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A Practice-oriented Review of Woodchip Bioreactors for Subsurface Agricultural Drainage

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A Practice-oriented Review of Woodchip Bioreactors for Subsurface Agricultural Drainage

Abstract
Woodchip or denitrification bioreactors are an innovative, engineering-based technology to reduce the amount of nitrate in agricultural drainage. Increased interest in improving water quality in areas impacted by agricultural drainage has given bioreactors a boost of publicity over the past several years. While bioreactors continue to be an area of active research and are not a silver bullet to address drainage water quality concerns, the growing number of bioreactor installations by practitioners not involved in research demonstrates a need for a practice-oriented review of important aspects of these systems. This article provides context for enhanced-denitrification treatment of agricultural drainage, discusses the design and installation of bioreactors, and presents factors affecting their nitrate removal performance. Additionally, this review offers ideas for management and monitoring of agricultural drainage bioreactors. Bioreactors are a promising technology for improving drainage water quality, but much work remains to understand and optimize their performance. With additional evaluation and improved monitoring of bioreactors, a more complete picture of the potential contribution of these systems will be developed.

Keywords
Agricultural drainage, Denitrification bioreactor, Nitrate, Woodchip

Disciplines
Agriculture | Bioresource and Agricultural Engineering

Comments
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ABSTRACT. Woodchip or denitrification bioreactors are an innovative, engineering-based technology to reduce the amount of nitrate in agricultural drainage. Increased interest in improving water quality in areas impacted by agricultural drainage has given bioreactors a boost of publicity over the past several years. While bioreactors continue to be an area of active research and are not a silver bullet to address drainage water quality concerns, the growing number of bioreactor installations by practitioners not involved in research demonstrates a need for a practice-oriented review of important aspects of these systems. This article provides context for enhanced-denitrification treatment of agricultural drainage, discusses the design and installation of bioreactors, and presents factors affecting their nitrate removal performance. Additionally, this review offers ideas for management and monitoring of agricultural drainage bioreactors. Bioreactors are a promising technology for improving drainage water quality, but much work remains to understand and optimize their performance. With additional evaluation and improved monitoring of bioreactors, a more complete picture of the potential contribution of these systems will be developed.

Keywords. Agricultural drainage, Denitrification bioreactor, Nitrate, Woodchip.
Denitrification is the microbially mediated anaerobic reduction of \( \text{NO}_3^- \) to dinitrogen gas (\( \text{N}_2 \)) (eq. 1), is one of the most important possible fates of \( \text{NO}_3^- \) in the soil (Tiedje, 1994). Artificial drainage modifies the nitrogen cycle as well as the hydrologic cycle in agricultural systems because 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 likelihood of the soil solution to be fully denitrified before it becomes drainage water (Moorman et al., 2010).

\[
5C + 4\text{NO}_3^- + 2H_2O \rightarrow 2N_2 + 4\text{HCO}_3^- + CO_2 \quad (1)
\]

Denitrification requires (1) N oxides (e.g., \( \text{NO}_3^- \), \( \text{NO}_2^- \), \( \text{NO} \), \( \text{N}_2\text{O} \); the electron acceptors) (eq. 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 begin using \( \text{NO}_3^- \) as electron acceptors in their respiration electron transport chain. This limiting DO level varies amongst the numerous denitrifying organisms (Korom, 1992), and 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, the majority of which are heterotrophic (Korom, 1992).

\[
\begin{align*}
\text{NO}_3^- \quad \text{nitrate reductase} & \rightarrow \text{NO}_2^- \quad \text{nitrite reductase} & \rightarrow \text{NO} \quad \text{nitric oxide reductase} \\
\text{N}_2\text{O} \quad \text{nitrous oxide reductase} & \rightarrow \text{N}_2
\end{align*} \quad (2)
\]

After nearly complete reduction of \( \text{NO}_3^- \) and with further decreases in reducing conditions, obligate anaerobes become active and use other electron acceptors, such as sulfate (\( \text{SO}_4^{2-} \)), manganese (Mn (IV)), and iron (Fe(III))(Korom, 1992). The order in which 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 \( \text{N}_2 \), carbon dioxide (\( \text{CO}_2 \)), and bicarbonate (\( \text{HCO}_3^- \)) (eq. 1). The main product of interest is usually the gaseous phase nitrogen, although the \( \text{HCO}_3^- \) can be important because this release of alkalinity increases the solution pH (Korom, 1992; Metcalf and Eddy, 2003). The predominant nitrogenous end product, \( \text{N}_2 \), is stable due to its molecular triple bonds, although denitrification can also produce nitrous oxide (\( \text{N}_2\text{O} \)) (eq. 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 \( \text{N}_2\text{O:} \text{N}_2 \) denitrification production ratio towards \( \text{N}_2\text{O} \) (Chapin III et al., 2002). Additionally, the microbiology of the bacteria may be important, with not all denitrifiers possessing genes capable of encoding the nitrous oxide reductase.

**Enhanced-Denitrification Treatment with Solid Carbon Sources**

Relatively recent developments in the field of water remediation have led to advancements with enhanced-denitrification permeable reactive barriers. The “enhancement” is provided by the added solid carbon source, which both encourages aerobic respiration to reduce solution DO so denitrification can proceed and offers a carbon source for denitrifiers (Schipper et al., 2005). In the first published work in this field, three 200 L 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 \( \text{NO}_3^- \) concentrations were reduced from 2 to 6 mg \( \text{NO}_3^- \)-N/L to less than 0.02 mg \( \text{NO}_3^- \)-N/L, thus validating the potential use of organic media to enhance \( \text{NO}_3^- \) removal. Similar work soon followed with the investigation of septic wastewater treatment (Robertson and Cherry, 1995); 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). Shortly after Blowes et al.’s (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). Recent 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., 2010b) 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 \( \text{NO}_3^- \) in agricultural drainage waters is now receiving increased interest. Table 1 provides a review of drainage denitrification bioreactor performance at multiple scales.
Table 1. Review of denitrification treatment for agricultural drainage.

<table>
<thead>
<tr>
<th>Source</th>
<th>Site</th>
<th>Volume (m³)</th>
<th>Influent NO₃⁻-N Concentration</th>
<th>Retention Time</th>
<th>Percent Reduction (concentration or load noted)</th>
<th>Nitrate-N Removal Rate</th>
<th>Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Field-Scale Drainage Treatment Studies</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Blowes et al., 1994</td>
<td>Ontario, Canada</td>
<td>0.2 (barrels)</td>
<td>2 to 6 mg/L</td>
<td>1-6 d</td>
<td>Nearly 100% concentration</td>
<td>NA</td>
<td>Partially buried in a stream bank</td>
</tr>
<tr>
<td>Wildman, 2001</td>
<td>South of Chatsworth, Ill. (#1)</td>
<td>27.2</td>
<td>approx. 4 to 16 mg/L</td>
<td>NA</td>
<td>Nearly 100% concentration</td>
<td>NA</td>
<td>4.0-ha treatment area</td>
</tr>
<tr>
<td>Wildman, 2001</td>
<td>South of Chatsworth, Ill. (#2)</td>
<td>27.2</td>
<td>approx. 1 to 18 mg/L</td>
<td>NA</td>
<td>Nearly 100% concentration</td>
<td>NA</td>
<td>5.3-ha treatment area</td>
</tr>
<tr>
<td>van Driel et al., 2006a</td>
<td>Ontario, Canada; lateral flow</td>
<td>17.2</td>
<td>11.8 mg/L (mean)</td>
<td>9 h (during tracer test)</td>
<td>33% concentration</td>
<td>NA</td>
<td>Fine and coarse wood media</td>
</tr>
<tr>
<td>Jaynes et al., 2008</td>
<td>Central Iowa</td>
<td>38.9</td>
<td>19.1 to 25.3 mg/L (control plot)</td>
<td>NA</td>
<td>40%-65% load</td>
<td>0.62 g N/m³/d</td>
<td>Flow-through woodchip walls on sides of tile pipe</td>
</tr>
<tr>
<td>Moorman et al., 2010</td>
<td>Central Iowa</td>
<td>38.9</td>
<td>20 to 25 mg/L</td>
<td>24 h required to reduce influent to ≤10mg/L</td>
<td>NA</td>
<td>23.6 mg N/kg wood/d</td>
<td>Retention time conclusion based on field data</td>
</tr>
<tr>
<td>Chun et al., 2010</td>
<td>Decatur, Ill. (west)</td>
<td>55.8</td>
<td>269.9 g NO₃⁻N/g (slug test)</td>
<td>4.4 h</td>
<td>47% load</td>
<td>NA</td>
<td>2.0-ha treatment area</td>
</tr>
<tr>
<td>Verma et al., 2010</td>
<td>Decatur, Ill. (west)</td>
<td>55.8</td>
<td>Approx. 5 to ≥20 mg/L</td>
<td>NA</td>
<td>81%-98% load</td>
<td>NA</td>
<td>2.0-ha treatment area</td>
</tr>
<tr>
<td>Woli et al., 2010</td>
<td>East-Central Illinois (De Land, Ill.)</td>
<td>76.9</td>
<td>2.8 to 18.9 mg/L</td>
<td>26 min to 2.8 h</td>
<td>23%-50% load</td>
<td>6.4 g N/m³/d</td>
<td>14-ha treatment area</td>
</tr>
<tr>
<td>Verma et al., 2010</td>
<td>East-Central Illinois (De Land, Ill.)</td>
<td>76.9</td>
<td>Approx. 3 to 16 mg/L</td>
<td>NA</td>
<td>42% - 48% load</td>
<td>NA</td>
<td>14-ha treatment area</td>
</tr>
<tr>
<td>Verma et al., 2010</td>
<td>Decatur, Ill. (east)</td>
<td>NA</td>
<td>Approx. 4 to 15 mg/L</td>
<td>NA</td>
<td>54% load</td>
<td>NA</td>
<td>6.5-ha treatment area</td>
</tr>
<tr>
<td>Ranaivoson et al., 2010</td>
<td>Claremont, Minn.</td>
<td>NA</td>
<td>Approx. 11 to 28 mg/L</td>
<td>32 h for 50% conc. reduction</td>
<td>18%-47% load</td>
<td>NA</td>
<td>10.5-ha treatment area</td>
</tr>
<tr>
<td>Ranaivoson et al., 2010</td>
<td>Dundas, Minn.</td>
<td>NA</td>
<td>Approx. 7 to 14 mg/L</td>
<td>NA</td>
<td>35%-45% load</td>
<td>NA</td>
<td></td>
</tr>
<tr>
<td>Christianson et al., 2012a</td>
<td>Central Iowa; Pekin</td>
<td>18</td>
<td>1.2 to 8.5 mg/L</td>
<td>NA</td>
<td>22%-74% load</td>
<td>0.38-3.78 g N/m³/d</td>
<td>1.2-ha treatment area</td>
</tr>
<tr>
<td>Christianson et al., 2012a</td>
<td>Northeast Iowa, NERF</td>
<td>128</td>
<td>9.9 to 13.2 mg/L</td>
<td>NA</td>
<td>12%-14% load</td>
<td>0.86-1.56 g N/m³/d</td>
<td>Trapzoidal cross section; 6.9-ha treatment area</td>
</tr>
<tr>
<td>Christianson et al., 2012a</td>
<td>Central Iowa, Greene Co.</td>
<td>127</td>
<td>7.7 to 15.2 mg/L</td>
<td>NA</td>
<td>27%-33% load</td>
<td>0.41-7.76 g N/m³/d</td>
<td>19-ha treatment area</td>
</tr>
<tr>
<td>Christianson et al., 2012a</td>
<td>Central Iowa, Hamilton Co.</td>
<td>102</td>
<td>7.7 to 9.6 mg/L</td>
<td>NA</td>
<td>49%-57% load</td>
<td>0.42-5.02 g N/m³/d</td>
<td>20.2-ha treatment area</td>
</tr>
</tbody>
</table>

| Laboratory- and Pilot-Scale Drainage Treatment Studies | | | | | | | |
| Christianson et al., 2011b | Central Iowa; pilot-scale | 0.71 | 10.1 mg/L (mean) | 4 to 8 h | 30%-70% concentration | 3.8-5.6 g N/m³/d | Mixed hardwood chips; different design geometries |
| Christianson et al., 2011c | Palmerston North, New Zealand; pilot-scale | 0.53 | 7.7 to 35.6 mg/L | 1.5 to >15 h | 14%-37% load | 2.1-6.7 g N/m³/d | Pinus radiata chips |
| Chun et al., 2009 | Illinois; lab column | 0.30 | 10.4 to 33.7 mg/L | 2.6-12.0 h and >15 h | 10%-40% and 100% concentration (respectively to retentions) | NA | Three parameter estimation, first order reaction |
| Greenan et al., 2009 | Illinois; lab column | 0.01 | 50 mg/L | 9.8, 3.7, 2.8, and 2.1 d | 100, 64, 52, and 30% load (respectively to retentions) | 11-15 mg N/kg woodchip/d | |
| Cooke et al., 2001 | Illinois; lab column | 0.001 | 25 mg/L | 8 h | Nearly 100% concentration | NA | Woodchips at 25°C |
| Doheny, 2002 | Illinois; lab column | 0.001 | 25 mg/L | 10 h | 60% concentration (i.e., to below 10 mg/L) | NA | Woodchips at 10°C |

[a] Annual flow-weighted mean.
One of the first peer-reviewed studies of enhanced denitrification to directly treat drainage water was presented by Van Driel et al. (2006a), who investigated a bioreactor in Canada consisting of alternating layers of fine and coarse woody material. In the United States, Cooke et al. (2001) were the first to explore enhanced-denitrification treatments for tile drainage. Early work from this group in Illinois explored carbon media (Cooke et al., 2001), additions of gravel to reduce compaction (Wildman, 2001), and retention time requirements for different media under 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 United States.

The Environmental Programs and Services division of the Iowa Soybean Association has funded installation and management of a number of bioreactors in Iowa (ISA, 2010). Government officials and programs in Iowa have also 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 from Iowa investigated carbon-media selection and properties, flow rate and retention time impacts, and design geometry (Greenan et al., 2006; Greenan et al., 2009; Christianson et al., 2010). Field studies from Iowa and Minnesota have documented performance, longevity, N$_2$O emissions, and removal of compounds other than NO$_3^-$ (Christianson et al., 2012a; Jaynes et al., 2008; Moorman et al., 2010; Ranaivoson et al., 2010). Applied research of drainage bioreactors has recently been facilitated by several NRCS Conservation Innovation Grants (CIGs) in South Dakota, Ohio, and Iowa (South Dakota NRCS, 2011; USDA NRCS, 2011; Ohio NRCS, 2012). Outside the U.S. Midwest, studies in coastal states have used a research-scale bioreactor containing immobilized sludge to treat NO$_3^-$ in drainage waters (Hunt et al., 2008).

### Denitrification Bioreactor Performance Factors

Many factors can affect bioreactor NO$_3^-$ removal performance, including retention time, temperature, and microbiology. Schipper et al. (2010a) provided a complete discussion of reaction kinetics and longevity; thus these are not discussed in detail (table 2). There has been no consensus reached on enhanced-denitrification NO$_3^-$ removal kinetics, though the review by Schipper et al. (2010a) reported the design of these systems could functionally use zero-order kinetics. Bioreactor longevity depends upon several factors, including the type and amount of carbon source, flow characteristics, consistency and level of saturation, and physical changes in the media over time (Schipper et al., 2010a). Performance life estimates are often on the order of several decades, with empirical data showing at least 10 years (table 2). The

<table>
<thead>
<tr>
<th>NO$_3^-$ Removal Kinetics</th>
<th>Longevity and Performance Life</th>
<th>Deleterious Effects</th>
</tr>
</thead>
<tbody>
<tr>
<td>Van Driel et al., 2006a</td>
<td>Stream-bank pilot bioreactors: 72-yr estimate based on stoichiometry</td>
<td>Cameron and Schipper, 2010; Gibert et al., 2008; Healy et al., 2012; Schipper et al., 2010a</td>
</tr>
<tr>
<td>Assumed zero order for field reactor</td>
<td>Assumed zero order for field reactor</td>
<td>Start-up leaching can contain high concentrations of dissolved organic carbon (DOC), biological/chemical oxygen demand (BOD/COD) and nitrogen (NH$_4^+$, TKN); minimize through use of higher C:N media or starting-up under high flow conditions with by-pass flow occurring</td>
</tr>
<tr>
<td>Blowes et al., 1994</td>
<td>Stream-bank pilot bioreactors: 72-yr estimate based on stoichiometry</td>
<td>Start-up leaching can contain high concentrations of dissolved organic carbon (DOC), biological/chemical oxygen demand (BOD/COD) and nitrogen (NH$_4^+$, TKN); minimize through use of higher C:N media or starting-up under high flow conditions with by-pass flow occurring</td>
</tr>
<tr>
<td>Robertson and Cherry, 1995</td>
<td>Septic treatment layer: 20-yr estimate based on stoichiometry and safety factor of only 10% carbon available</td>
<td>Cameron and Schipper, 2010; Gibert et al., 2008; Healy et al., 2012; Schipper et al., 2010a</td>
</tr>
<tr>
<td>Robertson et al., 2008</td>
<td>Septic treatment wall: empirical observation of 15 yr</td>
<td>Moorman et al., 2010</td>
</tr>
<tr>
<td>Robertson et al., 2008</td>
<td>Septic treatment wall: empirical observation of 15 yr</td>
<td>No significant difference between N$_2$O released from woodchip wall vs. control</td>
</tr>
<tr>
<td>Robertson et al., 2009</td>
<td>Septic treatment wall: empirical observation of 15 yr</td>
<td>No significant difference between N$_2$O released from woodchip wall vs. control</td>
</tr>
<tr>
<td>Leverenz et al., 2010</td>
<td>Tile line woodchip walls: 37-yr estimate based on half-life of saturated chips, empirical observation of 9 yr</td>
<td>Elgood et al., 2010</td>
</tr>
<tr>
<td>First order due to low NO$_3^-$ concentrations and temperatures (pilot testing)</td>
<td>Elgood et al., 2010</td>
<td>Less than 2% of removed NO$_3^-$ was released as dissolved N$_2$O, and N$_2$O loss was highest in winter months during less complete NO$_3^-$ removal</td>
</tr>
<tr>
<td>Roberton et al., 2010</td>
<td>Groundwater treatment wall: 66-yr estimate based on total carbon remaining, empirical observation of 14 yr</td>
<td>Woli et al., 2010</td>
</tr>
<tr>
<td>Zero order for lab study (inflow: 3.1 to 49 mg NO$_3^-$-N/L)</td>
<td>Surface N$_2$O bioreactor emissions were negligible</td>
<td>Woli et al., 2010</td>
</tr>
<tr>
<td>Long et al., 2011</td>
<td>Surface N$_2$O bioreactor emissions were negligible</td>
<td>Shih et al., 2011</td>
</tr>
<tr>
<td>Woli et al., 2010</td>
<td>Surface N$_2$O bioreactor emissions were negligible</td>
<td>In-stream bioreactor was a source of methyl mercury when sulfate reducing conditions were present; to minimize, maintain &gt; 0.5 mg NO$_3^-$-N/L in effluent</td>
</tr>
</tbody>
</table>

Table 2. Review of reaction kinetics, longevity, and deleterious effects of enhanced-denitrification systems.
The performance life of drainage bioreactors may be complicated by their fluctuating flow depths and levels of saturation. Less consistently saturated woodchips towards the surface of these systems will degrade more quickly than deeper-placed chips (Moorman et al., 2010; Christianson et al., 2012b). Currently, most drainage bioreactors are designed for an expected life of at least 10 years (USDA NRCS, 2009).

**RETENTION TIME AND HYDRAULICS**

The retention time in the bioreactor is dictated by the reactor flow rates combined with the design factors of media porosity and bioreactor flow volume. In reactor design, the retention time of the liquid and solutes inside the reactor is important because there must be sufficient time for the desired treatment processes to occur. 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 agricultural production.

Operationally, very low retention times experienced in drainage denitrification bioreactors may not be sufficient to reduce the influent drainage DO to a level that allows denitrification to proceed. Very high retention times provide excellent NO$_3^-$ removal but also the potential for oxidation reduction potentials (ORPs) indicative of undesirable processes, like sulfate reduction and mercury methylation. Relatively higher retention times in drainage denitrification systems typically correlate with higher NO$_3^-$ removal (table 1). For example, Chun et al. (2009) reported NO$_3^-$-N concentration reductions of 10% to 40% at retention times of generally less than 5 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. Retention time has also been correlated with NO$_3^-$ removal at the field scale in Iowa and Illinois (Christianson et al., 2012a; Woli et al., 2010).

The use of inflow and outflow control structures in Midwestern bioreactors 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 (fig. 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

![Figure 1. Schematic of denitrification bioreactor for agricultural drainage with locations of pressure transducers as a suggested flow monitoring scheme noted with *(Adapted from Christianson and Helmers, 2011).](image-url)
the most in-field management (fig. 1). Active management of the stop logs is an important part of bioreactor operation. The stop log height in this capacity control structure can be lowered during low flows (e.g., late summer) to prevent the retention time from becoming too high and can be increased during 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 (Van Driel et al., 2006b; Robertson and Merkley, 2009).

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; Van Driel et al., 2006a; Robertson and Merkley, 2009; Shih et al., 2011). Sulfate reduction is of concern because (1) it represents a loss of carbon for denitrifiers, (2) it produces hydrogen sulfide that can be a noxious gas (though agricultural bioreactors are typically not in confined spaces), and (3) it is closely linked to the methylation of mercury. Bioreactors can be designed and managed to minimize sulfate reduction by retaining very low concentrations of NO$_3^-$ in the effluent (e.g., 0.5 mg NO$_3^-$/N/L) (Robertson and Merkley, 2009; Shih et al., 2011). If hydrogen sulfide (i.e., a rotten egg smell) is noted around the outflow control structure, the stop log height should be lowered to allow water to flow unrestricted through the reactor (Christianson and Helmers, 2011).

Hydrologically, many drainage systems experience very low flows or dry periods even during an active drainage season. Fortunately, bioreactor start-up once flow resumes after dry periods has not been problematic (Van Driel et al., 2006a). 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 immediately preceding 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).

**Temperature**

Drainage water entering a bioreactor will likely have temperatures that vary seasonally, with early spring temperatures just above freezing and late summer temperatures at greater than 15°C (Christianson et al., 2012a). As a biologically mediated transformation, denitrification in a bioreactor is influenced by water temperature, though NO$_3^-$ removal has been documented at water temperatures as low as 2°C to 4°C (Robertson and Merkley, 2009). Many studies show increased NO$_3^-$ removal at higher temperatures (Volokita et al., 1996; Diaz et al., 2003; Cameron and Schipper, 2010; Hoover, 2012), and the Q$_{10}$ value (i.e., the factor by which the reaction rate increases for every 10°C rise in temperature) for these systems ranges from less than 1 to nearly 3, with most values around 2 (approximately ± 0.5) (Hoover, 2012; Cameron and Schipper, 2010; Robertson and Merkley, 2009; Van Driel et al., 2006a; Warneke et al., 2011a). Cooke et al. (2001) used the Van’t Hoff-Arrhenius Law to show increased retention times were required at lower temperatures. Because this temperature relationship is so important, Cameron and Schipper (2011) attempted to artificially increase the temperature at a denitrification bed in New Zealand with passive solar heating; this attempt yielded only a 3.4°C mean bioreactor temperature increase and no significant increase in N removal rate.

Temperature also interacts with other factors in ways that impact bioreactor design. For example, drainage N loads are greatest in the spring, out of synch with maximum temperatures (and enhanced NO$_3^-$ removal) in summer, which makes bioreactor treatment optimization a challenge (Mirek, 2001; Randall and Goss, 2001). The effect of temperature on bioreactor performance is significant (Christianson et al., 2012a), but with better understanding of operational parameters, like seasonal retention time management, it is possible that sensitivity to temperature can be reduced. 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 anyway in the spring in the Midwest to address the higher spring flow rates. Recommendations for control-structure management for bioreactors in Illinois are available (University of Illinois, 2011), but management approaches will likely differ throughout the Midwest depending upon seasonality of drainage and timing of field operations.

**Microbiology**

Denitrifiers are abundant in the environment meaning no inoculation has been required for these systems to date (Schipper et al., 2010a) other than the addition of soil, typically in small amounts [e.g., 1 kg by Blowes et al. (1994) or 1 L by Christianson et al. (2011c)]. However, slow bioreactor start-up after one early spring installation was attributed to the slow growing microbial community (Wildman, 2001).

Denitrifiers are the primary denitrification vehicle, but fungi may also provide an important enhancement due to their ability to release soluble carbon substrates for use by denitrifiers (Appleford et al., 2008). Appleford et al. (2008) reported that denitrifiers were present on both woodchip surfaces and in bioreactor solution, and Moorman et al. (2010) reported woody media walls supported higher levels of denitrifiers than the surrounding soil. Denitrification sites may not be limited to the chip surface; 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). Bioreactor microbial communities vary with depth, in the direction of flow, and also may change over the year (Andrus, 2011). Recently, Warneke et al. (2011b) documented that the bacterial community in a small-scale
woody media bioreactor 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. This work also reported that the genes required for denitrification were present at four times the concentration (number of nitrite reductase gene copies/g dry substrate) at 27.1°C versus 16.1°C (Warneke et al., 2011b). Andrus (2011) suggested increased consideration of the microbial community within a bioreactor, either through inoculation of optimal species or environmental management to shift the community to high performing NO$_3^-$ removers, could enhance bioreactor performance.

Bacteria other than denitrifiers, mainly observed as the presence of biofilms, have been documented at bioreactor sites (Chun et al., 2009). These biofilms may cause clogging in the lines or control structures. Flushing (via stop log control, if possible) or agitation may be the best management option (Van Driel et al., 2006a; 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 a field-scale drainage bioreactor.

**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., 13 mm/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 very rarely operate at this maximum flow rate. Flow rates within a given year range from zero to this maximum (or greater, since this is theoretical) with low and high 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 peak flow rate at a chosen design retention time. This downsizing of the peak estimated flow rate concurs with reports that designing a bioreactor to treat the peak drainage flow rate may not be economical (Van Driel et al., 2006a). Similar design method is used by the USDA NRCS in Iowa to design bioreactors that are seeking cost-share funding 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 \times W) and treatment area on an efficacy or performance curve. For example, approximately 9.3 m$^2$ of bioreactor surface area would be required for every 1.2 to 1.4 ha of drainage area (100 ft$^2$ per 3.0 to 3.5 acre) to achieve a 60% load reduction (Verma et al., 2010; University of Illinois, 2011). A design table from Wildman (2001) allows estimation of a required bioreactor volume based on drainage area and drainage coefficient; unfortunately, the exact drainage area and coefficient are not known for many drainage systems. Another informal method from the Midwest has used the rough estimate of approximately 3.3 m of bioreactor length for every 0.4 ha of drainage (UMAN Extension and MN Department of Ag., 2011). Schipper et al. (2010a) suggested a simple design approach based upon mass removal, in which published reaction rates can be used to calculate a required volume given the proposed bioreactor design avoids NO$_3^-$ limiting conditions (which may not always be valid for drainage bioreactors). Finally, the stoichiometry of the denitrification reaction (eq. 1) can be used to develop the volume of carbon required, but this theoretical approach may be prone to error as many other microbial reactions will also utilize the carbon (Wildman, 2001). The Christianson et al. (2011a) and the University of Illinois (2011) design methods currently used in Iowa and Illinois, respectively, are the most widespread design procedures available in the Midwest. There has been no consensus regarding one “optimal” drainage bioreactor design method to date, and various methods result in a range of bioreactor sizes (fig. 2). Ongoing work with several CIGs aims to address this, and additional field-scale performance information will facilitate more informed design decision-making.

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, and Robertson and Merkley (2009) installed an in-stream bioreactor in a drainag ditch. 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). Inclusion of baffles within a bioreactor or designing bioreactors in series or parallel (Cooke et al., 2001) may help maximize treatment. These ideas may be interesting in the research realm; however, such thoughts 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 worth consideration (Christianson and Tyndall, 2011); bioreactors can 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 to consider prior to installation of a drainage bioreactor include site evaluation, component availability, and construction details (table 3). Bioreactors are not suitable for every field or tile drainage system, and consideration should be given to the factors presented in table 3 before design work begins.
Installation generally consists of positioning the control structures, excavating and filling the trench, laying geofabric 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 (Van Driel et al., 2006a; University of Illinois, 2011; K. Kult at Iowa Soybean Association, 2011, personal communication). This highlights the importance of site evaluations that allow comprehensive consideration of any potential bioreactor designs for sites with highly permeable soils. A minimum of 1.5 m of non-perforated tile pipe should separate the inlet or outlet manifolds, which are usually assembled of perforated tile pipes, from the control structures (University of Illinois, 2011).

A mounded soil cover is used to help prevent subsidence as the woodchips settle (Schipper and Vojvodic-Vukovic, 1998; University of Illinois, 2011). Additionally, a soil cover may be beneficial for mitigating N₂O emitted through the bioreactor surface. Nitrous oxide emissions from the soil cover of pilot bioreactors have been observed to be lower than emissions directly from the surface of the woodchips (Christianson, unpublished data). Similar, but non-significant, results were documented at non-soil-covered versus soil-covered bioreactors in Illinois (Woli et al., 2010). Elgood et al. (2010) suggested designing systems for complete NO₃⁻ removal to mitigate N₂O emissions, but this may exacerbate sulfate reduction and mercury methylation. In terms of the total nitrogen balance in a watershed, Moorman et al. (2010) noted that if NO₃⁻ in drainage is denitrified less efficiently downstream, more N₂O may ultimately be released than if the drainage water was treated in a bioreactor.

Table 3. General factors to be considered for denitrification bioreactor installation.

<table>
<thead>
<tr>
<th>Pre-Design</th>
<th>Site Conditions</th>
<th>Materials Availability</th>
<th>Construction</th>
</tr>
</thead>
<tbody>
<tr>
<td>Drainage area</td>
<td>Available space for the bioreactor</td>
<td>Inflow and outflow control structures</td>
<td>Uniform and consistent filling of media</td>
</tr>
<tr>
<td>Tile locations</td>
<td>Soil type</td>
<td>Suitable fill media</td>
<td>Use of a liner and woodchip covering fabric</td>
</tr>
<tr>
<td>Tile size</td>
<td>Proximity to sensitive or public water bodies</td>
<td>Plastic lining and geofabric to cover woodchips</td>
<td>Mounding soil cover</td>
</tr>
<tr>
<td>Tile slope</td>
<td>Equipment traffic-ability</td>
<td>Non-perforated pipe near the structures</td>
<td>Reseeding with appropriate seed mixture</td>
</tr>
<tr>
<td>Drainage coefficient</td>
<td>Accessibility for installation, management, and sampling</td>
<td>Construction labor and equipment</td>
<td>Transport and use of surplus spoil</td>
</tr>
<tr>
<td>Number of surface intakes (aim to minimize bioreactor influent sediment)</td>
<td>Identification of individual(s) to manage and monitor the bioreactor</td>
<td>Labor for annual maintenance</td>
<td>Construction safety equipment and procedures</td>
</tr>
<tr>
<td>Installation timing: availability of contractors and restrictions due to crops in the fields or nearby nesting birds</td>
<td>Seed for cover</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
CARBON MEDIA

The type of carbon fill is one of the most important considerations of denitrification systems because media properties affect 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 a variety of materials may be most practical in different locations with tested media including corn cobs, corn stalks, wood media (multiple sizes and species), wheat and barley straw, and pine and almond shells (Soares and Abeliovich, 1998; Diaz et al., 2003; Greenan et al., 2006; Cameron and Schipper, 2010; Hashemi et al., 2010). In general, woody media is preferred due to cost, conductivity, longevity, and C:N (Schipper et al., 2010a). Recently, however, Warneke et al. (2011b) suggested a mixture of maize cobs and woodchips would provide a combination of maximized \( \text{NO}_3^- \) removal rates due to the cobs, and good microbial properties for denitrifiers and minimized deleterious leaching from the woodchips. Nevertheless, in an agricultural setting, it may be difficult to uniformly mix two media types to curtail potential preferential flow.

Media chemical properties can most notably affect longevity and organic flushing. Woody media is the most recommended material, although there can be a wide variety of C:Ns among tree species, with lower C:N materials generally not recommended due to flushing losses or potential mass degradation (Gilbert et al., 2008; Christianson et al., 2012b). Organic flushing can be minimized through selection of more-optimal material or starting up under high-flow conditions (table 2) (Schipper et al., 2010a; K. Kult at Iowa Soybean Association, 2011, personal communication). Site selection is also important as careful consideration should be given to proposed bioreactors that discharge to sensitive or public waters.

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 (Schipper and Vojvodic-Vukovic, 1998; Cameron and Schipper, 2010), both of which have a C:N in the range of several hundred (Greenan et al., 2006; McLaughlan and Al-Mashaqbeh, 2009). In terms of agricultural-system practicality, piles of municipal yard waste have garnered attention for potential use as bioreactor fill. Though this source may be readily available and inexpensive, it is doubtful this material will have the longevity (Christianson et al., 2012b) or sustained media properties (e.g., porosity, hydraulic conductivity) required to avoid frequent replenishment.

Physical properties of the fill material (i.e., porosity, particle size, and hydraulic conductivity) influence bioreactor hydraulics and can change over time. Porosities of woody chipped media typically range from 0.6-0.86 (Ima and Mann, 2007; Chun et al., 2009; Christianson et al., 2010a; Robertson, 2010), with \textit{in situ} values reported at 0.65 to 0.79 (Van Driel et al., 2006a; Chun et al., 2010; 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 colloquial term “woodchip.” Commonly, chipped material described by Christianson et al. (2010a) had 50% of particles fall between 13 to 25 mm sizes, and Chun et al. (2009) and Woli et al. (2010) used chips of which 66% and 62%, respectively, fell in the 6- to 25-mm range. Several studies reported no consistent, significant differences in \( \text{NO}_3^- \) removal for coarse versus fine or ground materials (Greenan et al., 2006; Van Driel et al., 2006a) and have recommended coarse materials for preferable flow properties (Van Driel et al., 2006a). 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. A common question related to woodchip compaction involves the ability to farm over bioreactors. Currently no field-scale work has investigated the sustained compaction potential of farm equipment on woodchip bioreactors, thus bioreactors are recommended as an edge-of-field practice.

Hydraulic conductivity is one of the most important physical parameters of the media considering the relatively high flow rates a drainage denitrification bioreactor may experience. Schipper et al. (2004) reported incorrect estimation of conductivity led to preferential flow around a groundwater denitrification wall. Average conductivities for wood material range from 0.35 to 11.6 cm/sec for sawdust to 61-mm chips, respectively (Cameron and Schipper, 2010), and the design model from Christianson et al. (2011a) was based on an average conductivity of 9.5 cm/s (Christianson et al., 2010a). \textit{In situ} values range from 1.2 to 11 cm/s (Robertson et al., 2005a; Van Driel et al., 2006a), and over time, conductivity can decrease, possibly due to biofilm formation (Chun et al., 2009; Robertson and Merkley, 2009) or consolidation. Reactors containing larger particles may experience relatively larger reductions in conductivity compared to reactors with smaller particles (Cameron and Schipper, 2010).

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 ha to over 40 ha) (Christianson et al., 2012a), with most bioreactors averaging between $7,000 to $9,000. The largest installation cost components of these reactors were the woodchips, which ranged from 13 to 55% of the total cost, and the contractor fees, which ranged from 23 to 54%. Across a range of bioreactors, the average cost of woodchips, contractor fees, control structures, and supplies has been approximately $2,900, $3,300, $1,800, and $500, respectively (Christianson et al., 2012a). Other cost reports for bioreactors have been on the order of $2,030 USD (i.e., $2,000 Canadian) and $3,200 USD (Van Driel et al., 2006a; UMN Extension and MN Department of Ag., 2011). Schipper et al. (2010a) provided the first cost efficiency calculation of a denitrification system at $2.39 to $15.17
per kg N. The low end of this range was very similar to a newer cost report of $2.27 per kg N ± $0.99 for bioreactors from Christianson (2011) (not including governmental cost-share incentives). This cost efficiency comparison showed constructed wetlands, controlled drainage, and bioreactors all had mean cost efficiencies less than $2.00 per kg N when government cost share was included (Christianson, 2011).

**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 sampling to better understand the denitrification process and nitrogen balance in these systems (Schipper and Vojvodic-Vukovic, 1998; Greenan et al., 2006; Elgood et al., 2010; Moorman et al., 2010; Long et al., 2011; Warneke et al., 2011a). These methods provide interesting and valuable research data, but it is unlikely such methods will be used to monitor farmer-managed bioreactors, and thus a description of simpler methods is useful. The most basic representation of drainage bioreactor function is provided through comparison of inflow and outflow NO\(_3\)-N concentrations based on grab sampling. Although this method is easiest, without supporting evidence provided by some of the relatively straightforward monitoring techniques described below, many questions would remain about the bioreactor’s performance.

**SAMPLING**

Grab sampling from the inflow and outflow structures is the most fundamental 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. In terms of sampling frequency, Wang et al. (2003) reported that for estimation of N mass losses from drainage water, the probability of being within ±15% of the “true” mass loss was 92% for weekly samplings and 68% for monthly samplings. Rodrigue (2004) recommended bioreactor sampling every 4 d, though this may be more intensive than is practical at demonstration sites. A number of denitrification system 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 that are of research interest but may not directly pertain to NO\(_3\) removal performance (e.g., BOD, TKN, NH\(_4\), DOC, SO\(_4\)^2-; Blowes et al., 1994; Robertson and Merkley, 2009; Van Driel et al., 2006a). For locations in which the bioreactor discharges directly to a stream, it may be important to collect samples from the receiving stream, especially start up. Regarding timing of sample collection, Van Driel et al. (2006a) did not collect samples within 48 h of a rainfall event, to avoid diluted samples; similarly, Woli et al. (2010) did not collect samples during two high drain flow events under the assumption that no NO\(_3\) removal would occur during these conditions.

To capture higher resolution data, an autosampler (e.g., Model 6712 Portable Sampler, Teledyne Isco, Lincoln, Nebr.) can be used to collect samples from a control structure over the course of a drainage high-flow event or during a tracer test. Sampling can also 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 at which flow volumes were also recorded.

**FLOW MEASUREMENT**

Sampling and analysis of NO\(_3\)-N concentrations provides some insight on NO\(_3\) removal, although the addition of flow rate data allows a more complete analysis of NO\(_3\) load reduction. The most elementary flow monitoring method utilizes a container of known volume (e.g., a bucket) and a stop watch and can be done at bioreactor sites that discharge directly to surface water. Limitations of this method include that the inflow cannot be measured and that bioreactor flow cannot be determined if bypass is occurring. Moreover, 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 data loggers to record water depth in the control structures (e.g., Model 3001 Levellogger Junior, Solinst, Georgetown, Ontario, Canada or WL16 Water Level Loggers, Global Water Instrumentation, Inc., Gold River, Calif.) Placement of these devices is noted in figure 1, with the inflow and outflow structure transducers allowing measurement of bypass flow and bioreactor flow, respectively. Limitations here are that the transducers may break if the drainage water freezes, and that they give no indication of water movement, which makes standing water or backwards flow in the structure problematic for flow calculations. Moreover, conservation of the mass of water within the reactor must be assumed when only two pressure transducers are used; in other words, the non-measured bioreactor inflow is assumed to equal the measured outflow. 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 et al., 2012a; Woli et al., 2010). Other more expensive flow monitoring methods include the use of doppler-based velocity meters (e.g., 2150 Area Velocity Module, Teledyne Isco, Lincoln, Neb. or FloPro, MACE, Shawnee Mission, Kan.) or digital or mechanical totalizing flow meters with data loggers (Jaynes et al., 2008). Other non-drainage denitrification treatments have used mechanical water meters, inline sonic flow meters, and impellor water meters (Schipper et al., 2010b; Warneke et al., 2011a).
IN SITU MEASUREMENTS

Additional information provided by measurement of parameters such as dissolved oxygen (DO), temperature, pH and oxidation reduction potential (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 common laboratory equipment (e.g., 3300i pH field meter, WTW Inc., College Station, Tex.) and provide interesting information as temperature impacts the microbiology of denitrification and pH is typically increased by the denitrification process (Warneke et al., 2011a). Proper calibration and maintenance of probes will help maintain scientific rigor for these readings.

Anoxic conditions conducive to denitrification can be monitored using DO measurements. Dissolved oxygen has been reduced to below 0.5 mg DO/L within approximately 25% of the length of the inlet at several sites (Christianson et al., 2011b; Van Driel et al., 2006a; Warneke et al., 2011a). Media bags are another useful in situ research tool for investigating longevity and carbon dynamics, as there have been differences noted in woodchips exposed to anaerobic vs. aerobic bioreactor zones (Christianson et al., 2012b; Moorman et al., 2010).

An ORP probe (AKA redox or oxidation reduction potential probe; e.g., SenTix ORP Electrode Probe, WTW Inc., College Station, TX) allows slightly more insight than DO measurements into conditions conducive to denitrification (Blowes et al., 1994; Christianson et al., 2011b; Van Driel et al., 2006a). Because the use of different electron acceptors (i.e., oxygen, NO\textsubscript{3}\textsuperscript{-}, SO\textsubscript{4}\textsuperscript{2-}, etc.) varies based on the strength of the reducing conditions, ORP measurements can provide supporting data for the occurrence of these various reduction reactions. For example, sulfate reduction occurs at ORPs of -50 mV to -250 mV, which is below the optimal ORP range for denitrification (+50 mV to +50 mV ORP; YSI Environmental, 2008). 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 (YSI Environmental, 2001).

TRACER TESTING

Tracer tests are commonly used in reactor engineering to investigate hydraulic performance and residence characteristics. Non-ideal hydraulic performance in plug-flow reactors includes short-circuiting, where a certain volume of flow arrives at the outlet of the reactor earlier than expected, and dead zones, where a certain volume of the reactor traps or detains flow. These conditions can be caused by poor mixing, poor design, and the location of inlets and outlets (Metcalf and Eddy, 2003; Cameron and Schipper, 2011).

In denitrification systems, bromide or chlorine are typically used as conservative tracer compounds to better study hydraulic properties and flow characteristics (Schipper et al., 2004; Schipper et al., 2005; Van Driel et al., 2006a; Cameron and Schipper, 2011; Christianson et al., 2011b; Christianson et al., 2011c; Christianson et al., 2012c). Tracer testing by van Driel et al. (2006a) allowed investigation of fine and coarse woody media layers within one bioreactor and showed the majority of flow was transmitted through the coarse layer. Cameron and Schipper (2011) used tracer testing to determine vertical flow regimes (i.e., up- or down-flow) minimized short circuiting; however, most agricultural drainage bioreactors are designed for horizontal flow due to practicality. It is suggested that by-pass flow be avoided during a drainage bioreactor tracer test (Chun et al., 2010).

WELLS AND PIEZOMETERS

The installation of wells or piezometers in a bioreactor is useful for sampling to determine approximate zones in which NO\textsubscript{3}\textsuperscript{-} removal or other processes are occurring and to provide locations for in situ probe measurements (e.g., temperature, electrical conductivity, pH, ORP, DO) (Christianson et al., 2012c; Van Driel et al., 2006a; Warneke et al., 2011a). Samples are usually collected from the piezometers via a pump or syringe with well evacuation or purging recommended prior to sample collection (Van Driel et al., 2006a). Pressure transducers can likewise be fitted in the wells to determine the head difference across the reactor and to document in situ drainage hydrographs (Christianson et al., 2012c; Chun et al., 2010). Depth to water can also be manually measured in wells with the use of a measuring tape (e.g., Model 101 or 102 Water Level indicators, Solinst, Georgetown, Ontario, Canada) (Christianson et al., 2011b). Installation of piezometer “bundles,” with each individual piezometer screened at a different depth, allows measurement and sampling of the depth axis of the reactor in addition to across the surface area axes (Van Driel et al., 2006a; Van Driel et al., 2006b).

CONCLUSIONS

Although fully addressing drainage water quality issues will require a variety of in-field and edge-of-field approaches, enhanced-denitrification systems to reduce NO\textsubscript{3}\textsuperscript{-} loadings from agricultural drainage are a promising new technology. However, this new water quality option is not without limitations or additional research needs. More field-scale bioreactor data are urgently needed to evaluate design methods, quantify potential deleterious effects, and develop better management procedures for optimized performance as this technology begins to move quickly from the research to demonstration phase. 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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