Denitrification bioreactors for agricultural drainage nitrate treatment in New Zealand

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Abstract
Nitrate loadings from agricultural drainage causes serious water quality concerns in many parts of the world including the US Midwest and New Zealand. Targeted nutrient management is necessary to increase the environmental sustainability of agricultural systems in these locations. Denitrification bioreactors for reducing nitrate loads in agricultural drainage are one such option that has shown potential as a nutrient removal technology in the US. Denitrification bioreactors consist of carbon-filled (wood media) trenches through which drainage is routed allowing enhancement of denitrification. Work presented here provides a background on denitrification bioreactor design, nitrate removal effectiveness, and possible application to drainage systems in New Zealand's intensively grazed pasture lands.

Key Words
Denitrification, water quality, nitrate, drainage, nutrient management

Introduction
Nitrate loadings from agricultural drainage affect surface waters worldwide. In the US Midwest, this nitrate pollution has been implicated as a major cause of the hypoxic zone in the Gulf of Mexico (Rabalais *et al.* 1996; USGS 2000). Reducing nitrate loadings in the Mississippi River Basin leading to the Gulf of Mexico is a pressing water quality issue in the United States (USEPA 2007). Similarly, nitrate pollution in agricultural drainage has caused concern in New Zealand in recent years. Much of this degradation in water quality in New Zealand can be attributed to the rapid expansion of the dairy industry over the past two decades. In particular, nitrate enrichment of surface waters is a consequence of dairying on artificially drained grazing land (Monaghan *et al.* 2002; Houlbrooke *et al.* 2004).

In the US Midwest, fields that are artificially drained with subsurface tile drains are predominantly cropped with corn and soybeans (Dinnes *et al.* 2002). Drainage patterns in the upper Midwest (Iowa, Minnesota, Illinois) typically consist of precipitation-dependent high flows in the spring and summer (February-July) (Gentry *et al.* 2009) trailing off in the late summer around harvest time (Gentry *et al.* 2000; Randall 2004). Drain flow in winter (December-January) is uncommon in many northern Midwestern soils because there is little precipitation and the soil may be frozen (Kalita *et al.* 2006). In the Midwest, drainage systems are typically designed for flows of 9.5 to 12 mm /d (ISU Extension 2008). In contrast, New Zealand's climate results in the majority of the drainage volume occurring in the high rainfall winter months (June – August) (Monaghan *et al.* 2002) with drainage occurring as discrete events with relatively high short-term flow rates of up to 4 mm/hr (Magesan, *et al.* 1995).

Typical NO_3-N concentrations in tile drainage in the US Midwest range from 10 to 20 mg NO_3-N/L (Mirek 2001; Kalita *et al.* 2006) with maximum values of 52 mg NO_3-N/L (Gentry *et al.* 2000). These concentrations provide annual loadings mostly in the range of 25-30 kg NO_3-N/ha (Kalita *et al.* 2006), though loadings can be as high as 65-70 kg NO_3-N/ha (Kladivko *et al.* 1991; Jaynes *et al.* 1999). The majority of the N load in drainage from New Zealand's intensive pastoral land also occurs as NO_3-N at average concentrations and annual N loadings similar to the values reported for the Midwest (Monaghan *et al.* 2002; Houlbrooke *et al.* 2004).

Subsurface drainage systems act as short-circuiting, conduits for nitrate loadings to surface waters, reducing the opportunity for denitrification to occur in the soil (Mirek 2001; Kellman 2005). Several options have been proposed to reduce these nitrate loadings including increased use of wetlands and cover crops and better nutrient management and crop rotations (Dinnes *et al.* 2002; Jaynes *et al.* 2004). However, regardless of better management strategies, nitrate concentrations in drainage waters can still exceed the drinking water standard because of the variability of agricultural systems (Kladivko *et al.* 2004). In New Zealand, the main source of nitrate leaching from intensively grazed pastoral soils is cattle urine spots (Di and Cameron 2002a). There are a limited number of mitigation strategies, such as decreased grazing times or use of nitrification inhibitors, that have been shown to reduce nitrate leaching from grazed soils (de Klein and Ledgard 2001; Di and Cameron 2002b). However, provision of a wide range of options for reducing nitrate leaching is important to enable farmers to choose a strategy that will work well for their farming system.

Denitrification bioreactors could provide an important new option for treatment of waters from artificial drainage systems under pasture in New Zealand. Recently, denitrification treatments have been successful in reducing nitrate in groundwater in New Zealand and for treating septic effluent in Canada (Schipper and Vojvodic-Vukovic 2000; Schipper *et al.* 2005; Robertson and Cherry 1995). Edge-of-field nitrate removal from agricultural drainage can be enhanced with the use of subsurface, carbon-filled excavations through which drainage is routed (i.e. a denitrification bioreactor). This treatment system has proven effective in the US Midwest, where nitrate concentration reductions within bioreactors can be very high (upwards of 60%) (Appleford *et al.* 2008; Jaynes *et al.* 2008). For agricultural bioreactors, wood media is recommended as it provides good longevity, good performance and is readily available (Cooke *et al.* 2001; Greenan *et al.* 2006). Bioreactors can be incorporated into grassed buffers and occupy an area that is typically 0.1% of the drainage treatment area (drainage areas of 10 to 40 ha).

Currently in the US Midwest, denitrification bioreactors are designed to treat up to twenty percent of the peak drainage flow rate at between four and eight hours of retention. It would be too costly to build bioreactors capable of treating 100% of the peak estimated drainage flow rate (Van Driel *et al.* 2006a). Installing denitrification bioreactors into artificial drainage systems in New Zealand will be a unique challenge given their very high flow rates (Magesan *et al.* 2004). It may be possible to treat only a relatively small percentage of the drainage water (i.e. significantly less than the 20% of peak flow rate achieved for the Midwest situation) (Table 1).Where practicable, containment ponds may be constructed to stabilize flow rates, and thereby allow treatment of a greater portion of the drainage event. The type of bioreactor fill media, temperature and the required retention time to allow denitrifiers to become competitive in New Zealand systems may also differ from what has been previously found in the US Midwest.

Nitrate removal rates from denitrification systems have been consistently high especially in comparison with wetland systems (Table 2). Robertson and Merkley (2009) and Van Driel *et al*. (2006a) found bioreactor removal rates could be forty times greater and an order of magnitude greater than wetland rates, respectively.

Conclusion
Denitrification bioreactors have shown potential to reduce agricultural drainage nitrate loads in the US Midwest and, therefore, their application as a mitigation strategy in New Zealand is also worth investigation. *In situ* trials are necessary to test this technology under New Zealand's climate and field conditions before it can be offered as a viable edge-of-field nitrate reduction option to landowners.

Table 1. Possible design differences for Midwestern and New Zealand denitrification bioreactors using the current Iowa design model based on peak flow rate and required retention time.

	Area drained (ha)	Peak Flow Rate (mm/hr)	Proportion of peak flow rate treated	Volume of bioreactor material required (m^3)
Midwest USA	16	0.5	20%	266
Manawatu – NZ	5	4	10%	338

Table 2. Nitrate removal rates for denitrification treatment systems reported in literature.

Reference	System description	Location	Nitrate Removal Rate	
			System Volume	System Surface Area
Schipper and Vojvodic-Vukovic 2000	Sawdust flow-through wall for groundwater	Cambridge, New Zealand	0.11-0.43 g $N/m^3/d$	N/A
Schipper *et al.* 2005	Sawdust flow-through wall for groundwater	Cambridge, New Zealand	1.4 g $N/m^3/d$	N/A
Fahrner 2002 thesis cited in Schipper *et al.* 2005	Sawdust flow-through wall for groundwater	Busselton, Australia	15 g $N/m^3/d$	N/A
Jaynes *et al.* 2008	Flow-through woodchip wall between crop rows	Iowa, USA	0.62 g N/m^3/day	N/A
Van Driel *et al.* 2006a	Fine and coarse wood media agricultural drainage reactor	Ontario, Canada	2.3 g $N/m^3/d$	2.5 g $N/m^2/d$
Van Driel *et al.* 2006b	Fine wood media, riparian groundwater treatment	Ontario, Canada	1.2-5 g $N/m^3/d$	0.7 to 3.5 g $N/m^2/d$
Christianson ,2009	Pilot scale woodchip bioreactors	Iowa, USA	3.8-5.6 g $N/m^3/d$	1.5-3.4 g $N/m^2/d$
Xue *et al.* 1999	Constructed wetland (drainage water)	Illinois, USA	N/A	0.05-0.28 g $N/m^2/d$
Hernandez and Mitsch 2007	Constructed wetland (river water)	Ohio, USA	N/A	0.005-0.043 g $N/m^2/d$

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