross-section denitrification systems have been installed (Christianson et al., 2009; Schipper et al., 2010a), the specific design effect of cross-sectional shape has not been investigated at the field scale.

Retention time, or the relationship between the media porosity, active flow volume and flow rate through the reactor, is a performance parameter that combines controllable design factors with uncontrollable environmental elements. The selection of fill media and the design dimensions of the bioreactor are controllable but the variable flow rate, and to some extent, depth of water in the reactor make designing for a specific retention time challenging (Christianson et al., 2011a; Woli et al., 2010). The USDA NRCS interim design standard for denitrifying bioreactors in Iowa specifies a retention time that allows sufficient reduction in NO_3^- concentration (USDA NRCS, 2009), however, an "adequate" retention time may most likely vary based on hydraulic loading and temperature (Christianson et al., 2011a).

Significant environmental factors that may affect performance include influent NO_3^- concentration, hydraulics, temperature, and bioreactor age. There has been discussion in literature regarding the impact of influent NO_3^- concentration on removal with some indicating NO_3^- removal rates will be constant regardless of concentration (zero order reaction) (Gibert et al., 2008; Robertson, 2010), and others indicating increasing NO_3^- concentrations will increase the removal rate (first order reaction) (Chun et al., 2009). Robertson (2010) noted NO_3^- removal followed zero order kinetics due to insensitivity to influent NO_3^-concentrations; the reaction may be controlled by an independent parameter like labile carbon availability. This carbon availability can be impacted by bioreactor hydraulics with Woli et al. (2010) noting several dry periods in a bioreactor may have precipitated greater labile carbon availability and thus high removal for the subsequent high flow events. In situ temperatures can also be important with NO_3^- removal typically increasing by a factor of approximately 2 for every 10°C increase in temperature (i.e. $Q_{10} \approx 2$) (Cameron and Schipper, 2010; Warneke et al., 2011). Greenan et al. (2009) performed lab scale denitrification experiments at 10°C to simulate drainage water temperatures, and though this is a good yearly average approximation for drainage water, bioreactor water temperatures can vary from just above freezing to greater than 15˚C. Finally, longevity can impact performance, but there seem to be no field-scale bioreactors in operation long enough to have failed due to carbon exhaustion (Schipper et al., 2010b). Robertson (2010) found that NO_3^- removal was similar between new chips and seven year old chips which had been removed from a functioning bioreactor. Moorman et al. (2010) and Long et al. (2011)

documented denitrification systems had sufficient carbon to continue operation after nine and fourteen years, respectively.

Though a number of bioreactors are now in operation in the U.S. Midwest, there is a lack of comprehensive peer-reviewed evaluation of performance from multiple sites. Past work has highlighted the potential for promising NO_3^- removal from these systems, but performance optimization and prediction requires more advanced analysis and modeling techniques. The object of this work was to analyze NO_3^- removal performance from four existing bioreactors in Iowa with particular attention paid to the factors affecting performance (retention time, L:W, cross section, influent concentration, temperature, age, and flow rate). A second objective was to utilize statistical modeling to identify the environmental and design factors most affecting their NO_3^- removal performance. It was hypothesized that denitrification bioreactor design parameters and field conditions affect *in situ* performance. Lastly, a cost analysis was included to allow economic comparisons of bioreactors with other water quality technologies.

Methods

Four denitrification bioreactors in Iowa, each with a different design and drainage treatment area, were used for this comparison (Table 4.1). Greene Co. and Hamilton Co. bioreactors were monitored by the Iowa Soybean Association/Agriculture's Clean Water Alliance (ISA/ACWA) while the North East Research and Demonstration Farm (NERF) and the Pekin bioreactors were monitored by Iowa State University researchers. The parameters under analysis included length to width ratio (L: W), cross sectional shape, flow rate, temperature, age, and NO_3^--N influent and effluent concentration with the derivative factors of retention time and NO_3^- removal rate calculated. Contributing areas (i.e. drainage treatment areas) were determined based on knowledge of the existing tile drainage network; however, when unknown (e.g. NERF), contributing area was assessed based on estimated annual subsurface water transmission. Although the drainage area for each bioreactor site was not exactly know, parameters such as kg N lost/ha and water drainage depths were nevertheless calculated to allow comparisons between this and other field-scale studies; note, small errors in the drainage area estimation could have significant impact upon load values.

Table 4.1 Description of four bioreactors in Iowa used in investigation

Bioreactor	Location	Year Installed	Drainage Treatment Area (ha)	L (m)	W (m)	D (m)	Vol. (m^3)
Pekin	Southeast Iowa	August 2002	1.3	30	0.5	1.2	18
NERF	Northeast Iowa	April 2009	14.2	36.6	4.6 top, 2.4 bottom	1.0	128
Greene Co.	Central Iowa	August 2008	19.0	15.2	7.6	1.1	127
Hamilton Co.	Central Iowa	June 2009	20.2	30.5	3.7	0.9	102

Pekin, Iowa

As one of the oldest denitrification bioreactors in the state, the bioreactor in Pekin, Iowa yielded this study's longest data record. Installed in August 2002, it was filled with a mixture of gravel and woodchips and received drainage from a research plot, hence the small treatment area of 1.3 ha (plots of 300 x 460 ft). In addition to the gravel used in the fill, this reactor differed from the other three bioreactors in that it only had one control structure on the inlet side rather than two structures and did not have a bypass line. The bioreactor likely receives lateral flow from neighboring research plots. Drainage from the research plot was routed through the inlet structure into a sump from where it was pumped into the bioreactor (i.e. the single inlet control structure was not used to divert flow as at other sites). Bioreactor outflow free-flowed into an outlet sump where it was pumped through a flow meter (Neptune T-10 meters for both inflow and outflow). Flow proportional samples were collected from both sumps from late spring through late summer from 2005 to 2011 with sampling procedures described by Lawlor et al. (2008). Nitrate-N analysis for this site was done using second-derivative spectroscopy in the Wetland Research Laboratory at Iowa State University (Crumpton et al., 1992).

Northeast Research and Demonstration Farm (NERF)

Located in Northeast Iowa, the 100% woodchip NERF bioreactor was installed in April 2009 with a trapezoidal cross-section. The NERF has been the location of a number of agricultural field studies since 1976 and the 14.2 ha NERF bioreactor drainage area was in a corn and soybean rotation during the period investigated here. After installation, all drainage flow was routed to bypass the reactor until flow monitoring equipment was installed in October 2009. Bypass flow depth in the inflow structure and bioreactor flow depth in the outflow structure were continuously logged with pressure transducers (Global Water Instrumentation, Inc. WL16 Water Level Loggers from October 2009 to April 2011; Solinst Levelogger Junior from April 2011 to August 2011). Outflow control structure transducer data was used for both bioreactor inflow and outflow values by assuming bioreactor inflow equaled bioreactor outflow. Transducer depth data was reduced to daily average values to increase data workability and to allow synchronization with sample event collection days. These daily transducer depths were adjusted based on stop log height in the structures to give flow depth over the stop logs. During periods of pressure transducer logging failure, depths logged by area velocity meters (Teledyne ISCO 2150 area velocity module) installed upstream of both structures were used. Flow equations developed by Chun and Cooke (2008) (Equation 4.1) for 15 cm control structures were used to convert flow depths to flow rates for data until 8 April 2011 when 45° v-notch

weirs were installed in the structures and a corresponding v-notch weir flow equation was used (Equation 4.2).

$$Q = 0.02(L - 0.437H)H^{1.48} \quad for\ H \leq 0.44L \quad OR \quad Q = 0.027LH^{1.2} \quad for\ H > 0.44L \qquad \text{Equation 4.1}$$

$$Q_{v1} = 415.4 \times (h_w + 0.0519)^{2.5} \qquad \text{Equation 4.2}$$

where Q was the flow rate in the structure (L/s), L was the stop log width (cm), and H was the flow depth above the stop log (cm), Q_{v1} was the discharge over the weir (gpm), and h_w was the head over the weir (ft). The coefficients in Equation 4.2 were developed through calibration of the v-notch weir (Heikens, 2011).

The by-pass stop logs in the inflow structure and the capacity control stop logs in the outflow structure were periodically managed during the study. In flow calculations, the total allowable flow into the system was capped based on the drainage pipe size and estimated tile slope in the field. Conservation of NO_3^--N mass in the by-pass line was assumed for this and the following two reactors. The total inflow and outflow loads consisted of the inflow bioreactor load plus the bypass load and the outflow bioreactor load plus the bypass load, respectively.

Grab samples from the control structures were collected by the farm staff at least twice weekly and were analyzed in the Iowa State University Agricultural and Biosystems Engineering Water Quality Research Laboratory (ISU ABE WQRL) for NO_3^--N + NO_2^--N using a Cd-reduction method (Lachat Quick-Chem 8000 automated analyzer). Additionally, sulfate samples were analyzed in the ISU ABE WQRL using the Hach® sulfate method 8051 (USEPA SulfaVer 4 method; barium sulfate precipitation). Water temperature of the samples was recorded immediately after sample collection from the structures with a handheld digital thermometer (Fisher Scientific Thermometer).

Greene County, Iowa

The Greene Co. bioreactor was installed in summer 2008 in central Iowa with the lowest L: W in this comparison. The 19 ha drainage treatment area was continuously cropped in a corn and soybean rotation and the bioreactor was fed by a 30 cm tile pipe. Logging pressure transducers (Agri-Drain solar powered logging system) in the inflow and outflow structures were used to determine bypass and bioreactor flow, respectively. On selected sampling dates, a five gallon bucket and stop watch were used to verify outflow rate and the depth of water in the structures was manually checked. In flow calculations, these manual bucket and depth measurements were used as calibration points, and where transducer data was missing, the manual water depth measurements were used with a

linear interpolation to estimate flows. For example, the inflow transducer stopped working in March of 2010 and was not replaced until January 2011, thus this data was interpolated.

A 45˚ v-notch weir was installed in the structures on 31 March 2010; equations from Chun and Cooke (2008) for 30 cm control structures (Equation 4.3) were used until this date while Equation 4.4 was used to calculate flow when the v-notch weirs were in place.

$$Q = 0.02(L - 0.74H)H^{1.48} \quad for\ H \leq 0.27L \quad OR \quad Q = 0.021LH^{1.37} \quad for\ H > 0.27L \quad \text{Equation 4.3}$$

$$Q_{v2} = 4.28C_e \tan\left(\frac{\theta}{2}\right) \times (h_1 + k_h)^{\frac{5}{2}} \quad \text{Equation 4.4}$$

where Equation 4.3 components were as described by Equation 4.1 and where Q_{v2} was the discharge over the weir (ft^3/sec), C_e was an effective discharge coefficient, θ was the v-notch angle in degrees, h_1 was the head over the weir (ft),and k_h was a head correction factor. For a 45° v-notch weir, C_e and k_h were approximately 0.58 and 0.005, respectively, with the original equation (Equation 4.4) based in English units (USBR, 2001). A compound weir equation was used for several dates in June and August 2010 when the flow depth was greater than the depth of the "v". This calculation allowed flow calculation for the full "v" height (16 cm) with the additional flow calculated by Equation 4.3 for the marginal depth above this "v" height. In addition to analysis for NO_3^-, the Greene and Hamilton Co. bioreactor grab samples were analyzed for pH, dissolved oxygen (DO), nitrite (NO_2^-), and sulfate at the Des Moines Water Works.

Hamilton County, Iowa

The Hamilton Co. bioreactor was installed in central Iowa in 2009 with similar surface dimensions to the NERF bioreactor though this reactor utilized a rectangular cross-section and received drainage from a larger area than the NERF site. Cropping patterns, bioreactor flow monitoring and calculations, and sample analyses were similar to the Greene Co. site except with 15 cm structures used here rather than 30 cm structures. Chun and Cooke (2008) flow equations (Equation 4.1) were used for the pressure transducer data until 19 August 2010 when 45 ˚v-notch weirs were installed (Equation 4.4). After removing several periods of bioreactor flooding in 2010 from the dataset, there was no need for compound weir calculations.

Performance Modeling

Statistical modeling of the four bioreactors was done using a regression procedure in a statistical software package (SAS™ Proc Reg). A regression model describing the percentage load reduction and a regression model describing the removal rate were developed for each site.

Independent factors in both models included retention time, influent $NO3^-$-N concentration, influent water temperature, flow rate, and bioreactor age. Retention time was calculated as the active flow volume multiplied by an assumed porosity of 0.6 for all sites (Ima and Mann, 2007) divided by the reactor flow rate. The active flow volume was based up on the flow depth assuming a linear head difference between the water depths in the inflow and outflow structures. The reactor flow rate was the incremental difference in outflow volume between two sampling events divided by the change in time between the events. Because nitrate samples were not collected every day, daily incremental flow volumes occurring after the previous sample event and including the day of the sample event of interest were summed; this cumulative flow volume was used with the latter sample concentration for the mass NO_3^--N calculation at that latter date. Area-based loads (kg N/ha) were calculated by dividing the mass load from each sample date by the drainage treatment area. Percentage load reduction was calculated by dividing the difference of the inflow and outflow loads by the inflow load with a similar procedure used for calculating percent concentration reduction. Removal rate (g $N/m^3/d$) consisted of the mass of NO_3^--N removed between two sampling events divided by the entire bioreactor volume and the difference in days between sampling events. For calculation of annual summary percent reduction and removal rate values, the annual summed inflow and outflow loads and the difference between the first and last sample date for each year were used.

Two regression models (percent load reduction and removal rate) were additionally created for a combined dataset from all four reactors. These comprehensive models included the above independent factors as well as the L:W and a factor for cross-section shape. Regression procedure results included parameter estimates for each of these independent factors along with an indication of model fit (R^2). Significance of each independent parameter in the site specific models was determined at $\alpha = 0.01$, 0.05, and 0.10. In the combined dataset models, a stepwise selection procedure was used to eliminate independent parameters from the model unless they were significant at the $\alpha = 0.05$ statistical level.

Results and Discussion

Nitrate Removal

The influent flow-weighted NO_3^--N concentrations were generally lowest in the Pekin bioreactor (annual means of 1.23 mg NO_3^--N/L to 8.54 mg NO_3^--N/L) with this bioreactor also having the three lowest mean flow-weighted effluent concentrations (0.63, 1.31, and 1.89 mg NO_3^--N/L) (Table 4.2). The annual mean flow-weighted influent concentrations were fairly comparable at the other three sites ranging from 7.70 mg NO_3^--N/L to 15.18 mg NO_3^--N/L. Influent values usually peaked in

summer months at greater than 15 mg NO_3^--N/L (Figure 4.1 b-d, non-flow-weighted concentrations). Nevertheless, effluent concentrations at the Hamilton Co. and the Greene Co. site were less than 10 mg NO_3^--N/L for all but one sample at each site (13 May 2010 and 27 June 2011, respectively). Effluent concentrations from the NERF bioreactor exceeded this 10 mg NO_3^--N/L maximum contaminant level for NO_3^- (USEPA, 2011) more frequently which was also reflected in the elevated NERF annual mean flow-weighted effluent concentrations (8.51 mg NO_3^--N/L and 11.62 mg NO_3^--N/L) compared to the other sites. Percent flow-weighted concentration reduction for samples from all sites and years spanned 11.9% to 75.2% (mean 43.0% ±21.3%) (Table 4.2).

Table 4.2 Annual mean influent and effluent NO_3^--N concentrations and loads by bioreactor or total (including bypass) for four denitrification bioreactors in Iowa

		Nitrate-N Concentration†					Nitrate-N Load					Water Depth		
	Water Year	Mean In (mg NO_3^--N / L)	Mean Out (mg NO_3^--N / L)	Mean Reduction %‡	Bioreactor In (kg N/ha)	Bioreactor Out (kg N/ha)	Mean Bioreactor Reduction %‡	Removal Rate (g/m³/d) §	Total In (kg N/ha)	Total Out (kg N/ha)	Mean Total Reduction %‡	Bioreactor (cm)	Total (cm)	Depth Treated %
Pekin	2004-2005	4.21	1.89	55.2	5.0	2.8	43.7	1.07				11.9¶¶	14.6††	81.7
	2005-2006	4.98	2.22	55.5	1.2	0.7	43.5	0.75				2.3¶¶	2.6†††	88.6
	2006-2007	8.54	4.66	45.4	33.6	21.1	37.4	3.78				39.4¶¶	49.5††	79.5
	2007-2008	2.86	2.46	14.0	14.8	8.4	43.8	2.53		NA§§		51.9¶¶	51.6††	101
	2008-2009	3.84	2.49	35.2	7.1	5.0	29.1	0.57				18.5¶¶	20.6††	89.8
	2009-2010	1.23	0.63	49.1	7.4	5.8	22.0	0.67				60.5¶¶	56.2††	108
	2010-2011	1.88	1.31	30.5	2.0	0.5	74.0	0.38				10.5¶¶	2.3†††	456
NERF	2009-2010	9.93	8.51	14.3	34.7	29.7	14.6	1.56	37.3	32.2	13.6	34.0	37.4	90.9
	2010-2011¶	13.18	11.62	11.9	21.4	18.9	11.7	0.86	21.9	19.4	11.5	18.1	18.4	98.6
Greene Co.	2008-2009#	15.18	4.97	67.2	20.2	6.5	68.0	7.76	41.4	27.6	33.3	20.9	39.2	53.4
	2009-2010	7.70	4.67	39.4	33.6	18.1	46.0	6.69	50.1	34.6	30.9	44.6	65.2	68.4
	2010-2011††	9.55	6.18	35.2	1.6	0.8	50.4	0.41	2.9	2.1	27.3	1.5	3.0	50.8
Hamilton Co.	2009-2010	7.74	1.92	75.2	10.8	2.6	75.7	5.02	14.4	6.3	56.6	16.2	18.7	87.0
	2010-2011‡‡	9.59	2.47	74.3	0.8	0.2	73.9	0.42	1.2	0.6	48.6	0.9	1.2	73.2

† Flow weighted concentrations

‡ Mean % reduction calculated as reduction between mean inflow/outflow concentrations or loads, not the mean of reductions of individual sample events

§ Based on entire reactor volume and the time between sample events

¶ Through 22 August 2011

No flow monitoring until 1 January 2009

†† Through 25 July 2011

‡‡ Through 6 July 2011

§§ Not applicable as the Pekin bioreactor had no by-pass; Pekin bioreactor water depth was based on outflow

¶¶ For Pekin bioreactor, this was bioreactor inflow depth

†††For Pekin bioreactor, this was bioreactor outflow depth

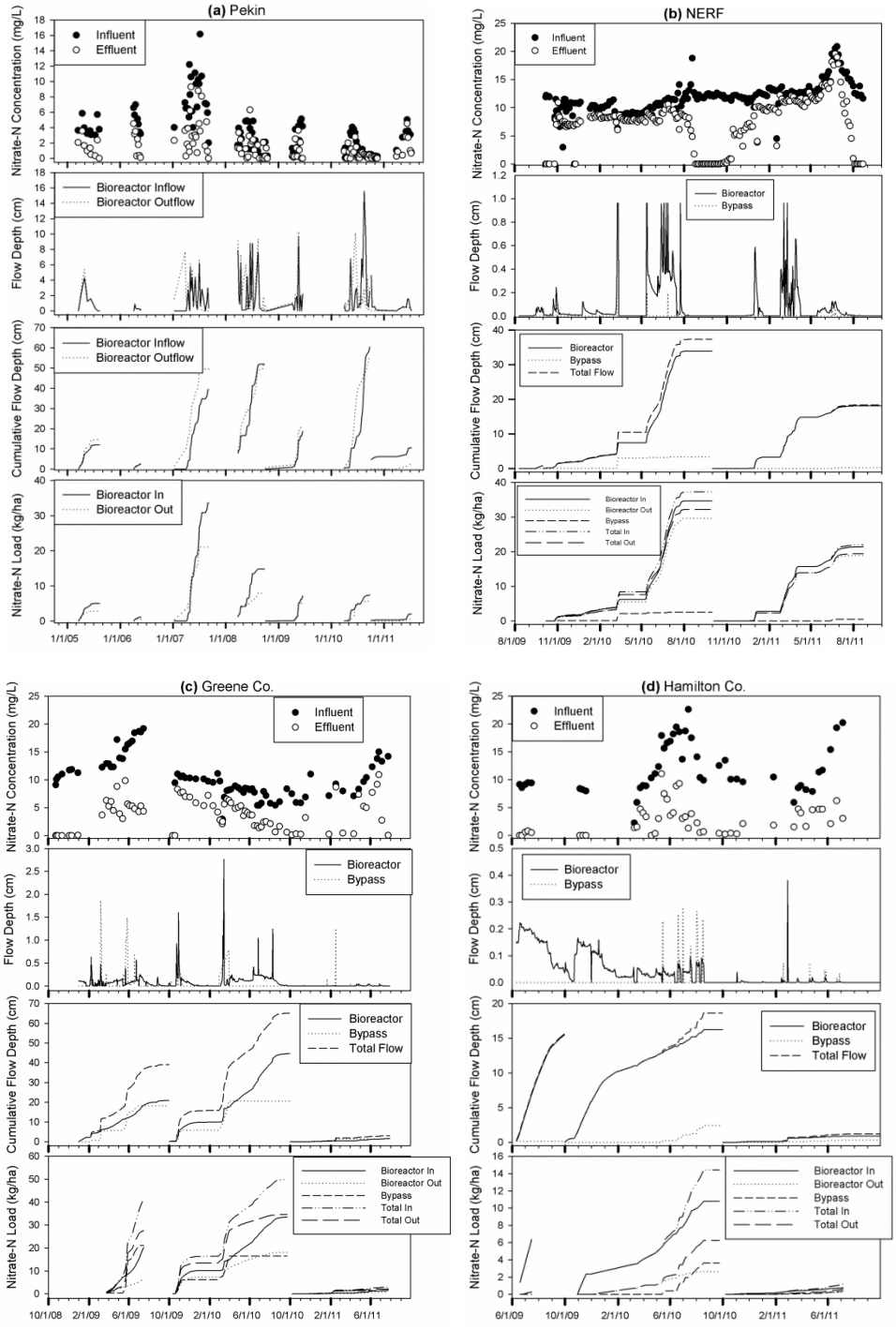

Figure 4.1 Influent and effluent NO_3^--N concentrations, bioreactor and bypass flows, and cumulative NO_3^--N loads for four bioreactors in Iowa; flow depths shown were normalized by drainage treatment area (i.e. are not the depths over weir); Note different scales on axes

Bioreactor influent loads ranged from 0.8 kg N/ha to 34.7 kg N/ha while effluent loads were between 0.2 kg N/ha and 29.7 kg N/ha for all four bioreactors in all years (Table 4.2, Figure 4.1). Load reductions for bioreactor flows ranged from 11.7% to 75.7% (mean of all site years 45.3% ± 21.6%). When the by-pass flow volume at the sites was considered in addition to the bioreactor flow volume (i.e. "Total" loads as opposed to "Bioreactor" loads), the resulting total inflow and outflow loads were 1.2 kg N/ha to 50.1 kg N/ha and 0.6 kg N/ha to 34.6 kg N/ha, respectively (Table 4.2, Figure 4.1). The total percent load reductions were lower than the bioreactor flow-only percent load reductions because the by-pass flow was untreated; total loads were reduced by 11.5% to 56.6% (mean 31.7% ± 16.7%). The Hamilton Co. bioreactor had the highest total percent load reductions at greater than 48% in both years, though this only equated to 8.1 kg N/ha and 0.6 kg N/ha removed. The Greene Co. reactor had the highest total area-based load removal of 15.5 kg N/ha in 2009-2010; however, this was only a 30.9% total load reduction.

Because the Pekin bioreactor in-and outflows were pumped through flow meters and there was no by-pass flow, these inflow and outflow depths rather than bioreactor and total depths were shown in Table 4.2; lateral seepage at the site likely accounts for the discrepancy between these values in the percent flow treated column. Neglecting this site, the NERF bioreactor treated the highest percentages of water (greater than 90%), though this site had the lowest percentage bioreactor and total load reductions (Table 4.2). The Hamilton Co. bioreactor also treated the majority of drainage water in both years of performance (greater than 73% treated), but was able to treat these waters with higher load reductions than at the NERF site. This indicates that while it is useful to route as much drainage as possible though the bioreactor, there may be a useful comprise between treating slightly less water at a better treatment rate. The impact of treating too little of the drainage can been seen with data from the Greene Co. reactor. This reactor treated between 50.8% and 68.4% of drainage in three years of operation which greatly reduced its treatment efficiency from 46% to 68% (bioreactor load reduction) to less than 34% total load reduction. Note, rainfall in the 2010-2011 water year was lower than the long term averages at the Greene Co. and Hamilton Co. sites which may account for the low amount of drainage in those site-years.

In addition to the percentage of drainage water treated, reactor hydraulics were another important consideration for bioreactor performance. Based on preliminary tracer testing at the NERF site, flow short circuiting within the reactor was very likely (L. Christianson, Unpublished data). Short circuiting causes a portion of the drainage to remain in the bioreactor for a shorter period than

indicated by the theoretical retention time, thus decreasing the reactor's NO_3^- removal potential. This furthermore accounts for the poor performance at the NERF bioreactor.

The high frequency of grab sampling at the NERF bioreactor allowed documentation of individual flow event impacts on bioreactor NO_3^- concentration reduction. Following several drainage events in March 2010, June/July 2010, and June/July 2011 bioreactor effluent concentration values began to noticeably match their influent counterparts (Figure 4.2 a and b). In pilot-scale experiments, this decreased NO_3^- concentration reduction has been shown to occur during simulated storm events due to decreased retention time in the reactor (Christianson et al., 2011b; Christianson et al., 2011c). This was also thought to be the case in this field-scale work as percent concentration reductions for each of these three events were positively correlated with retention time (R^2=0.42, p value < 0.0001)(Figure 4.2 c).

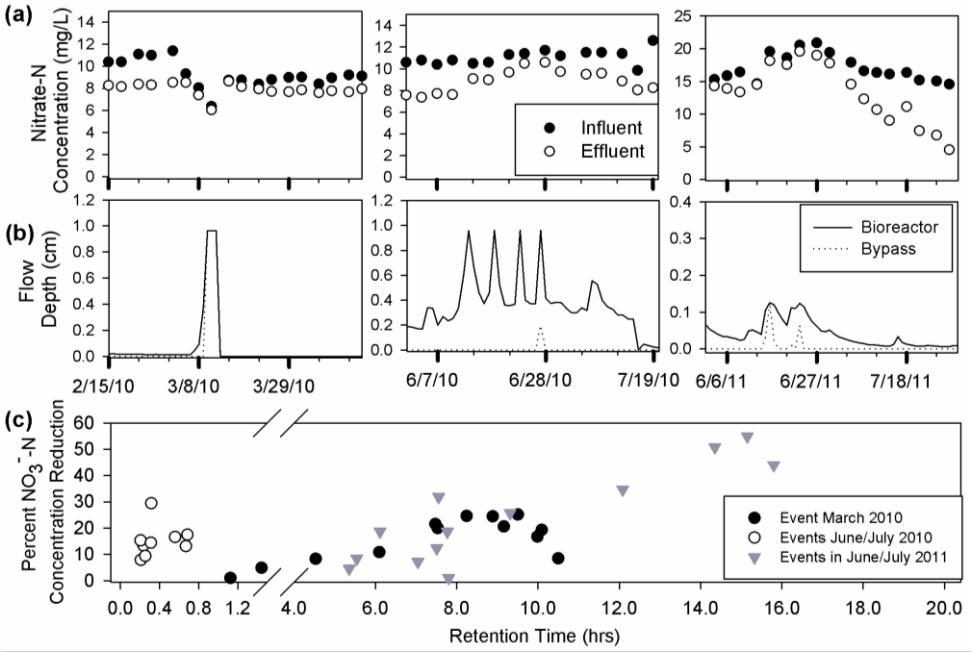

Figure 4.2 NERF bioreactor drainage event detail showing influent and effluent concentrations (a), flow depths (b), and a correlation of percent concentration reduction with theoretical retention time for each event (c)

Annual bioreactor NO_3^- removal rates ranged from 0.38 g N/m^3/d to 7.76 g N/m^3/d (Table 4.2). These values were similar to the range of published literature with a review by Schipper et al. (2010b) detailing NO_3^- removal rates of 2 to 22 g N/m^3/d for a variety of denitrification bed systems. More specific to drainage treatment, Christianson et al. (2011c) reported rates of 3.8 to 5.6 g N/m^3/d in pilot-scale work and Woli et al. (2010) reported a rate of 6.4 g N/m^3/d for a field-scale bioreactor in Illinois.

Additional Parameters

Bioreactor water temperature for all sites peaked in late summer months at typically greater than 15°C and was at its lowest around March of each year at less than 3°C (Figure 4.3). The annual flow weighted inflow temperature ranged from 6.08°C to 8.69°C (mean 7.09°C) for these three sites in these years. Like temperature, bioreactor influent DO fluctuated annually with highest influent DO in early spring months (≥ 8.5 mg DO/L) and lowest in summer (mid-July through late August, typically < 5 mg DO/L) (Figure 4.3). Regardless of influent DO concentration, this parameter was always reduced to less than 2.4 mg DO/L (and usually much less) indicating conditions conducive to denitrification were present within these bioreactors.

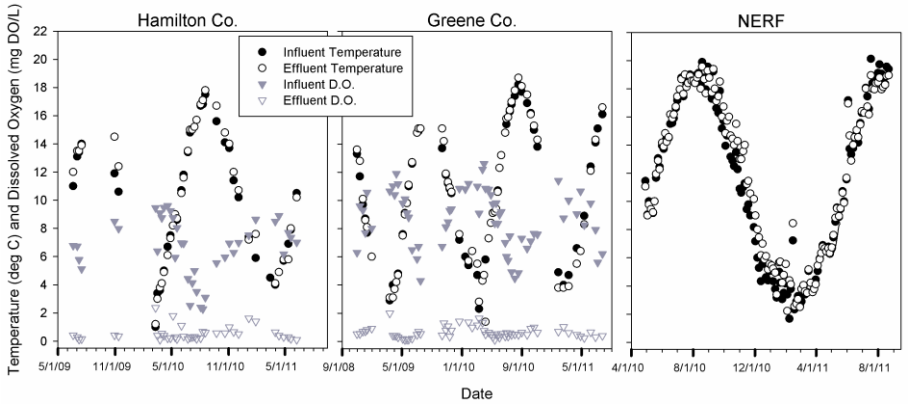

Figure 4.3 Influent and effluent temperature from three denitrification bioreactors and influent and effluent DO from two bioreactors in Iowa

Elevated NO_2^- concentrations were noted in effluent samples from the Hamilton Co. and Greene Co. bioreactors (Figure 4.4). These concentrations were never more than 2 mg NO_2^--N/L but could present a water quality issue as NO_2^- causes health concerns similar to NO_3^-. The maximum contaminant level for NO_2^- in drinking waters is 1 mg NO_2^--N/L (USEPA, 2011).

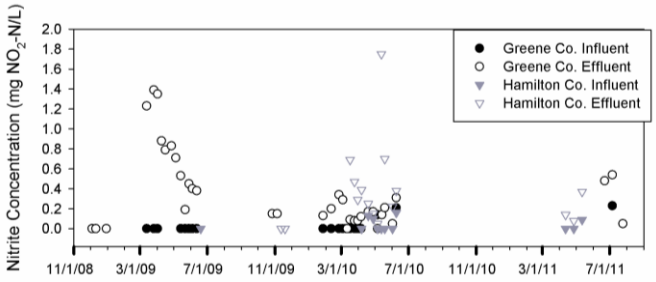

Figure 4.4 Influent and effluent nitrite-N concentrations from Greene Co. and Hamilton Co. bioreactors

Sulfate reduction was also documented in the Hamilton Co., Greene Co., and NERF bioreactors though not continuously during all sample events (Figure 4.5). Sulfate reduction is due to an excess of reducing capacity in the reactors once the influent NO_3^- is removed (Blowes et al., 1994). This process was most notable in winter months in the Hamilton and Greene Co. reactors (November 2009 and December 2008, respectively) when influent NO_3^- was reduced to nearly zero from concentrations of approximately 8 mg NO_3^--N/L and greater than 11 mg NO_3^--N/L at the two sites, respectively. More continuous sulfate reduction was documented in the NERF bioreactor from late August to mid October 2010 when influent NO_3^- was similarly reduced. The low flow rate though this reactor at this time (Figure 4.1 b) was indicative of high retention times and thus complete NO_3^- reduction and subsequent sulfate reduction.

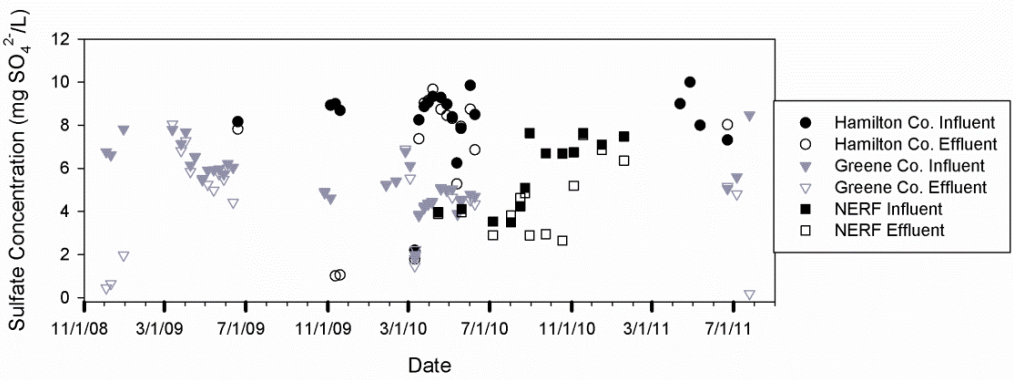

Figure 4.5 Influent and effluent sulfate concentrations for three bioreactors in Iowa

Modeling Results

Individual regressions models of the four bioreactors revealed that the dependent parameter percentage bioreactor load reduction was most strongly positively correlated with temperature (correlation at $\alpha = 0.01$ significance for two of three reactors where this parameter was measured) (Table 4.3 and Figure 4.6 a). Retention time was also noticeably correlated (Figure 4.6b) though only one site, NERF, was strongly significant ($\alpha = 0.01$ level; Table 4.3). Bioreactor age had a significant impact upon percent load reduction for three of the four bioreactors; this factor did not have a consistently positive or negative effect which confounded the importance of this factor (Table 4.3 and Figure 4.6c). Flow rate (Table 4.3 and Figure 4.6d) and influent NO_3^- concentration were only significant parameters at the Pekin and NERF sites, respectively.

Table 4.3 Percent load reduction regression model parameters for four bioreactors in Iowa

	Model Intercept	Retention Time (h)	Influent Concentration (mg NO_3^--N/L)	Temperature (°C)	Flow Rate (m^3/h)	Age (months)	R^2
Pekin	26.88	0.39§	0.44	NA	-5.04‡	0.41‡	0.34
NERF	17.53	0.73†	-4.68†	1.81†	-0.31	2.02†	0.74
Greene Co.	50.68†	0.74‡	-1.47	4.74†	-0.57	-1.78‡	0.49
Hamilton Co.	77.58†	0.04	-0.49	2.26§	-0.52	-1.40	0.36

† indicates significance at α=0.01
‡ indicates significance at α=0.05
§ indicates significance at α=0.10

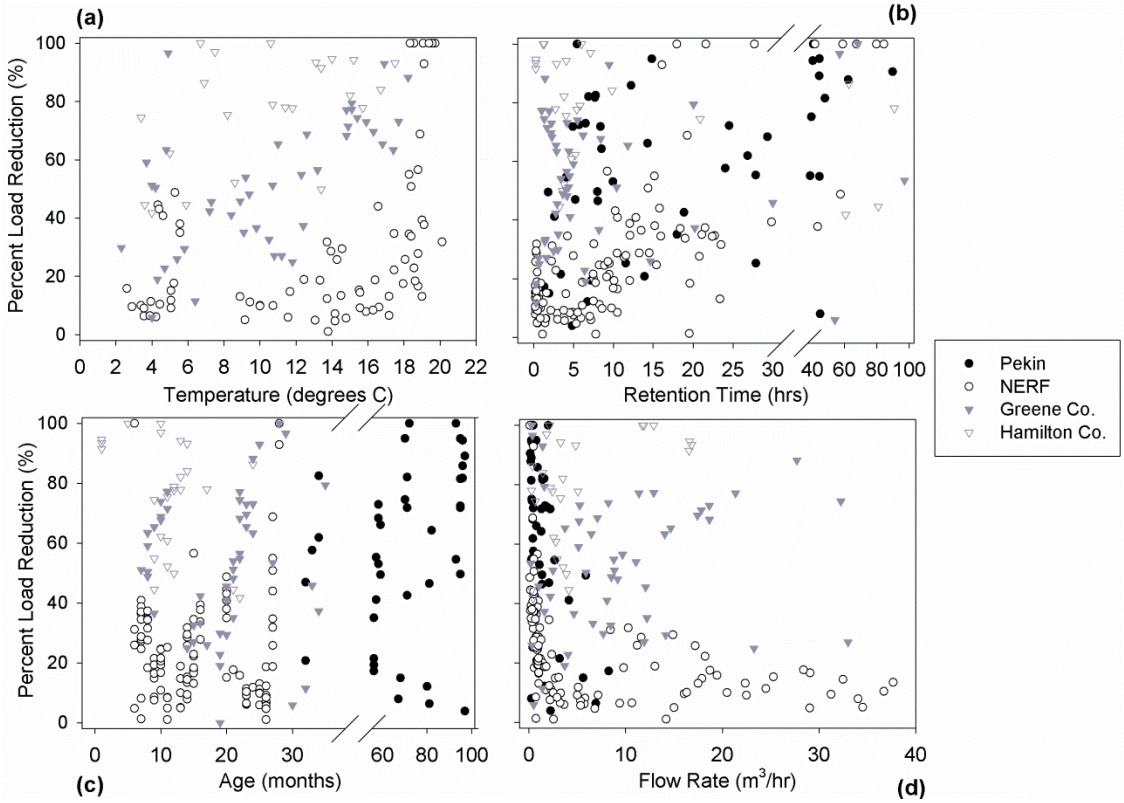

Figure 4.6 Percent bioreactor load correlations with temperature (a), retention time (b), age (c), and flow rate (d) for four bioreactors in Iowa

The NO_3^- removal rate regression models showed this parameter was most significantly affected by flow rate and influent NO_3^- concentration (Table 4.4 and Figure 4.7 a and b). Flow rates were strongly significant at the Green Co., Hamilton Co., and NERF sites (α = 0.01), while the influent concentration had the same level of significance at only the Green Co. and Hamilton Co. reactors (Table 4.4). Importantly, this dependence of removal rate upon flow rate was likely an artifact of calculation as similar original raw data were required in the computation of both parameters. The positive correlation of removal rate with influent concentration at at least the α =

0.05 level for three of the bioreactors may help clarify reaction kinetics. The parameter estimates for these three reactors indicated that a one mg NO_3^--N/L increase in influent concentration increased the removal rate by 0.44 to 1.25 g N/m^3/d assuming other parameters were held constant. This nearly proportional 1:1 relationship (on average 1:0.9) points strongly to first order kinetics for this data. This reaction order dictates a reaction rate which is dependent up on the availability of the reactant with any increase in reactant (i.e. NO_3^--N) availability proportionally increasing the reaction rate (Metcalf and Eddy, 2003). Importantly, however, this modeling approach assumed that all independent parameters other than influent concentration were held constant which was a major limitation of this approach. Under field conditions, reaction kinetics may be masked or convoluted by a number of environmental factors. Schipper et al. (2010b) noted denitrification systems may functionally use zero order kinetics though first order reactions most closely described a drainage bioreactor in Illinois (Chun et al., 2010) and an enhanced denitrification wetland system in California (Leverenz et al., 2010).

Removal rates generally increased with increasing temperature (Figure 4.7 c) and this influent water temperature was a significant performance factor at the NERF and Greene Co. bioreactors (α = 0.05). In modeling terms, the effect of temperature on a reaction may be express as a Q_{10}, or the factor by which the removal rate increases for every 10°C increase in temperature. Here, the removal rate model estimates for temperature indicated the Q_{10} for these reactors would range from 0.8 to 5.7. This range overlapped past work in this field showing Q_{10} values of approximately 0.8 to 2.4 (Cameron and Schipper, 2010; Warneke et al., 2011), with the higher value here more similar to extrapolations from work by Robertson and Merkley (2009) and Van Driel et al. (2006) which showed Q_{10}s from 2 to 3 with an extrapolation from Robertson et al. (2008) yielding a Q_{10} of 5.0.

Table 4.4 Removal rate regression model parameters for four bioreactors in Iowa

	Model Intercept	Retention Time (h)	Influent Concentration (mg NO_3^--N/L)	Temperature (°C)	Flow Rate (m^3/h)	Age (months)	R^2
Pekin	-6.67	0.03	1.02‡	NA	1.33§	0.07	0.21
NERF	1.35	-0.02	-0.11	0.08‡	0.23†	0.00	0.76
Greene Co.	-14.70†	0.05	1.25†	0.57‡	0.77†	-0.11	0.77
Hamilton Co.	-5.66‡	0.02	0.44†	0.21	1.99†	-0.09	0.97

† indicates significance at α=0.01
‡ indicates significance at α=0.05
§ indicates significance at α=0.10

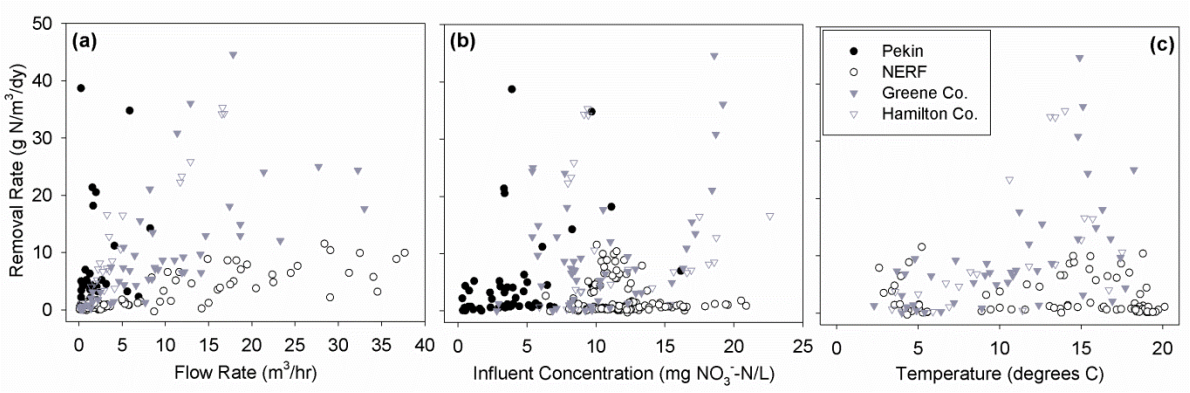

Figure 4.7 Removal rate correlations with flow rate (a), influent concentration (b), and temperature (c) for four bioreactors in Iowa

The use of the "stepwise" selection procedure in the development of the combined data set models allowed introduction of an independent parameter to the model only if it was significant at the $\alpha = 0.05$ level. These regression models for the combined dataset were:

Percent Load Reduction = 17.98 + 0.48 * Retention + 2.20 * Temperature + 3.39* L:W - 75.77 * Cross Section

Removal Rate = $1.96^{\P}$ + 0.54 * Influent Concentration + 0.34 * Temperature + 0.44 * Flow Rate - 0.30 * Age - 8.44 * Cross Section

With ¶ indicating this intercept was not significant at the 0.05 level, though it was nevertheless included in the model as the linear regression intercept. Results of these regression models shown versus measured performance are in Figure 4.8 a and b with R^2 values of 0.59 and 0.63 for the percent load reduction and the removal rate model, respectively. Not surprisingly based on the individual bioreactor models, temperature was significant in both combined dataset models. The significance of the L:W and Cross-section parameters in the percent load reduction model indicated there was a significant difference between the four sites, a difference which could also be seen by comparing the annual bioreactor reductions in Table 4.2. The poorer model performance at higher measured removal rates (Figure 4.8b) may have been an artifact of measured removal rate calculation as these high rates push the limits of other removal rates reported in literature (Schipper et al., 2010b).

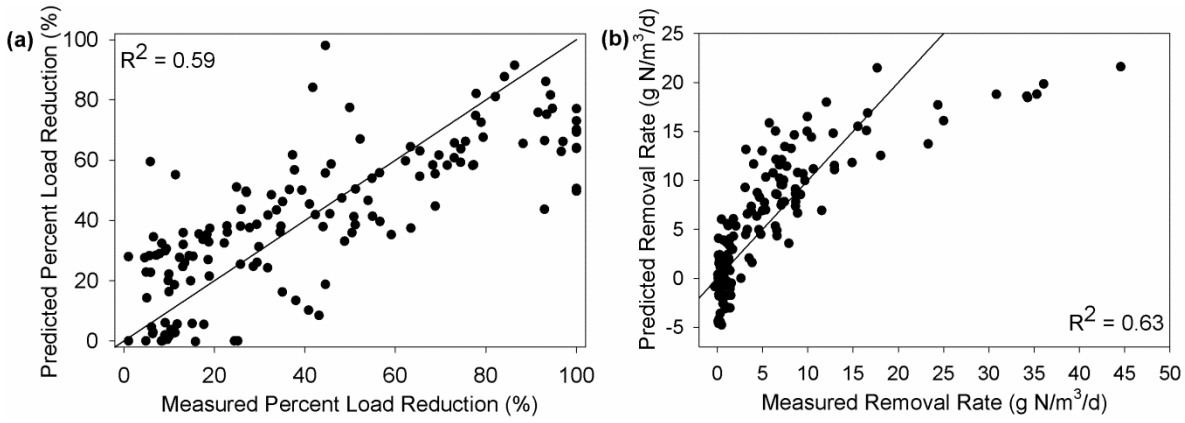

Figure 4.8 Modeled versus measured performance for percent load reduction (a) and N removal rate (b) shown with a 1:1 line

Cost

The total installation cost for six bioreactors in Iowa ranged from $4,390 to $11,820 and from $194.72/ha to $585.64/ha (Table 4.5). Treated areas ranged from 12 ha to 60 ha with the most expensive site on an area basis (i.e. Iowa 4) having the smallest drainage treatment area. The most expensive installation component for these bioreactors was either the contractor labor costs or the woodchips and transport depending upon the site. Ample local availability of woodchips can help minimize transport cost. Contracting fees from those who charge by the hour may eventually be reduced as increased experience with these systems may result in decreased installation time. Moreover, the cost of control structure manufacturing may decrease if there is a higher demand for these structures.

Table 4.5 Installation costs for six denitrification bioreactors treating agricultural drainage in Iowa

	Structure	Contractor	Woodchips	Supplies	Total	$ total/ha drained
Greene Co.	$2,750.00	$5,250.00	$1,245.00	$500.00	$9,745.00	$512.35
Hamilton Co.	$1,640.00	NA†	$2,400.00	$350.00	$4,390.00	$216.96
Iowa 1	$1,970.00	$1,800.00	$3,350.00	$560.00	$7,680.00	$316.30
Iowa 2	$1,270.00	$1,890.00	$3,000.00	$780.00	$6,940.00	$428.73
Iowa 3	$1,640.00	$5,030.00	$4,650.00	$500.00	$11,820.00	$194.72
Iowa 4	$1,480.00	$2,710.00	$2,520.00	$400.00	$7,110.00	$585.64

†contractor time donated

Using the average total influent load from the NERF, Greene Co. and Hamilton Co. sites (28 kg N/ha) combined with the drainage treatment area and the annual percent total load reduction for each reactor allowed an estimation of cost efficiency. This simple cost evaluation which assumed the annual percent load removals from the Greene Co. and Hamilton Co. bioreactors were maintained for fifteen years, resulted in cost efficiencies of $3.67/kg N to $4.72/kg N and $0.91/kg N to $1.06/kg N,

respectively. This range of values overlapped denitrification bed cost efficiencies developed by Schipper et al. (2010b) of $2.39/kg N to $15.17/kg N. The values calculated here were also similar to cost efficiencies of other agricultural drainage water quality practices with reports of approximately $2/kg N to $4/kg N for wetlands and controlled drainage (Baker, 2009; Hyberg, 2007; Iovanna et al., 2008). Like these other practices, cost sharing for bioreactor installations in Iowa is available through the Environmental Quality Incentive Program with a one-time payment of $3,999.50 (Iowa NRCS, 2010); this cost-share represents 34% to 91% of the total installation cost of the bioreactors in Table 4.5.

Conclusions

The objectives of this work were to analyze NO_3^- removal performance from four existing denitrification bioreactors in Iowa and to use statistical modeling to identify the environmental and design factors most affecting their performance. This work enhances understanding of denitrification bioreactors treating agricultural drainage by providing the first comprehensive performance evaluation of several such bioreactors in the US Midwest. Averaged over all years, the four bioreactor sites had NO_3^--N concentration, bioreactor load and total load reductions of 43%, 45% and 32%, respectively. Statistical modeling of these systems indicated that water temperature and influent NO_3^- concentration were the most important factors for percent bioreactor load reduction and removal rate, respectively. Retention time was also a significant factor in percent load reduction for three of the four bioreactors indicating that increased retention and warmer temperatures improves bioreactor NO_3^- removal performance. The simple economic assessment showed that at $0.91/kg N to $4.72/kg N, bioreactors had cost efficiencies comparable to other water quality technologies. More studies of field-scale performance from denitrification bioreactors designed with various methods and in various parts of the Midwest are needed to further improve understanding of the potential for these systems to positively impact water quality.

Acknowledgements

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CHAPTER 5 FINANCIAL COMPARISON OF SEVEN NITRATE REDUCTION STRATEGIES FOR MIDWESTERN AGRICULTURAL DRAINAGE

A paper submitted to the *Journal of Environmental Quality*

Laura E. Christianson, John Tyndall, and Matthew Helmers

Abstract

Much work has been invested in the development of practices and technologies that reduce nitrate losses from agricultural drainage in the US Midwest. While each individual practice can be valuable, the effectiveness will be site specific and the acceptability of each approach will differ between producers. To enhance decision making in terms of water quality practices, this work created cost effectiveness parameters for seven nitrate management strategies (controlled drainage, wetlands, denitrification bioreactors, nitrogen management rate and timing, cover crops, and crop rotation). First, for each practice, available published cost information was used to develop a farm-level financial model that assessed establishment and maintenance costs as well as examined financial effects of potential yield impacts. Next, these financial models, which were presented in terms of total present values, were transformed into equal annualized costs (EACs). Finally, each practice's EACs were combined with literature review of N reduction (% N load reduction) which allowed comparison of these seven practices in terms of cost effectiveness (dollars per kg N removed). Nitrogen timing modification followed by nitrogen application rate reduction were the most cost effective while the in-field vegetative practices of cover crop and crop rotation were the least cost effective. At less than $3.50 per kg N removed, controlled drainage, wetlands, and bioreactors were fairly comparable with each other. While no individual technology or management approach will be capable of addressing drainage water quality concerns in entirety, this analysis provides measures of cost effectiveness across these seven strategies that allows direct comparison.

Introduction

In the Midwestern "Corn Belt" region, artificial subsurface drainage is a major modification of the natural hydrologic and nitrogen (N) cycles. These drainage systems have allowed for increased productivity over the past century (Dinnes et al., 2002) but NO_3^- losses in drainage have caused significant multi-scale environmental concerns (USGS, 2000). There is no doubt that these Midwestern NO_3^- loadings have become a national water quality issue (Delgado and Follett, 2011).

Much work has been done developing and advancing practices to reduce NO_3^- losses in subsurface agricultural drainage. Dinnes et al. (2002) provide a comprehensive review of NO_3^- reducing technologies for the Midwest including in-field "preventative" nitrogen strategies (e.g. nitrogen management, cover crops, diversified rotations) and "remedial" strategies for nitrogen removal from drainage (e.g. controlled drainage, bioreactors, wetlands). While each strategy and individual practice can be valuable, the NO_3^- removal effectiveness will be site specific and the acceptability of each individual approach will differ between producers. Nevertheless, no individual technology or management approach will be capable of addressing drainage water quality concerns in entirety (Dinnes et al., 2002; Lemke and McKenna, 2008);as such a suite of approaches used across these landscapes will be required (Christianson and Tyndall, 2011).

On an individual basis, farmer adoption of environmental management practices designed to mitigate or prevent issues such as NO_3^- losses are motivationally different from production innovations largely because short-term economic advantages of adopting a mitigation technology are rare (Battel and Krueger, 2005; Gillespie et al., 2007). Farm level action involving use of technology is in large part influenced by owner and operator beliefs and attitudes (i.e., regarding environmental and financial risk) in combination with personal environmental goals and knowledge about technology (McCown, 2005). These, in turn, are shaped by external factors such as cost, overall complexity and effectiveness of available technology, and available technical/ financial support (Prokopy et al., 2008; Lemke et al., 2010). As such, crop producers require comprehensive information about water quality technologies with regard to the context for use, operational parameters, performance efficacy, and the full range of financial parameters (e.g. upfront and long term costs).. Of particular and universal concern for farmers is the financial feasibility of a particular technology in the context of their production system as well as comparative advantage across options. Moreover, comprehensive financial information is needed to calibrate agricultural conservation cost-share programming and better guide these types of federal and states services at the county level.

To enhance land-use decision making, this work investigated the financial parameters of seven NO_3^- management strategies; three were remedial N strategies: controlled drainage, wetlands, denitrification bioreactors and four were preventative N strategies: nitrogen management (i.e., rate and timing), cover crops, and crop rotation. The major objectives of this analysis were two-fold. First, for each water quality practice, available published cost information was used to develop a farm-level financial model that assessed establishment and maintenance costs as well as examined financial effects of potential yield impacts and regionally available governmental subsidies. Secondly, the financial models were combined with a literature review of N reduction effectiveness

which allowed comparison of these seven practices in terms of cost effectiveness (dollars per kg N removed). Ultimately, cost comparison analysis of this type is challenged by 1) limited availability of published cost information, 2) variable methodology in published financial assessments, 3) limited methodological transparency in published cost assessments, 4) variable discount rates, 5) inconsistent analysis horizons due to variable life spans or management horizons, and 6) many costs are often site specific and therefore can exhibit significant ranges. Our analysis is an attempt to normalize to the degree possible these issues and to develop measures of cost effectiveness across these seven N management strategies that are directly comparable.

Review of Seven N Water Quality Strategies

Controlled drainage (also known as water table management) is a strategy that addresses agricultural NO_3^- loadings through the use of a series of structures installed in drainage pipes or in drainage ditches which allow control of the water table depth (Gilliam et al., 1979; Cooke et al., 2011). Using the structures, the water table is maintained closer to the surface at non-critical intervals of the growing season when a high water table does not impede in-field operations and the water table is lowered during other intervals (Dinnes, 2004; Frankenberger et al., 2006). Done correctly, this practice can be used to optimize agronomic and/or environmental objectives by providing adequate water to the root zone when it is needed while also reducing the amount of drainage water, and thus NO_3^-, leaving the field (Cooke et al., 2011). A major limitation of controlled drainage is that it becomes cost prohibitive on slopes greater than 0.5% to 1% (Dinnes et al., 2002; Frankenberger et al., 2006).

Denitrification bioreactors are a novel option being investigated in the Midwest as an end-of-pipe, remedial technology for NO_3^- in agricultural drainage. Also known as woodchip bioreactors, denitrifying bioreactors, and biofilters, these systems use control structures to regulate drainage water flowing through an excavation (typically > 30 m long, >1 m wide) filled with a carbon source (Cooke et al., 2001; Christianson et al., 2009). The provision of a carbon source (i.e. wood) and the maintenance of anaerobic conditions in the bioreactor allows for an enhanced environment for denitrifying bacteria, which transform NO_3^- in the drainage to N gas (i.e. enhanced denitrification). These systems have been tested for treating drainage from "field sized" areas of approximately 20 ha and usually require very little to no land out of production (Christianson et al., 2009; ISA, 2011).

Constructed wetlands are a long-term NO_3^- reduction strategy intended for watershed-scale treatment (Kovacic et al., 2000; IDALS, 2009). The main NO_3^- removal mechanisms in wetlands include denitrification, plant uptake, and soil nutrient storage (Dinnes, 2004; Crumpton et al., 2008).

The overall process dynamics and N removal effectiveness of a wetland are a complex function of influent nutrients, landscape position, wetland hydraulics/retention time, temperature, carbon for denitrification, and vegetation type (Dinnes, 2004; Crumpton et al., 2008). A key consideration for N removal in wetlands is the treatment area ratio with increased removal at increased wetland: watershed area ratios (Kovacic et al., 2000; Baker and Crumpton, 2002; Crumpton et al., 2006; Crumpton et al., 2008). The Iowa Conservation Reserve Enhancement Program (CREP) specifies a wetland size of 0.5% to 2% of the treatment area (not including associated wetland buffer) (Iovanna et al., 2008; IDALS, 2009).

Nitrogen fertilizer management is one of the farm operator-controlled factors to reduce N losses in agricultural drainage (Randall and Goss, 2001; Randall and Mulla, 2001; Dinnes et al., 2002; Randall and Sawyer, 2008). With nearly one hundred percent of American corn acres fertilized at average rates of 135-168 kg N ha^{-1} (USDA ERS, 2011), N fertilizer inputs into conventional row cropped systems are substantial. Field studies show correlation between N application rate, N in drainage and yield (Baker, 2001; Jaynes et al., 2001; Randall and Mulla, 2001) indicating that reduced rates could provide incremental water quality improvement and provide economic benefits particularly in cases where over application is occurring (Sawyer and Randall, 2008). Water quality benefits of modified application rates will be a function of the original and the modified rate (Sawyer and Randall, 2008; Helmers and Baker, 2010) and can be described with:

$$N\ Concentration\ in\ Drainage = 5.72 + 1.33e^{(0.0104 \times N\ Rate)} \text{ (Corn/Soybean)} \quad \text{Equation 5.1}$$

where N concentration is in mg N L^{-1} and rate is applied fertilizer N (kg ha^{-1}) (Lawlor et al., 2008).

In addition to application rate modification, N application timing is another management decision to be considered for water quality improvement. In the Midwest, spring N application more closely synchronizes the application with plant uptake (Cassman et al., 2002; Dinnes, 2004), an outcome that is preferable from both water quality and agronomic perspectives (Randall and Sawyer, 2008). Nevertheless, fall N applications are a way to manage risk associated with uncertain spring weather and various spring-time field activities (USDA ERS, 1997; Stewart et al., 2009). Fall applications tend to correspond with higher N leaching losses compared to spring applications especially when combined with fall over-application (another common risk management tendency) (USDA ERS, 1997; Mitsch et al., 1999; Gentry et al., 2000; Cassman et al., 2002; Dinnes et al., 2002; Dinnes, 2004). The very important positive agronomic effect of a spring N application (i.e. increased yield) has been noted by many authors (Rejesus and Hornbaker, 1999; Randall and Mulla, 2001;

Randall et al., 2003a; Randall and Vetsch, 2005a; Randall, 2008). Regardless of potential water quality and yield benefits, it is likely fall fertilization will continue to some extent throughout this region (Lemke et al., 2010).

The use of a winter cover crop is another in-field, preventative management technique to reduce NO_3^- in agricultural drainage. The main mechanism for this reduction is plant uptake from the soil profile (Kaspar et al., 2007; Kaspar et al., 2008). Benefits of cover crops extend beyond NO_3^- water quality concerns to include erosion control, pest control and enhancement of soil productivity, among others (Dinnes, 2004; Kaspar and Singer, 2011). Common crops used as winter cover include rye, oat, winter wheat, barley, triticale, annual ryegrass, brassica (e.g. radish and mustard) and winter-hardy legumes (e.g. alfalfa, hairy vetch, clovers) which can also provide nitrogen fixation (Kaspar et al., 2008). Despite the potential benefits of cover crops, overall usage has been limited; between 2001 and 2005, a relatively small percentage (11%) of Midwestern farmers had tried cover crops (Singer et al., 2007). The main limitation for cover crops is that they need to grow well under non-ideal conditions like cool temperatures and shortening day lengths (Dinnes et al., 2002; Kaspar et al., 2008). An additional challenge for winter covers crops is that they usually must be killed before planting the main crop and can cause a corn yield reduction (corn following a rye cover, notably) (Kaspar et al., 2007; Kaspar et al., 2008).

Finally, a cropping rotation including perennials allows water quality benefits via N and water uptake during times annual crops may not support these processes (Mitsch et al., 1999; Huggins et al., 2001; Dinnes et al., 2002). The use of legumes such as alfalfa in a rotation can be beneficial in terms of reduction in fertilizer for the following corn crop and benefits from haying/grazing (Olmstead and Brummer, 2008; Stanger and Lauer, 2008). Additional environmental benefits of more diverse crop rotations are numerous and include improved soil quality, enhanced carbon sequestration, and reduced erosion (Olmstead and Brummer, 2008). The main limitations for this sort of rotation include access to markets, crop storage, and additional machinery required (Dinnes, 2004). Dinnes (2004) seemed to have confidence in this water quality strategy as he reported that diversifying cropping systems in Iowa has the most potential to reduce NO_3^- loadings compared with any other best management practice.

Financial Comparison and Cost-Effectiveness Methods

There is a limited availability of published cost information regarding these nitrate reduction strategies. For what data is available, variable methodology and limited transparency in assessment

makes comparison between published analyses difficult. The timing of costs particularly complicates comparisons of water quality practices; for example, controlled drainage, bioreactors and wetlands all have large initial capital outlays and intermittent management costs while nitrogen management, cover crops and crop rotations largely involve variable annual costs. This analysis has carefully constructed cost assessments for all seven practices with itemized cost parameters and unit cost data for each strategy collected from various secondary sources (e.g., published literature, published custom rate surveys, and when necessary personal communication with knowledgeable individuals). All practices were then compared using standard discounted cash flow procedures. Total present value costs (TPVCs) were assessed with the general functional cost model (Equation 5.2).

$$TPVC_{Practice} = C_{est,Practice}\ in\ year\ 1\ + C_{main}\ occuring\ over\ n\ years \qquad \text{Equation 5.2}$$

Where $TPVC_{Practice}$ is the total present value of the cost of a practice, $C_{est,Practice}$ is the full establishment cost, and C_{main} involves all annual and/or periodic maintenance costs of the practice applicable for and discounted over n years. The specific variations of this general model for each individual technology are presented in Appendix 1.

To develop a range of costs for each practice, minimum values and maximum values for each individual cost category were summed to develop a minimum TPVC and maximum TPVC, respectively. If only a single value (mean) was available for a cost, this value was used in both the minimum and maximum TPVC calculation for that practice. Following Burdick et al. (1982) and Tyndall (2009) these compiled TPVCs for each practice (\$ ha treated^{-1}) were, in turn, converted to an Equal Annual Cost (EAC) basis for the purpose of assessing cost effectiveness. Conversion to EACs was done using a Capital Recovery Factor (CRF):

$$EAC = TPVC \times CRF \qquad \text{Equation 5.3}$$

Where TPVC was the total present value of the cost of the practice and the CRF was calculated using:

$$CRF = \frac{i(1+i)^n}{(1+i)^n - 1} \qquad \text{Equation 5.4}$$

where *i* is the annual real discount rate and *n* is the number of years in the evaluation (i.e. planning horizon). The EAC is particularly useful when comparing investments of different projects that vary

in life span, financial terms and maintenance costs. By annualizing all costs, the EAC also allows landowners to examine costs more easily across a long term planning horizon (Kay et al., 2011).

In this synthesis the analysis was carried out over a 50 year planning horizon at a 4% real discount rate. A 4% discount rate represents the average real interest rate on Iowa farmland loans during 2008-2010 and was very similar to the 2011 rate for federal water projects (4.125%) (USDA NRCS, 2011). We combined calculated EACs with published measures of NO_3^- removal efficacy (% load reduction) to develop an efficiency parameter of dollars per kg N removed. Finally, because cost-share has been shown to be an important incentive for operators to make environmental mitigation decisions (e.g. Tyndall, 2009), the impact of existing government cost-share and incentive programming was assessed.

Results by Practice

Controlled Drainage

The major cost of controlled drainage is the capital expense of the structures and their installation. Because of this expense, land slope limitations are an important factor as more structures are needed at steeper sites. Another important consideration is the cost difference between implementing controlled drainage in existing versus newly designed drainage systems (Cooke et al., 2011).

For this evaluation of controlled drainage, the costs to retrofit an existing drainage system as well as the cost to implement a new system designed for controlled drainage were considered. To reflect the marginal cost of water quality improvement and not just the cost of new drainage systems, contractor tiling and materials expenses for new systems were not included. Full cost components are described in Table 5.1. Regarding more long term costs, the cost of maintenance included landowner time to manipulate the control structures which vary based on the number of structures, distance between them, and management intensity a landowner chooses. Additionally, for the planning horizon of this analysis, the structures would need to be replaced in year 40 (involving additional structure costs and contractor fees) with the stop logs/gates replaced every eight years. Based on a range of published component costs, the TPVC of controlled drainage ranged from \$202.69 ha^{-1} to \$844.52 ha^{-1} (Table 5.1). This range of TPVCs spanned both existing and new drainage systems.

Table 5.1 Itemized costs and total present value costs for controlled drainage at real discount rate of 4 % and analysis horizon of 50 years

	Item	Cost Timing (yr)	Min ($ ha^{-1})	Mean ($ ha^{-1})	Max ($ ha^{-1})	Notes and assumptions	Reference
Upfront costs	Structure cost	1	$61.78		$247.11	NEW: 1 structure/8 ha at $500-$2000/ea.	Frankenberger et al., 2006
		1	$123.55		$494.21	EXISTING: 1 structure/4 ha at $500-$2000/ea.	
	Transport structures	---				Assumed included above	
	Design cost	1		$80.63		For new drainage systems but also included as design cost of existing	Jaynes, 2011
	Contractor fees	1	$4.32	$9.47	$15.44	Structure installation: Back hoeing at $35.00/h, $76.65/h, $125.00/h for eight h to treat 65 ha	ISU Extension, 2010;
	Total cost of establishment		**$146.73**		**$343.18**	**NEW (TPVC)**	
			$208.51		**$590.29**	**EXISTING (TPVC)**	
Maintenance costs	Time to raise/lower	1-n	$0.99		$4.94	Four h x two to four times a year; labor at $8-$20/h, 65 h treatment area	ISU Extension, 2010
	Total cost of establishment and maintenance		**$167.96**		**$696.45**	**TPVC**	
	Control structure replacement	40	$13.77		$106.16	Single sum TPVC at 40 years: structures, contractor	
	Stop log/gate replacement	8, 16, 24, …	$20.95		$41.91	Summation of single sum TPV every 8 years for 5 gates per structure at original cost of $14.17 to $15.32 /ea. for 15 cm structures, 1 structure per 4 (Existing) or 8.1 (New) ha	Agri Drain Corp, 2011
	Total cost of establishment, maintenance, and replacement		**$202.69**		**$844.52**	**TPVC**	

Bioreactors

As with controlled drainage, bioreactor establishment costs included design, contractor and structure fees. However, unlike controlled drainage, bioreactor treatment area differed from the surface area of the technology. Here, the $ ha^{-1} values refer to the treatment area, not the bioreactor surface footprint. On an itemized basis, a maximum value for engineering fees of $40 hr^{-1} for 16 h of work was assumed, though if the bioreactor was designed by a technical service provider (e.g. NRCS), these fees may not apply. Bioreactors are typically less than 0.5% of the drainage treatment area so this area ratio was used for the seeding and mowing costs. Bioreactor full cost components are described in Table 5.2.

Farmer time for adjusting the control structures would be minimal compared to the controlled drainage practice due to fewer structures here. In addition to annual maintenance, the bioreactor material and, eventually, control structures need to be replaced within this 50 year horizon. It was

assumed that the chips would be replaced in year 20 and year 40 (involving costs associated with new woodchips, seeding and contractor fees), while the structures would be replaced in year 40. Similar to controlled drainage, the stop logs/gates would need to be replaced every eight years.

The TPVCs of bioreactors over 50 years ranged from \$354.01 ha^{-1} to \$732.25 ha^{-1} (Table 5.2). These cost estimates were within range of five bioreactor installations in Iowa with total costs of \$4,400 to \$11,800 to treat drainage from 12 ha to over 40 ha (Sutphin and Kult, 2010; ISA, 2011).

Table 5.2 Itemized costs and TPVCs for a denitrification bioreactor at real discount rate of 4 % and analysis horizon of 50 years

	Item	Cost Timing (yr)	Min ($ ha^{-1})	Mean ($ ha^{-1})	Max ($ ha^{-1})	Notes and assumptions	Reference
Upfront Costs	Both control structures	1	\$49.42		\$197.68	Two control structures at \$500 to \$2000 ea.; 20.2 ha treatment area	Agri Drain Corp, 2011
	Structure transport	---				Assumed included above	Assumption
	Woodchip cost	1		\$116.14		Two semi loads at \$975 chips + \$200 transport ea.; 20.2 ha treatment area	ISA, 2011
	Woodchip transport to farm	---				Included above	
	Design cost	1	\$0.00		\$31.63	Assumed: \$40/h for 2 d of work or NRCS service provider; 20.2 ha treatment	Assumption
	Contractor fees	1	\$27.68	\$60.61	\$98.84	Back hoeing at \$35.00/h, \$76.65/h, \$125.00/h for 16 h to treat 20.2 ha	ISU Extension, 2010; Assumptions
	Seeding bioreactor surface	1	\$0.05	\$0.11	\$0.15	Seeding grass, broadcast with tractor; for 20.2 ha treatment and 0.10 ha bioreactor at \$9.88, \$22.61, and \$29.65 per ha	ISU Extension, 2010
	Seed cost	1		\$1.11		Seed costs from dealer: \$222.27/ha (\$89.95/ac) for CRP Mix (CP23) Diversified mix; bioreactor surface 0.005 of treatment area	Prairie Land Management: http://www.habitatnow.com/store/shop/shop.php?pn_selected_category=37
	Misc. materials	1		\$8.80		6" tile \$890 per 305 m(1000 ft); Assume 61 m needed for control structure connections for 20.20 ha treatment area	Agri Drain Corp, 2011
	Total cost of establishment		**\$203.19**		**\$454.35**	**TPVC**	
Maintenance Costs	Time to raise/lower	1-n	\$1.19		\$2.97	Three h/yr with farm labor wages at \$8-\$20/h, 20.20 ha treatment area	Assumption; ISU Extension, 2010
	Mowing/maintenance	1-n	\$0.12		\$0.62	Spot mowing bioreactor at \$24.71 to \$123.55/ha for 20.2 ha treatment	ISU Extension, 2009
	Total cost of establishment and maintenance		**\$231.33**		**\$531.32**	**TPVC**	
	Replacement yr 20	20	\$65.66		\$98.18	Single sum TPVC at 20 years: woodchips, contractor, seeding	
	Replacement yr 40	40	\$40.26		\$85.98	Single sum TPVC at 40 years: woodchips, contractor, structures, seeding	
	Gate replacement	8, 16, 24,…		\$16.76		Summation of single sum TPV every 8 years for 5 gates per structure (\$14.17 to \$15.32 /ea. for 15 cm structure) 2 structures per 20.2 ha	Agri Drain Corp, 2011
	Total cost of establishment, maintenance, and replacement		**\$354.01**		**\$732.25**	**TPVC**	

Wetlands

Wetlands were unique to this analysis in that their capital expense can be very high but they are capable of treating drainage from far larger areas than the other strategies considered here. Design and construction are important components of wetland establishment but the largest single expense is the land acquisition cost. Longer-term economic considerations sometimes include the opportunity cost of lost crop income (e.g. Prato et al., 1995; Crumpton et al., 2008) as well as maintenance and mowing expense and potential income streams.

For the purposes of this comparison, a 405 ha treatment area was assumed with a wetland of 1% of this area (4 ha) consistent with the Conservation Reserve Enhancement Program (CREP) guidelines for Iowa (IDALS, 2009). Accordingly, in addition to the wetland basin, a grass buffer is required. The wetland buffer was assumed to have a 3.5:1 area ratio with the wetland (i.e. 3.5% of the treatment area in buffer, 14 ha) (IDALS, 2011). Because land acquisition costs are the largest portion of CREP wetland expense, this was included here though land for the other practices was assumed to be owned (forgone annual land rent would be another way to account for land costs). The cost per area for this practice reflected the area treated, not the area of the wetland and associated buffer. Wetland cost components are shown in Table 5.3.

Structural components include a water control structure and a weir plate which are used to control wetland flow. The annual maintenance cost included mowing 10% of the buffer area. Replacement costs of the control structure and sheet pile weir in year 40 were included. Also, over the life of a wetland, sediment removal and earthwork maintenance would be required, though those costs were not incorporated here as their timing would be difficult to estimate and may occur at greater than the 50 year planning horizon.

The overall TPVC estimated for constructed wetlands ranged from \$660.69 ha^{-1} to \$925.52 ha^{-1} (Table 5.3). This estimate compared well with cost assessments from IDALS CREP wetlands constructed in Iowa (CREP wetlands average approximately \$880 ha^{-1} total; land acquisition (\$513 ha^{-1}), establishment and maintenance costs (\$297 ha^{-1}), and engineering costs (\$69 ha^{-1})). To date, 72 wetlands have been installed under the CREP wetland program in Iowa with an average treatment area of 505 ha (1,250 ac) (IDALS, 2011).

Table 5.3 Itemized costs and TPVCs for a wetland at real discount rate of 4 % and analysis horizon of 50 years

	Item	Cost Timing (yr)	Min ($ ha^{-1})	Mean ($ ha^{-1})	Max ($ ha^{-1})	Notes and assumptions	Reference
Upfront costs	Design cost	1		$71.17		Assumed: $40/h for 90 d of work (8h /d) for 405 ha site	Assumption
	Contractor fees	1	$28.17	$34.43	$41.51	Building ponds at 8h /d for 15 d with Custom Rate Survey $/h for 405 ha wetland , not including seeding time	ISU Extension, 2010
	Seeding buffer	1	$0.35	$0.79	$1.04	Tractor broadcasting at $9.88, $22.61, or $29.65/ ha for 14 ha wetland buffer for 405 ha treatment	ISU Extension, 2010
	Seed cost	1	$7.43		$95.38	Seed costs from dealer: $212.39/ha for CRP wetland Program Mix to $162.09/kg for "wetland seed mix" at needed 16.8kg/ha	Prairie Land Management: http://www.habitatnow.com/store/shop/shop.php?pn_selected_category=37; Ernst Conservation Seeds: http://www.ernstseed.com/seed-mixes/
	Weir Plate	1		$14.83		$30 per sq. ft for 40 ft width x 5 ft sheet pile plate, for 405 ha site	http://www.eng-tips.com/viewthread.cfm?qid=161307&page=1
	Control Structure	1	$3.26		$7.25	One large control structure ($1320 to $2935 pe ea.), for 405 ha site	Agri Drain Corp, 2011
	Land acquisition	1	$529.08		$679.31	$11,757 to $15,095/ha for 4 ha wetland plus 14 ha buffer treating 405 ha; 2010 state-wide Iowa average for high and medium grade lands	ISU Extension, 2011b
	Total cost of establishment		**$654.28**		**$910.48**	**TPVC**	
Maintenance costs	Time to manage	1-n	$0.09		$0.43	Spot mowing 10% of buffer area at $24.71 to $123.55/ha	ISU Extension, 2009
	Total cost of establishment and maintenance		**$656.14**		**$919.77**	**TPVC**	
	Control structure and weir replacement	40	$4.55		$5.75	Single sum TPVC at yr 40 includes costs of a new structure and weir and 16 h of earth work	
	Total cost of establishment, maintenance, and replacement		**$660.69**		**$925.52**	**TPVC**	

Nitrogen Management

The establishment costs for both rate and timing nitrogen management practices were similar and relatively simple; both included the application machinery and fertilizer costs as described in Table 5.4. Because an N management practice is an annual occurrence, there are no long-term maintenance costs but, rather, establishment cost and revenue impacts occur every year.

For these nitrogen management strategies, a baseline scenario of fall applied 168 kg N ha^{-1} was developed for comparison. The marginal difference in TPVC between the baseline and the rate/timing alternative was used in the analysis rather than the absolute value of the rate/timing TPVC

themselves. Using these marginal costs of the lower rate practice and of the spring timing practice allowed evaluation of their cost solely due to water quality improvement.

Table 5.4 Itemized costs and TPVCs for nitrogen management for corn at real discount rate of 4 % and analysis horizon of 50 years

	Item	Cost Timing (yr)	Min ($ ha^{-1})	Mean ($ ha^{-1})	Max ($ ha^{-1})	Notes and assumptions	Reference
Upfront costs and yield impacts	Fertilizer application	1-n	$14.83	$24.09	$42.01	Anhydrous-injecting, w/tool bar	ISU Extension, 2010
	Diesel for equipment	---				Included above	
	Fertilizer cost	1-n		$156.40		North Central US mean 2008-2010 anhydrous ammonia price paid: $762.80/metric ton ($692/ton); 168 kg N/ha; Anhydrous ammonia:82-0-0 (82%)	USDA NASS, 2011
	Total cost of establishment for baseline application		$171.23		$198.41	Using Fertilizer cost: $156.40/ha considering application of 168 kg N/ha in Fall	USDA NASS, 2011
	Total cost of establishment at a lower rate (from 168 kg/ha to 140 kg/ha)		**$145.16**		**$172.34**	Using Fertilizer cost: $130.33/ha for application of 140 kg N/ha rather than $156.40 for 168 kg N/ha	USDA NASS, 2011
	Total cost of establishment of Spring application		**$178.42**		**$205.60**	Spring price of $798/metric ton at 168 kg N/ha application rate ($163.59/ha)	USDA ERS, 2011; USDA NASS, 2011
	Annual baseline revenue	1-n		$1,850.12		Iowa mean 2008-2010 yield of 10.84 metric ton/ha (173 bu/ac) and 2008-2010 mean corn price received of $0.17/kg ($4.38/bu); at 99% yield for 168 kg N/ha	ISU Agronomy Extension., 2011; USDA NASS, 2011
	Annual revenue from changed yields due to nitrogen management (Lower rate)	1-n		$1,831.44		Iowa mean 2008-2010 yield of 10.84 metric ton/ha and 2008-2010 mean corn price received of $0.17/kg; at 98% yield for 140 kg N/ha	ISU Agronomy Extension., 2011; USDA NASS, 2011
	Annual revenue from changed yields due to nitrogen management (Spring application)	1-n		$1,947.30		Iowa mean 2008-2010 yield of 10.84 metric ton/ha and 2008-2010 mean corn price received of $0.17/kg; with 4.2% yield boost for spring application	USDA NASS, 2011
Long term costs	Total cost of establishment and revenue impacts for baseline application		-$36,066.38		-$35,482.46	TPVC (negative represents a revenue)	
	Total cost of establishment and revenue impacts at a lower application rate		**-$36,224.89**		**-$35,640.97**	**TPVC (negative represents a revenue)**	
	Total cost of establishment and revenue impacts for Spring application		**-$37,999.43**		**-$37,415.51**	**TPVC (negative represents a revenue)**	
	Rate Marginal Cost		**-$158.51**		**-$158.51**	**Marginal TPVC**	
	Timing Marginal Cost		**-$1,933.05**		**-$1,933.05**	**Marginal TPVC**	

N Application Rate Reduction (168 kg N ha $^{-1}$ to 140 kg N ha^{-1})

Economic analysis of lowering the N application rate consists of less fertilizer expense in addition to the cost of potential yield loss depending upon the initial and final application rates (Sawyer and Randall, 2008). This analysis is greatly complicated by the variability of the impacts of initial and revised fertilizer rates. Challenges to N fertilizer rate reduction include the fact that the

optimum rate is indeterminable at application time (though soil testing can help) and is highly variable year to year. Sawyer and Randall (2008) provide a detailed explanation of these variable negative and positive returns based on initial and final fertilizer rates.

In analyzing the costs of reduced fertilizer rate here, "establishment" cost consisted of less fertilizer purchased (i.e. a cost savings) as well as the effect of potentially reduced yield. The Iowa State University N-Rate Calculator (ISU Agronomy Extension, 2011) was used to determine the yield impact from changing the fertilizer rate. Using a three-yr average (2008-2010) anhydrous ammonia price of \$762.80/metric ton (\$692 ton^{-1}) (USDA NASS, 2011) and a three-yr average (2008-2010) Iowa corn price of \$0.17 kg^{-1} (\$4.38 bu^{-1}) (USDA NASS, 2011), the calculated percent of maximum yield was 99% at an N application rate of 168 kg N ha^{-1} (and was approximately 98% at 140 kg N ha^{-1} (corn following soybean rotation). However, it's worth noting that shifting to this lower rate permanently may not be realistically sustainable over the entire 50 yr planning horizon if soil pools of N become depleted.

N Application Timing Modification

The cost of shifting application from the fall to the spring is affected by differences in both fall/spring fertilizer price and yield. Because current fall vs. spring fertilizer prices are no longer published by USDA statistical services (USDA NASS), the average historical difference in the fall and spring fertilizer prices, on a percentage basis, was used to calculate the average increase in expense for spring anhydrous application. Between 1960 and 1994, the average prices for September/October were \$184.14/metric ton (\$167.05/ton) and for April/May were \$192.62/metric ton (\$174.74/ton) (USDA ERS, 2011), thus an increase of 4.6% over the average 2008-2010 price of \$762.80/ metric ton was used for spring (spring: \$798/metric ton).

Multiple authors report lower NO_3^- loadings in drainage with corresponding higher corn yields for spring versus fall N applications (Rejesus and Hornbaker, 1999; Randall and Mulla, 2001; Randall, 2008). Spring N fertilizer applications may increase yield by 8% to 14% compared to fall applications (Randall and Mulla, 2001; Randall, 2008); however this may not always be the case, for example there was no corn yield difference between fall and spring applications at two different application rates during a study in Iowa (Lawlor et al., 2011 Accepted). Despite this variability, an overall 4.2% corn yield boost was included for the practice of spring application (site year average from Randall et al., 2003a; Vetsch and Randall, 2004; Clover, 2005; Randall and Vetsch, 2005a; Lawlor et al., 2011 Accepted).

The marginal TPVCs of the N management rate and timing practices were -$158.51 ha^{-1} and -$1,933.05 ha^{-1}, respectively (Table 5.4). These negative TPVCs represent cost-savings from these practices. With regard to spring applications, Randall and Sawyer (2008) also noted long-term economic gains of $46.46 ha^{-1} yr^{-1} to $126.02 ha^{-1} yr^{-1} (seven and fifteen year averages). However, it's worth considering that a complete shift to fall fertilization could be expensive for individual producers in terms of both additional infrastructure required for spring applications (storage, equipment, labor, handling, application, etc.) and in the potential loss of yield by a delayed planting date (Otto, 2008). A final note for modification of fertilizer rate is that when lower N rates are applied, the risk of a yield loss is increased compared to higher application rates if it is a year that is more responsive to N inputs. In these years, probability of obtaining a certain yield percentage declines when lower rates are applied; this probabilistic variability was not reflected here. Any such potential increased risk for either of these nitrogen management practices is an important factor in terms of producer decision making.

Cover Crops (Cereal Rye)

For the purposes of this evaluation, cereal rye as a cover crop was studied as this crop has good potential to improve water quality in cool Midwestern climates (Kaspar et al., 2007) and is popular in this region (Singer, 2008). First year costs of a cover crop as shown in Table 5.5, assumed in a no-till system here, included planting and herbicide application as cereal rye overwinters (Kaspar et al., 2008). Because a cover crop here was considered an annual practice, there are no long-term maintenance costs but rather annual establishment costs. A yield reduction for corn following rye was also an important part of the analysis. A 6.2% corn yield reduction was assumed compared to a baseline where no cover crop was used (site year average from Strock et al., 2004; Kaspar et al., 2007; Sawyer et al., 2009; Pederson et al., 2010; Sawyer et al., 2010; PFI, 2011; Qi et al., 2011 in press). This corn revenue reduction was assumed to occur every other year during the planning horizon (i.e. a corn/soybean rotation).

The TPVC of this rye cover crop was $2,649.90 ha^{-1} to $3,865.52 ha^{-1} (Table 5.5). However, several additional comments are important. First, costs to kill the cover are contingent upon producer actions. For example, in a no-till system as assumed here, an early burn-down application of herbicide may be done regardless if a cover crop was present; likewise, in a tilled system, a producer may do a second tillage pass in the spring regardless of a cover crop. Second, rye cover crop implementation costs can be $10 ha^{-1} to $15 ha^{-1} lower if a landowner chooses not to use a custom operator (Saleh et al., 2007). Next, potential negative yield impacts will likely be reduced or minimized through several

years of experience with cover crop management. This increased experience also likely means a more effective cover, though returns to farm management can improve under highly skilled managers regardless of the production practice. Finally, some of the N taken up by a cover crop will be returned to future crops. It is difficult to place an economic value on this, but it is worth noting the multiple benefits to the soil provided by cover crops (Kaspar and Singer, 2011).

Table 5.5 Itemized costs and TPVCs for a cover crop at real discount rate of 4 % and analysis horizon of 50 years

	Item	Cost Timing (yr)	Min ($ ha^{-1})	Mean ($ ha^{-1})	Max ($ ha^{-1})	Notes and assumptions	Reference
Upfront costs and yield impacts	Seed costs	1-n	$14.83		$29.65	Planted at 63 kg/ha (1 bu/ac); cereal rye	Kaspar et al., 2008; Kaspar, 2011
	Planting Drill	1-n	$18.53	$32.12	$49.42	Drilling small grain	ISU Extension, 2010
	Diesel for equipment	---				Included above	
	Spraying	1-n	$11.12	$15.07	$21.99	Ground, broadcast, tractor	ISU Extension, 2010
	Herbicide cost	1-n		$14.09		Herbicides, Glyphosate (Roundup), 480 kg/m^3 (4 lb/gal), Price paid, US Total, 2010: $6023/$m^3$ ($22.8/gal); 0.0023 m^3/ha (1 qt/acre)	Kaspar et al., 2008; USDA NASS, 2011
	Total cost of establishment		**$58.56**		**$115.15**	**TPVC**	
	Annual baseline revenue (no cover crop)	1-n		$1,868.81		Iowa mean 2008-2010 yield of 10.84 metric ton/ha (173 bu/ac) and 2008-2010 mean corn price received of $0.17/kg ($4.38/bu); at 100% yield	USDA NASS, 2011
	Annual revenue from changed yields due to cover crop	1-n		$1,752.95		Iowa mean 2008-2010 yield of 10.84 metric ton/ha (173 bu/ac) and 2008-2010 mean corn price received of $0.17/kg ($4.38/bu); at 6.2% yield reduction for corn following rye	USDA NASS, 2011
	Difference in annual revenue from baseline			$115.87		Considered a cost of cover crop with corn grown in every other year	
	Total cost of establishment and revenue impacts		**$2,649.90**		**$3,865.52**	**TPVC**	

Crop Rotation (Multiple Years of Alfalfa)

The number of possible rotation combinations is quite large; therefore to simplify this work investigated a multi-year incorporation of alfalfa into a corn rotation. The major costs for such a crop rotation are the seed, planting, and harvesting. Only one year of alfalfa in a rotation may not be as beneficial (e.g., fertilizer reduction, alternative product) as several years considering high seed cost and potential low alfalfa yield in the establishment year (Olmstead and Brummer, 2008). This diversified crop rotation consisted of two years of corn (years 1-2) followed by three years of alfalfa (years 3-5). The cost components of this rotation are shown in Table 5.6.

Within the rotation, enterprise budget information published by Iowa State University was used to specifically estimate the costs of corn following soybean (for year 1, 6, 11, etc.) and for corn following corn (in years 2, 7, 12, etc.) (ISU Extension, 2011a). Default ISU Ag Decision Maker (ISU Extension, 2011a) values were used after removing land rent costs (i.e. assumed land owned) and substitution of average Iowa 2010 corn yield from (USDA NASS, 2011).

A multiple year alfalfa rotation provides monetary benefit via reduced fertilizer requirements, reduced tillage and other field trips, and revenue from the alfalfa harvest. Here only direct revenue streams were considered with alfalfa revenue in years 3-5 and corn revenue in years 1-2. The first year of alfalfa was assumed to only have one cutting rather than the three cuttings done in the production years (i.e. establishment years had one third of the yield experienced in production years). Corn following alfalfa may have an increased yield of 19% to 84% compared to corn after corn, according to a review by Olmstead and Brummer (2008), but Liebman et al. (2008) showed more moderate corn yield increases averaging 4.5% which was used here for the first year of corn.

Additionally, the TPVCs for this crop rotation scenario were compared against TPVs for a traditional corn/soybean rotation. Similarly to the nitrogen management practice, this allowed evaluation of the cost of this water quality practice (i.e. marginal cost). The corn/soybean baseline scenario was evaluated using the same 5 year framework as the extended rotation with cost values taken from ISU Ag Decision maker for corn following soybeans and herbicide tolerant soybeans following corn with default values except for removal of land rent costs and use of average yields (2008-2010, USDA NASS data) (Table 5.6) (ISU Extension, 2011a).

The resulting total marginal TPVC of this diversified rotation was \$2,117.22 ha^{-1} to \$3,847.01 ha^{-1} (Table 5.6). One major caveat is that if this rotation were done by a large numbers of producers in a limited area, the alfalfa price could be severely affected. The values developed here were contrary to values from Olmstead and Brummer (2008) who showed a diversified rotation was more profitable than a conventional rotation.

Table 5.6 Itemized costs and TPVCs for a diversified crop rotation at real discount rate of 4 % and analysis horizon of 50 years

	Item	Cost Timing (yr)	Min ($ ha^{-1})	Mean ($ ha^{-1})	Max ($ ha^{-1})	Notes and assumptions	Reference
Upfront cost and [illegible] of	Seed costs	year 3 of every 5	\$101.19		\$140.48	Legume, alfalfa, public and common seed or proprietary seed, price paid, National, 2010: \$273-\$379/cwt; planted 16.8 kg/ha	USDA NASS, 2011
	Planting Drill	year 3 of every 5	\$18.53	\$32.12	\$49.42	Drilling small grain	ISU Extension, 2010
	Diesel for equipment	---				Included above	Assumption
	Soil preparation	year 3 of every 5		\$34.10		Disking, harrow: Default values from ISU Ag Decision Maker	ISU Extension, 2011a (alfalfa)
	Herbicide	year 3 of		\$37.81		Default values from ISU Ag Decision	ISU Extension,

Table 5.6 Itemized costs and TPVCs for a diversified crop rotation at real discount rate of 4 % and analysis horizon of 50 years (continued)

		every 5				Maker (machinery and chemical)	2011a (alfalfa)
	Labor	3-5 of every 5		$81.54		Pre-harvest labor: 7.4 h/ha (3 h/ac) at $11.00/h	ISU Extension, 2011a (alfalfa)
	Fertilizer	3-5 of every 5	$307.15		$481.36	Default values from ISU Ag Decision Maker for establishment year (min) and production year (max); machinery and chemical	ISU Extension, 2011a (alfalfa)
	Harvesting - mowing	3-5 of every 5	$19.77	$30.64	$37.07	Mowing/conditioning	ISU Extension, 2010
	Harvesting - baling	3-5 of every 5	$74.13	$123.55	$172.97	Haying baling - small square: $0.30-$0.70/bale; 12.4 ton/ha (5 ton hay/ac) at 45.4 kg/bale (100 lb/bale)	ISU Extension, 2010; Assumption
	Total cost of alfalfa establishment	year 3 of every 5	**$674.23**		**$860.55**		
	Total cost of alfalfa maintenance	Year 4 and 5	**$656.81**		**$772.95**		
	Corn in year 1	Year 1 of 5		$1,183.64		Cost of corn establishment (corn following soybean to be more accurate for years 6, 11, etc.); land rent removed, 10.84 metric ton/ha (173 bu/ac) yield	ISU Extension, 2011 (corn following soybean)
	Corn in year 2	Year 2 of 5		$1,312.13		Cost of corn establishment (corn following corn); land rent removed, 10.84 metric ton/ha (173 bu/ac)	ISU Extension, 2011 (corn following corn); USDA NASS, 2011
	Total costs for five year diversified rotation		**$4,214.00**		**$4,588.79**	**TPVC: Corn in yrs 1 and 2 with alfalfa establishment in yr 3 and alfalfa maintenance in yrs 4-5**	
	Alfalfa revenue	4-5 of every 5		$1,511.46		Alfalfa average yield 12.4 ton/ha (5 ton hay/ac) (assuming 3 cuttings); Iowa mean 2008-2010 alfalfa hay price received: $134.85/metric ton ($122.33/ton)	USDA NASS, 2011; ISU Extension, 2011
	Corn revenue	1-2 of every 5		$1,868.81		Iowa mean 2008-20109 corn yield: 10.84 metric ton/ha (173 bu/ac) and 2008-2010 mean corn price received of $0.17/kg ($4.38/bu)	USDA NASS, 2011
	Total revenue for five year diversified rotation			**$6,850.51**		**TPV: Corn revenue in yr 1 plus 4.5% yield boost, corn revenue in yr 2, alfalfa revenue divided by 3 (only 1 cutting) in alfalfa establishment year, and alfalfa revenue in yr 4-5**	**Liebman et al., 2008**
	Total costs and revenue for diversified crop rotation for 50 yr horizon		**-$12,168.61**		**-$10,438.81**	**TPVC (negative represents a revenue)**	
	Cost of corn and soybean five yr rotation			$4,469.53		TPVC: Five year cost of corn soybean rotation; starting with corn (ISU Decision Maker, corn following soy, yield 10.8 metric ton/ha); soybean cost: $637.53/ha ($258/ac) (ISU Ag Decision Maker for herb. tolerant soy follow corn, yield 3.33 metric ton/ha (49.5 bu/ac); land rent removed	ISU Extension, 2011a
	Revenue of corn and soybean five yr rotation			$7,564.77		TPV: Five year revenue of corn soybean rotation, starting with corn; corn revenue described above; soybean revenue: Iowa mean 2008-2010 yield of 3.33 metric ton/ha (49.5 bu/ac) and mean price $0.38/kg ($10.47/bu) yields $1,281.05/ha ($518.43/ac)	USDA NASS, 2011
	Total costs and revenue for corn and soybean rotation for 50 yr horizon			-$14,285.83		TPVC (negative represents a revenue)	
	Marginal cost		**$2,117.22**		**$3,847.01**	**Marginal TPVC**	

Comparative Cost Effectiveness of Nitrogen Mitigation

The TPVCs from the seven practices ranged from a cost savings of \$1,933.05 ha^{-1} for spring applied N fertilizer to a cost of \$3,865.52 ha^{-1} for a cover crop (Table 5.7). Overall the highest TPVCs were associated with the two in-field vegetated practices, cover crops and crop rotations, and the lowest costs were associated with the N management strategies. The two field-scale constructed practices, controlled drainage and bioreactors, had very similar TPVC ranges at \$202.69 ha^{-1} to \$844.52 ha^{-1} and \$354.01 ha^{-1} to \$732.25 ha^{-1}, respectively. The minimum and maximum TPVCs for each practice (Table 5.1 through 5.6) were then used to develop a range of Equal Annual Costs (EACs) for the strategies (Table 5.7 at 4% discount rate and 50 yr planning horizon; government subsidy not considered). Similar to the TPVC comparison, N timing had the lowest EAC range (-\$89.98 ha^{-1}) and cover crops the highest (\$179.94 ha^{-1}).

Table 5.7 Summary of TPVCs and EACs for seven drainage water quality practices without government payments (i = 4%, n = 50 yrs)

	TPVC \$/ha		EAC \$/ha	
	Minimum	Maximum	Minimum	Maximum
Controlled Drainage	\$202.69	\$844.52	\$9.44	\$39.31
Bioreactors	\$354.01	\$732.25	\$16.48	\$34.09
Wetland	\$660.69	\$925.52	\$30.76	\$43.08
Nutrient Management - Rate	-\$158.51	-\$158.51	-\$7.38	-\$7.38
Nutrient Management - Time	-\$1,933.05	-\$1,933.05	-\$89.98	-\$89.98
Cover Crop	\$2,649.90	\$3,865.52	\$123.35	\$179.94
Crop Rotation	\$2,117.22	\$3,847.01	\$98.56	\$179.08

Nitrogen reduction values associated with the practices analyzed here were taken from literature (Table 5.8) to develop a range of cost efficiencies for these water quality practices on the basis of kg N removed. Dividing the EAC of each strategy by the amount of NO_3^--N removed is a standard way to present total costs per unit of N removed (Van Note et al., 1975; Burdick et al., 1982). To do so, a Midwestern-representative load of 31.4 kg N ha^{-1} was developed from an average of Jaynes et al. (1999) tile and drain N loads and Lawlor et al. (2011 Accepted) drainage N loads at 168 kg N ha^{-1} application rate. Then, the minimum and maximum EAC for each practice from Table 5.7 were each applied to that practices' range for N load reduction (mean, median, 25^{th}, and 75^{th} percentiles from Table 5.8 and Figure 5.1 which are shown in Table 5.9) (Equation 5.5). This load reduction range was developed using each minimum, mean, or maximum percent N load reduction value from Table 5.8 as a data point in the creation of Figure 5.1 which showed these practices varied widely in terms of N removal effectiveness. For example, modification of fertilizer timing had comparatively low N removal, even ranging into the potential for negative water quality impacts, while the constructed practices tended to have good water quality performance. Bioreactors had the

smallest range between 25th and 75th percentiles with mean and median values above 35% load reduction. The other two constructed practices, controlled drainage and wetlands, had similarly high load reduction potential (means and medians ≥ 40%). Note, because the 25th percentile for application timing was a negative value (-2.5%), indicating a contribution to the N load, the resulting marginal increase to the baseline load was used to calculate the $/kg N for this value.

Table 5.8 N reduction strategies comparison of N load reduction

N Load Reduction (%)	Min	Mean	Max	Notes
Controlled Drainage				
Cooke et al. (2008)	30		40	Overview of N management practices
Frankenberger et al. (2006)	15		75	Controlled drainage factsheet
Lalonde et al. (1996)	48	75	100	Mean load reduction, six months of free drainage vs. controlled; Ontario, Canada
Cooke et al. (2011)		30		Overview of N management practice
Gilliam et al. (1979)	10		20	An original paper on drainage control
Drury et al. (1996)		43		Controlled drainage/sub-irrigation system, Canada
Dinnes (2004)	0		50	N technology comparison
Thorp et al. (2008)	31	44	51	Simulation of Midwestern region with Root Zone Water Quality Model-Decision Support System for Agrotechnology Transfer (RZWQM -DSSAT)
Luo et al. (2010)		26		Mean of DRAINMOD-NII simulated N losses for drain spacing 18 m to 36 m for conventional vs. controlled drainage; Waseca, Minnesota
Bioreactors				
Christianson, Unpublished data	11		13	Bioreactor in Iowa
ISA (2011)	47		57	Bioreactor in Iowa
ISA (2011)	27		33	Bioreactor in Iowa
Jaynes et al. (2008)	40	55	65	Denitrification trenches surrounding tile drain, Iowa
Woli et al. (2010)	23	33	50	Bioreactor in Illinois
Chun et al. (2010)		47		Bioreactor in Illinois, slug of nitrate-nitrogen injected
Ranaivoson et al. (2010)	18		47	Bioreactor in Minnesota
Ranaivoson et al. (2010)	35		36	Bioreactor in Minnesota
Wetland				
Crumpton et al. (2006)	25		78	Review table
Kovacic et al. (2000)	33	40	55	Annual N load reduction for three wetlands, three years of data; Champaign County, Illinois
Miller et al. (2002)		33		Wetland in Illinois
Baker and Crumpton (2002)	9		15	Mean annual N load reduction for two years from wetland with area treatment ratio of 1046:1; Iowa
Baker and Crumpton (2002)	34		44	Mean annual N load reduction for two years from wetland with area treatment ratio of 349:1; Iowa
Baker and Crumpton (2002)	55		74	Mean annual N load reduction for two years from wetland with area treatment ratio of 116:1; Iowa
Dinnes (2004)	20		40	N technology comparison
Iovanna et al. (2008)	40		90	Summary of CREP in Iowa
Nitrogen Management - Fall vs. Spring				
Randall et al. (2003b)	-67	6.4	44	Load difference between fall and spring for all corn yrs; 150 kg N anhydrous ammonia per ha for corn
Randall and Vetsch (2005b)	0	27	41	Load difference between fall and spring for all corn yrs; 135 kg N anhydrous ammonia per ha for corn
Randall and Mulla (2001)	24		30	6-yr period at Waseca, Minnesota
Dinnes (2004)	-10		30	N technology comparison
Rejesus and Hornbaker (1999)	14	35	52	Modeling simulation for central Illinois; Fall vs. spring application at five rates ranging from 112 kg N ha^{-1} to 224 kg N ha^{-1} for Drummer soil
Lawlor et al. (2011 Accepted)	-62	-23	7.4	N load difference between spring and fall applied at 168 and 252 application rates
Nitrogen Management - Rate				
Randall and Mulla (2001)	21		28	6-yr period at Waseca, Minnesota; 134 kg ha^{-1} vs. 202 kg ha^{-1} application
Dinnes (2004)	20		70	N technology comparison
Jaynes et al. (2001)	17		40	Central Iowa; loadings of 48 kg N ha^{-1}, 35 kg N ha^{-1}, and 29 kg N ha^{-1} for high, medium and low N application rates, respectively
Cover Crops				
Strock et al. (2004)		13		Southwestern Minnesota, three year study
Jaynes (2011)		40		Based on review
Qi et al. (2011 in press)	-13.5	-3.3	7.6	Four year loads and mean for corn vs. corn with rye cover; Gilmore City, Iowa
Kaspar et al. (2007)		61		Four year average; Boone County, Iowa
Dinnes (2004)	10		70	N technology comparison
Crop Rotation				
Olmstead and Brummer (2008)	14		77	Review
Kanwar et al. (2005)	11		14	Six year average losses from corn/soybean or soybean/corn vs. rotation with three years alfalfa followed by corn, soybean, oats; Nashua, Iowa
Huggins et al. (2001)	18	48	80	Conversion from alfalfa pasture; three yr study, compared with corn and soybean and continuous corn rotations; Lamberton, Minnesota
Dinnes (2004)	-50		95	N technology comparison

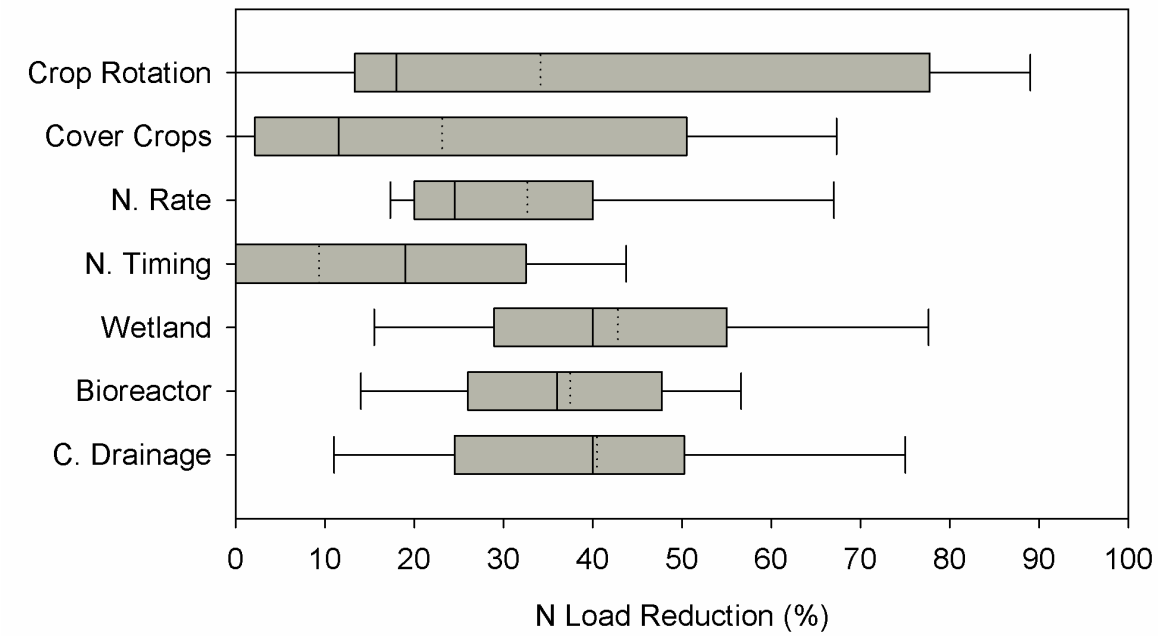

Figure 5.1 Comparison of N load reductions obtained from literature for seven water quality improvement strategies; the box boundaries represent the 25th and 75th percentiles, the solid line represents the median, the dotted line represents the mean, the whiskers show the 10th and 90th percentiles, and the x indicates outliers.

$$\frac{EAC\ \$}{kg\ N} = \frac{minimum\ or\ maximum\ EAC\ \$}{ha} \div \left(\frac{31.4\ kg\ N\ lost\ baseline}{ha} \times Load\ removal\ percentage\ mean, median, 25^{th}, or\ 75^{th}\right)$$

Equation 5.5

In the case of modified N application rate, rather than use load reduction values from literature (i.e. from Table 5.8 and Figure 5.1), a correlation from Lawlor et al. (2008) was used (Equation 5.1). For this practice, literature values proved to be too variable as they were not for the specific rates used in this comparison. After drainage NO_3^--N concentrations were developed via Equation 5.1 for the two application rates, a constant drainage volume was assumed to develop a percent N load reduction. Summary statistics (median and mean) of the resulting \$ kg N^{-1} values were reported for all practices (Table 5.9).

Table 5.9 EACs indicating N removal effectiveness for seven drainage water quality practices without government payments

	EAC $/ha		Load Reduction (%) from Figure 2				EAC $ / kg N removed	
	Min	Max	25th%	75th%	Mean	Median	Mean (StDev)	Median
Controlled Drainage	$9.44	$39.31	26.0	50.0	40.5	40.0	$2.10 ($1.53)	$1.83
Bioreactors	$16.48	$34.09	27.0	47.0	37.5	36.0	$2.27 ($0.99)	$2.13
Wetland	$30.76	$43.08	30.9	55.0	42.8	40.0	$2.91 ($0.83)	$2.83
Nutrient Management - Rate	-$7.38	-$7.38	14.5	14.5	14.5	14.5	-$1.62 ($0.00)	-$1.62
Nutrient Management - Time	-$89.98	-$89.98	-2.5	31.3	9.3	19.0	-$13.76 ($11.92)	-$12.13
Cover Crop	$123.35	$179.94	4.9	45.3	23.1	11.5	$43.18 ($38.19)	$29.50
Crop Rotation	$98.56	$179.08	14.0	77.0	34.1	18.0	$18.72 ($12.57)	$17.08

Modification of nitrogen application timing was the most cost effective option for removing N from drainage (mean \$13.76 kg N^{-1} cost savings or revenue) and cover crop the least (Table 5.9). However, it's important to note that nutrient management practices alone may not be sufficient to meet all nitrogen water quality goals in the Midwestern Region. More constructed practices of controlled drainage, bioreactors and wetlands all had mean costs of less than \$3.00 kg N^{-1}.

To put these cost efficiencies in context of other reported values is difficult in light of the variable methodology and limited transparency of these other assessments, as previously mentioned. Nevertheless several practices were in the range of literature, while others were distinctly different. For example, the cost efficiency of controlled drainage in this analysis was \$2.10 kg N^{-1} (±\$1.53) which was similar to reports which are often in the range of \$2 to \$4 kg N^{-1} (Baker, 2009; Jaynes et al., 2010). Moreover, the cost efficiency of wetlands is often reported at approximately \$3 to \$4 kg N^{-1} (Ribaudo et al., 2001; Hyberg, 2007; Iovanna et al., 2008; Baker, 2009); the value here was \$2.91 kg N^{-1} (±\$1.83). The other practices either did not have many reports of cost efficiencies (e.g. crop rotation) or were significantly different from such values. Only one report was available for bioreactors; in a multi-year cost analysis of a theoretical denitrification system, Schipper et al. (2010) calculated \$2.39 kg N^{-1} to \$15.17 kg N^{-1}. This range was slightly higher than what was estimated here for a bioreactor (\$2.27 kg N^{-1} ± \$0.99). Finally, cover crops have been reported to be less expensive per kg N removed than this analysis' mean value of \$43.18 kg N^{-1} (±\$38.19) with literature ranging from \$1.26 kg N^{-1} to \$11.06 kg N^{-1} (Saleh et al., 2007; Kaspar et al., 2008; Baker, 2009); these previous reports may not have included corn yield impacts.

Government Incentives

In Iowa, Environmental Quality Incentive Program (EQIP) payments were available for each of the practices evaluated here except for modification of fertilizer rate (Iowa NRCS, 2010) (Table 5.10). EQIP cost rates used were standard rate, not the higher rates available for historically underserved groups. Incentives for controlled drainage, bioreactors and wetlands were treated as one time, present value payments (year 1) while the others were terminating annual series occurring in years 1-n with time limits set by the EQIP payment schedules. Though EQIP values were available for wetlands, cost share payments from the CREP were more appropriate because the wetland in this analysis was sized based upon Iowa CREP guidelines. For a CREP 30 year easement agreement, compensation included 15 annual rental payments of 150% the soil rental rate, cost-share for 100% of the wetland installation (90% federal, 10% state), and a one-time incentive payment ($247.11 ha-1, $100 ac-1) (IDALS, 2009; Hyberg, 2007). The soil rental rate was assumed to be the average cash rental rate for 2008-2010 for the state of Iowa ($447.26 ha-1, $181 ac-1) (ISU Extension, 2011c).

Table 5.10 EQIP payment schedule rates for Iowa for seven N reduction practices (Iowa NRCS, 2010)

	EQIP Practice Name	Practice Code	Payment schedule cost	Payment Unit	Min. Life (yr)	Yr of Payment	Payment ($/ha treated)	$TPVC_{Govt}$ ($/ha)
Controlled Drainage[†]	Drainage Water Management	554	$364.08	Per number of water control zones	1	1	$44.98	$44.98
Bioreactors[‡]	Denitrifying Bioreactor	747	$3999.50	Per bioreactor	10	1	$197.66	$197.66
Wetland[§]	Wetland Creation	658	$680.00	Per acre	15	1	$16.80	$16.80
Nitrogen Management – Rate[¶]	---	---	---	---	---	---	$0.00	$0.00
Nitrogen Management – Time[¶, ††]	Nutrient Management	590	$11.00	Per acre	1	1-3	$27.18	$78.45
Cover Crop[#, ††]	Cover Crop (and Green Manure)	340	$53.26	Per acre	1	1-3	$131.61	$379.83
Crop Rotation[††]	Conservation Crop Rotation	328	$52.00	Per acre	1	1-3	$128.50	$370.85

[†] Controlled drainage: Used scenario of 65 ha (160 ac), requiring eight zones.

[‡] Bioreactors: the EQIP specifies treatment of drainage from 12.1 ha (30 ac) which was less than the treatment area assumed here of 20.2 ha (50 ac). EQIP cost-share was not used in replacement years for bioreactors or controlled drainage.

[§] Wetlands: used CREP 30 yr contract incentives rather than EQIP cost share shown here.

[¶] Nitrogen management: A mid-range payment rate requiring only two additional enhancement practices was chosen.

[#] Cover crop: Used "cover crop winter hardy" rate for a winter cover of rye.

[††] EQIP funding for nutrient management, cover crop and crop rotation practices have three consecutive year payment time limits. Therefore, payments for these three practices were assumed to happen in the first three years of the total analysis period.

Inclusion of EQIP or CREP payments increased the cost effectiveness of the practices (Table 5.9 versus Table 5.11). Without government payments, the practices in order of cost effectiveness were (based on mean value): N timing modification, N application rate reduction, controlled drainage,

bioreactors, wetlands, crop rotation and cover crops. When government payments were included, wetlands and bioreactors became the third and fourth most cost effective practices, respectively (Figure 5.2).

Table 5.11 N removal EACs for seven drainage water quality practices including government payments and additional revenue (i = 4% and n = 50 yrs)

	Costs + Government Payment (EAC $/kg N)	
	Mean (StDev)	Median
Controlled Drainage	$1.92 ($1.51)	$1.64
Bioreactors	$1.44 ($0.92)	$1.27
Wetland	$0.12 ($0.32)	$0.09
Nutrient Management - Rate	-$1.62 ($0.00)	-$1.62
Nutrient Management - Time	-$14.30 ($12.43)	-$12.63
Cover Crop	$38.15 ($34.13)	$25.83
Crop Rotation	$16.39 ($11.46)	$14.75

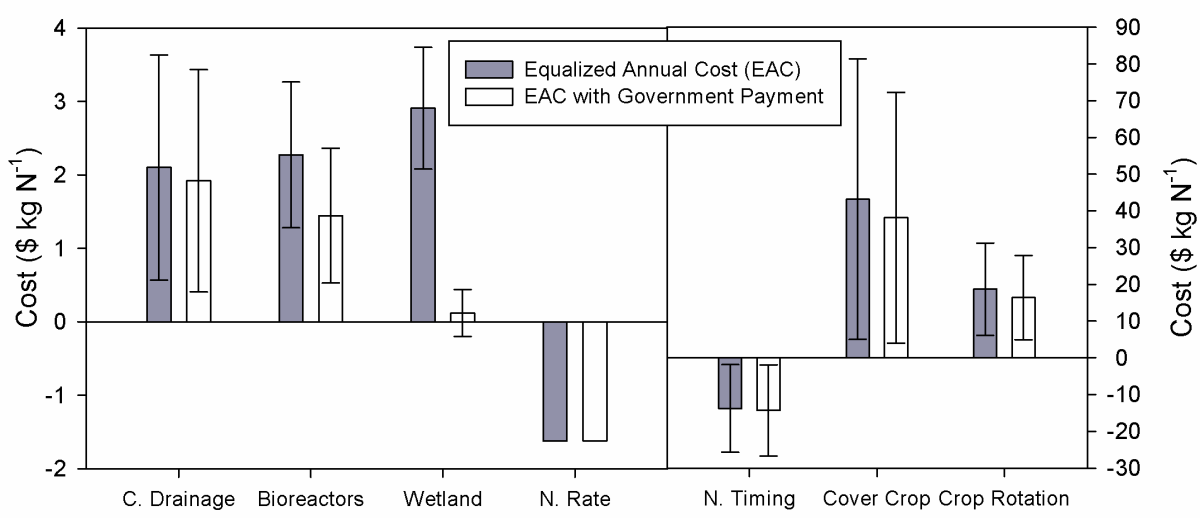

Figure 5.2 EAC $/kg N removed for seven agricultural practices with and without government payments

Conclusions

Each strategy provides landowners an additional distinct option for drainage water quality improvement and different strategies or combinations of such will be applicable in different locations. Here the nitrogen management practices were the most cost effective as both lowering the application rate (from 168 kg N ha^{-1} to 140 kg N ha^{-1}) and moving applications to spring resulted in negative costs. Of course, the scenarios here were limited in scope and there is a very wide range of possibilities with nitrogen management and application that could yield different results. Importantly,

a complete ban of fall fertilization could have large-scale economic effects which were not investigated in this farm-level analysis. The least cost effective practices were the in-field vegetative practices (i.e. cover crop and crop rotation) though these cost efficiencies had wide standard deviations and benefits like soil productivity and erosion protection were not quantified. The three constructed practices were comparable in terms of N removal effectiveness and in pre-cost share$ kg N^{-1}; wetlands were very cost effective when CREP incentives were included. A final important note is that while this study focused on water quality nitrate mitigation, several of these practices provide significant additional ecosystem services not quantified here.

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Supplemental Material: Appendix 1

Appendix 1 provides more detailed information on the TPVC calculations for each individual practice. The general functional models are described in the above text with Equations 5.2 through 5.4, but nuances unique to the practices can be explored with this supplemental material.

Controlled Drainage

The present value cost of establishment in year 1 for a controlled drainage system was calculated using:

$$C_{est,CD} = C_{CS} + C_E + C_C \qquad \text{(Eq. S1)}$$

Maintenance cost was a terminating annual series over the planning horizon of 50 years. The TPVC of controlled drainage establishment and maintenance was described as:

$$C_{CD} = C_{est,CD} + C_{main}\frac{(1+i)^n - 1}{i(1+i)^n} \qquad \text{(Eq. S2)}$$

Additionally, for the 50 year planning horizon, the structures would need to be replaced in year 40 (i.e. structures and contractor fees, Eq. S3) with the stop logs/gates replaced every eight years (Eq. S4):

$$C_{RepSt} = (C_{CS} + C_C)\left(\frac{1}{(1+i)^{40\,yr}}\right) \qquad \text{(Eq. S3)}$$

$$C_{RepG} = \sum^{every\ 8\ years}\left[(C_{Gate})\left(\frac{1}{(1+i)^{8\,yr}}\right)\right] \qquad \text{(Eq. S4)}$$

$$TPVC_{CD} = C_{CD} + C_{RepSt} + C_{RepG} \qquad \text{(Eq. S5)}$$

Where the $TVPC_{CD}$ was the total present value of costs associated with the establishment, maintenance, and replacement of parts for controlled drainage.

Table 5.12 Cost parameters for Controlled Drainage

Cost symbol	Item
C_{CS}	Control structures (including structure transport)
C_E	Design cost
C_C	Contractor fees
$C_{est, CD}$	**Total cost of establishment for controlled drainage**
C_{main}	Time to raise/lower Mowing/maintenance
C_{CD}	**Total cost of establishment and maintenance for controlled drainage**
C_{RepSt}	Control structure replacement at yr 40
C_{RepG}	Stop log/gate replacement every 8 yrs
$TPVC_{CD}$	**Total cost of establishment, maintenance and replacement for controlled drainage**

Bioreactors

The TPVC of bioreactor establishment in year 1 was:

$$C_{est,Bio} = C_{CS} + C_{WC} + C_E + C_C + C_S + C_{Misc} \quad \text{(Eq. S6)}$$

General maintenance and mowing of the bioreactor surface was considered a terminating annual series cost. The total present value for establishment and maintenance of a bioreactor was:

$$C_{Bio} = C_{est,Bio} + C_{main}\left[\frac{(1+i)^n - 1}{i(1+i)^n}\right] \quad \text{(Eq. S7)}$$

Replacement values were calculated with single sum, present value equations:

$$C_{Rep20} = (C_{WC} + C_C + C_S)\left(\frac{1}{(1+i)^{20\ yrs}}\right) \quad \text{(Eq. S8)}$$

$$C_{Rep40} = (C_{CS} + C_{WC} + C_C + C_S)\left(\frac{1}{(1+i)^{40\ yrs}}\right) \quad \text{(Eq. S9)}$$

$$C_{RepG} = \sum^{every\ 8\ years}\left[(C_{Gate})\left(\frac{1}{(1+i)^{8\ yr}}\right)\right] \quad \text{(Eq. S10)}$$

$$TPVC_{Bio} = C_{Bio} + C_{Rep20} + C_{Rep40} + C_{RepG} \quad \text{(Eq. S11)}$$

where $TPVC_{Bio}$ was the total present value of costs associated with the establishment, maintenance, and replacement of parts for a denitrification bioreactor.

Table 5.13 Cost parameters for Bioreactors

Cost symbol	Item
C_{CS}	Both control structures (including structure transport)
C_{WC}	Woodchip cost (including woodchip transport)
C_E	Design cost
C_C	Contractor fees
C_S	Seeding bioreactor surface (broadcast with tractor plus seed cost)
$C_{misc.}$	Misc. materials
$C_{est, Bio}$	**Total cost of establishment for a denitrification bioreactor**
C_{main}	Time to raise/lower stop gates and mowing/maintenance time
C_{Bio}	**Total cost of establishment and maintenance for a denitrification bioreactor**
C_{Rep20}	Replacement in yr 20 of woodchips, contractor fees and surface seeding
C_{Rep40}	Replacement yr 40 of woodchips, structures contractor fees and surface seeding
C_{RepG}	Gate replacement every 8 yrs
$TPVC_{Bio}$	**Total cost of establishment, maint., and replacement for a denitrification bioreactor**

Wetlands

The TPV of establishment costs in year 1 were calculated as:

$$C_{est,Wet} = C_E + C_C + C_S + C_{CS} + C_{WP} + C_{Land} \quad \text{(Eq. S12)}$$

The total present value for wetland establishment and maintenance was:

$$C_{Wet} = C_{est,Wet} + (C_{main}) \left[\frac{(1+i)^n - 1}{i(1+i)^n}\right] \quad \text{(Eq. S13)}$$

With replacement costs of the control structure and sheet pile weir in year 40 included (Eq. S14), the TPVC of wetland establishment, maintenance, and replacement of given components was given with Eq. S15.

$$C_{RepWet} = (C_{CS} + C_{WP} + C_C)\left(\frac{1}{(1+i)^{40\ yrs}}\right) \quad \text{(Eq. S14)}$$

$$TPVC_{Wet} = C_{Wet} + C_{RepWet} \quad \text{(Eq. S15)}$$

Table 5.14 Cost parameters for Wetlands

Cost symbol	Item
C_E	Design cost
C_C	Contractor fees
C_S	Seeding buffer (broadcast with tractor and seed cost)
C_{WP}	Weir Plate
C_{CS}	Control Structure
C_{Land}	Land acquisition
$C_{est,\ Wet}$	**Total cost of establishment for wetland**
TC_{main}	Time to manage
C_{Wet}	**Total cost of establishment and maintenance for a wetland**
C_{RepWet}	Control structure and weir replacement at yr 40 (contractor fees at reduced rate from initial installation)
$TPVC_{Wet}$	**Total cost of establishment, maintenance, and replacement for a wetland**

Nitrogen Management

The establishment costs for both rate and timing nitrogen management practices were similar and relatively simple:

$$C_{est,NM} = C_M + C_F \quad \text{(Eq. S16)}$$

Because an N management practice is an annual occurrence, there were no long term maintenance costs but, rather, the establishment cost and revenue impacts occurred every year (terminating annual series). The total present value for N management was calculated as:

$$C_{NM} = C_{est,NM}\left[\frac{(1+i)^n - 1}{i(1+i)^n}\right] - R_{yield,NM}\left[\frac{(1+i)^n - 1}{i(1+i)^n}\right] \quad \text{(Eq. S17)}$$

Costs for the baseline scenario were calculated in the same way and the marginal difference between the baseline and the practice was the final TPVC comparison value.

Table 5.15 Cost parameters for Nitrogen Management

Cost symbol	Item
C_M	Fertilizer application (including diesel for equipment)
C_F	Fertilizer
$C_{est,Baseline}$	Total cost of establishment for baseline application (168 kg/ha in Fall)
$C_{est,NM\ Rate}$	**Total cost of establishment at a lower rate (from 168 kg/ha to 140 kg/ha)**
$C_{est,NM\ Time}$	**Total cost of establishment of Spring application**
$R_{yield,\ Baseline}$	Annual baseline revenue
$R_{yield,\ Rate}$	Annual revenue from changed yields due to nitrogen management (Lower rate)
$R_{yield,\ Time}$	Annual revenue from changed yields due to nitrogen management (Spring application)
$C_{Baseline}$	Total cost of establishment and revenue impacts for baseline application
$C_{NM,Rate}$	**Total cost of establishment and revenue impacts at a lower application rate**
$C_{NM,Time}$	**Total cost of establishment and revenue impacts for Spring application**
$TPVC_{NM,Rate}$	**Marginal cost of lower application rate**
$TPVC_{NM,Time}$	**Marginal cost of Spring application**

Cover Crops (Cereal Rye)

Rye cover crop cost of establishment in year 1 was:

$$C_{est,CC} = C_{CovS} + C_P + C_{Kill} \qquad \text{(Eq. S18)}$$

Due to the yield loss of corn following a rye cover crop, a revenue loss was assumed to occur every other year during the planning horizon (i.e. a corn/soybean rotation). This was applied as a series of periodic annual payments where n was the number of years the practice was done (n = 50 yrs) and t was the period of the corn/rye/soybean rotation (t = two yrs). Thus, the total present value for cover crops was calculated as:

$$TPVC_{CC} = C_{est,CC}\left[\frac{(1+i)^n-1}{i(1+i)^n}\right] + R_{Difference}\left[\frac{(1+i)^{nt}-1}{((1+i)^t-1)(1+i)^{nt}}\right] \qquad \text{(Eq. S19)}$$

where $R_{Difference}$ was the revenue difference between a baseline corn crop and corn following a rye cover crop.

Table 5.16 Cost parameters for a Cover Crop

Cost symbol	Item
C_{CovS}	Seed costs
C_P	Planting Drill (including diesel)
C_{Kill}	Spraying (broadcast from tractor) plus herbicide cost
$C_{est,CC}$	**Total cost of establishment for cover crops**
$R_{yield,\ Baseline}$	Annual baseline revenue (no cover crop)
$R_{yield,\ CC}$	Annual revenue from changed yields due to cover crop
$R_{Difference}$	Difference in annual revenue from baseline
$TPVC_{CC}$	**Total cost of establishment and maintenance for cover crops**

Crop Rotation (Multiple Years of Alfalfa)

The cost of establishing alfalfa in year 3 was calculated as:

$$C_{est,CR} = C_{RotS} + C_P + C_{Prep} + C_{Herb} + C_{Labor} + C_F + C_H \qquad \text{(Eq. S20)}$$

In years 4 and 5 of the rotation, the alfalfa maintenance costs were estimated using:

$$C_{main,CR} = C_{Labor} + C_F + C_H \quad \text{(Eq. S21)}$$

For one five-year rotation, the TPVC was calculated with single sum costs of corn or alfalfa discounted by the appropriate years:

$$C_{CR,5} = C_{corn\ follow\ legume} + \left[\left(C_{corn\ follow\ corn}\right)\left(\frac{1}{(1+i)^1}\right)\right] + \left[\left(C_{est,CR}\right)\left(\frac{1}{(1+i)^2}\right)\right] + \left[\left(C_{main,CR}\right)\left(\frac{1}{(1+i)^3}\right)\right] + \left[\left(C_{main,CR}\right)\left(\frac{1}{(1+i)^4}\right)\right] \quad \text{(Eq. S22)}$$

A second total present value equation was developed for the revenue of this five year rotation:

$$R_{CR,5} = [R_{corn} \times 4.5\%\ yield\ boost] + \left[\left(R_{corn}\right)\left(\frac{1}{(1+i)^1}\right)\right] + \left[\left(\frac{R_{CR}}{3}\right)\left(\frac{1}{(1+i)^2}\right)\right] + \left[\left(R_{CR}\right)\left(\frac{1}{(1+i)^3}\right)\right] + \left[\left(R_{CR}\right)\left(\frac{1}{(1+i)^4}\right)\right] \quad \text{(Eq. S23)}$$

Total present value costs for the entire rotation were calculated as a series of periodic annual payments where n was the number of years the practice was done (n = 50 yrs) and t was the period of the total rotation (t = five yrs). This long-term TPVC for a rotation including alfalfa was described as:

$$TPVC_{CR} = C_{CR,5}\left[\frac{(1+i)^{nt}-1}{((1+i)^t-1)(1+i)^{nt}}\right] - R_{CR,5}\left[\frac{(1+i)^{nt}-1}{((1+i)^t-1)(1+i)^{nt}}\right] \quad \text{(Eq. S24)}$$

A similar procedure was done for a baseline corn soybean rotation and the marginal difference between this baseline TPVC using a 50 yr horizon and the diversified rotation TPVC using a 50 yr horizon was the final TPVC comparison value.

Table 5.17 Cost parameters for a Diversified Crop Rotation

Cost symbol	Item
C_{RotS}	Seed costs
C_P	Planting Drill (including diesel)
C_{Prep}	Soil preparation
C_{Herb}	Herbicide
C_{Labor}	Labor (pre-harvest)
C_F	Fertilizer
C_H	Harvesting – mowing and baling
$C_{est,CR}$	**Total cost of alfalfa establishment** (in yr 3)
$C_{main,CR}$	**Total cost of alfalfa maintenance** (in yr 4-5)
$C_{corn\ follow\ legume}$	Cost of growing corn in year 1
$C_{corn\ follow\ corn}$	Cost of growing corn in year 2
$C_{CR,5}$	**TPVC for five year diversified rotation**
R_{CR}	Alfalfa revenue (annual)
R_{corn}	Corn revenue (annual)
$R_{CR,5}$	**TPV revenue for five year diversified rotation**
$TPVC_{CR}$	**Total costs and revenue for diversified crop rotation for 50 yr horizon**
$C_{Baseline,5}$	Corn and soybean five yr rotation
$R_{Baseline,\ 5}$	Corn and soybean five yr rotation
$TPVC_{Baseline}$	Total costs and revenue for corn and soybean rotation for 50 yr horizon
Marginal Cost	**Marginal cost of the practice** ($TPVC_{Baseline}$ - $TPVC_{CR}$)

CHAPTER 6 GENERAL CONCLUSIONS

Denitrification Bioreactor Design

Designing agricultural drainage denitrification bioreactors for successful and consistent nitrate reduction is challenging in consideration of variable drainage flow rates, nitrate concentrations, and temperatures. However, the design process allows engineers to attempt to manage these "uncontrollable" parameters with "controllable" factors like bioreactor design geometry and length to width ratio. Here, a key finding at the pilot-scale was that nitrate removal was not significantly impacted by design geometry. This conclusion was confounded at the field-scale as the bioreactor with a unique trapezoidal cross-section (NERF bioreactor) had poor performance in general, perhaps unrelated to this design factor (e.g. management could be optimized to route less water though the reactor).

Another development was the discovery that the design method used here dictated retention time solely based on the design length. Longer reactors were theoretically capable of longer retention times, thus may have improved nitrate removal especially at higher flow rates. In the field-scale comparison, the shortest bioreactor (Greene Co.) maintained bioreactor load reductions of 46% to 68%, but because this reactor treated the lowest percentages of water between the sites, its total load reductions were only 27% to 33%. Load reductions at this bioreactor were significantly correlated with retention time ($\alpha = 0.05$), but perhaps with a wider and/or longer reactor at this site, a higher percentage of water could be treated while maintaining a high correlation between retention time and nitrate removal.

In an applied sense, this work has importantly contributed to a USDA NRCS interim design standard for denitrifying bioreactors in the state of Iowa. Though there is still much to learn about optimizing the design of these systems, the availability of an interim design standard has allowed cost-sharing through the EQIP program, and thus, has increased the number of demonstration and monitoring sites in Iowa. These efforts will hopefully precipitate greater understanding of these systems.

Retention time

In reactor engineering, retention time is often an important design parameter and this proved to be the case here. Because drainage waters naturally contain dissolved oxygen (DO), designing and operating a drainage bioreactor with a retention time sufficient to allow reduction of this DO is vital for the anaerobic process of nitrate reduction via denitrification. In the two pilot studies here, it was

clear that increased retention times provided increase nitrate removal. Interestingly, even at high retention times, if the bioreactor was not operating at a "steady-state" condition (i.e. the reactor had recently experienced fluctuating flow rates due to a drainage event), this retention time did not provide the expected removal.

The field-scale work here showed that bioreactor load reduction was positively correlated with retention time at three of four sites (significance of at least $\alpha = 0.10$), though load reduction was more strongly correlated with temperature. However, varying drainage flow rates mean that bioreactors will rarely operate under ideal design specifications. For example, during late summer, flow rates may be exceedingly low resulting in retention times which provide an excess of reducing conditions (e.g. conditions suitable for sulfate reduction and mercury methanogenesis; NERF bioreactor in September 2010). Alternatively, spring's high flow rates may not provide retention times which allow either DO removal or much denitrification to occur. Determining *in situ* retention times at field-scale denitrification bioreactors will be difficult without flow monitoring that allows estimation of flow rate and depth; tracer testing can also help elucidate *in situ* retention times and bioreactor hydraulics via comparison with tracer residence times and dispersion indices.

Hydraulics

The hydraulics of water moving through a denitrification bioreactor was shown to be important as, in controlled studies, increased flow rates during drainage hydrographs caused decreased retention times and reduced nitrate removal (Chapter 2 and 3). This was also observed at the field-scale with high flow events at the NERF bioreactor resulting in increased bioreactor effluent nitrate concentrations. Concentration reduction for these events was positively correlated with retention time, highlighting the interaction between flow events, retention time and nitrate removal performance.

Another important hydraulic issue at the field-scale was the occurrence of bypass flow. This untreated water can greatly reduce the overall efficiency of a bioreactor as mentioned above (e.g. Greene Co. site). However, it may also not be desirable to treat all the drainage volume as evidenced by the NERF bioreactor's low nitrate removal percentages; this bioreactor reduced bioreactor loads from 12% to 15% while treating 91% to 99% of the drainage volume. There may be a balance between bypass volume and load reduction that has yet to be optimized.

Cost

The financial work presented here is the first of its kind for drainage water quality practices in terms of its comprehensiveness and consistency (Chapter 5). Previously, comparison of water quality practices on a cost efficiency basis was difficult as past authors used varying methodologies which were not comparable with each other. Here, this was overcome by itemizing and discounting the costs for seven practices in a consistent manner, allowing "apples to apples" cost comparisons. At $1.44/kg N removed (mean value including government payments, 4% discount rate, 50 year planning horizon), denitrification bioreactors were a cost competitive practice. The other similar constructed remediation options of controlled drainage and wetlands were $1.92/kg N and $0.12/kg N, respectively. Using field-scale performance data from two sites in Chapter 4, a simplistic cost evaluation yielded similar cost competitive results. The Greene Co. and Hamilton Co. bioreactors had cost efficiencies ranging from $0.91/kg N to $4.72/kg N for an assumed fifteen year life (no discounting included).

It is hoped that the cost comparison work in Chapter 5 can be used as a land-owner decision tool as well as in policy development. However, it would be very dangerous for a policy-maker to make a blanket decision about water quality practices based solely on this cost-evaluation or any such individual assessment. Each specific farm and individual farmer presents unique characteristics that require individual approaches. No individual technology or management practice will meet all water quality, soil conservation and agricultural production needs throughout any given watershed.

Future work

Chapter 1 of this dissertation concluded denitrification bioreactors were a viable option for reducing the amount of nitrate in agricultural drainage, but more research on their design and performance is needed. This is the primary research need for this technology to progress fully into the research and demonstration phase before moving into a possible more full-scale implementation phase. Suggestions of studies include field-scale comparisons of different design methods consisting of multiple years of data from a number of sites. Additionally, tracer testing to investigate bioreactor hydraulics at diverse sites should be done; this could help identify successful design approaches and operational parameters like retention time. Performing such research will be valuable not only in the US Midwest, but in many locations around the globe where nitrate pollution from agriculture is problematic. Continued research in denitrification systems worldwide under different environmental conditions will help significantly strengthen the body of science in this field.

Additionally, more robust studies on deleterious effects should be done. Organic flushing upon start up is well established, and research in this area should now focus on documenting the

effectiveness of techniques to minimize this. Moreover, investigation of nitrous oxide emissions under fluctuating field conditions, both from the bioreactor surface and from liquid effluent, is needed. Mercury methylation is a final important issue that could potentially be very inflammatory. More work is needed to improve the understanding of this process within bioreactors with this followed by investigation of management techniques for abatement.

Most importantly, it is vital that future work be communicated amongst the scientific community and especially amongst those doing work in this field. Research which is not communicated does not contribute to the body of science and may result in other research groups "re-inventing the wheel". Unfortunately, such "research-for-the-sake-of-research" ultimately does not benefit society. This invitation for communication is made here so that advancements in the design and operation of denitrification bioreactors for agricultural drainage can continue to be conveyed allowing these systems to potentially have a greater positive impact upon water quality.