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Artificial Sinks: Opportunities and Challenges for Managing Offsite Nitrogen Losses

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Abstract: Advanced control measures are needed after nitrogen (N) leaves agricultural fields and begins to flow through a catchment. Bioreactors – simple, wood-chip filled trenches – and constructed wetlands afford additional treatment to reduce N from agricultural lands, but their success requires informed adoption and placement. Rates of N removal in field studies of bioreactors vary based on carbon substrate, hydrologic setting, temperature, N loading and hydraulic residence time. From limited measures in constructed wetlands, N removal can be substantial and influenced by seasonality. Additional research is needed to understand uncertainty and variability associated with these systems. Further synthesis of data from research and demonstration sites combined with geospatial tools will help guide the design and siting of these systems at regional and local scales. Professional and student training and web-based data information exchange will advance the adoption and strategic placement of appropriate bioreactor and constructed wetland designs to remove edge-of-field N contributions.

Keywords: *Nitrogen, contructed wetlands, denitrification*

The Green Revolution of the 20th Century generated unparalleled levels of agricultural productivity based on advances in crop breeding – and ample inputs of inexpensive nitrogen (N) fertilizer. Unintended losses of N from production agriculture contribute to increases in algal biomass in estuaries and marine waters leading to loss of fisheries and spawning habitats and creation of multitudes of hypoxic "dead zones" across the globe (Conley et al. 2009; Howarth et al. 2000). Providing food security for a global population that is projected to exceed 9 billion by 2050 (U.N. Population Division 2007) is likely to include even more intensive use of N fertilizer – with great implications for coastal waters. In addition, reactive N can undergo transformations that generate nitrous oxide (N_2O) – a potent greenhouse gas – leading the U.S. National Academy of Engineering to declare managing the N cycle as one of the Grand Challenges of the 21st Century (http://www.engineeringchallenges.org/cms/challenges.aspx) – and the Stockholm Resilience Center to proclaim the N cycle as one of three planetary boundaries that has been exceeded globally (Rockstrom et al. 2009).

Improved crop varieties, cropping systems, precision management, and soil and plant testing hold promise for greater N use efficiency at the field scale (Cassman et al. 2002). However, reactive N is notoriously leaky, suggesting that additional control measures are needed after reactive N leaves the field and begins to flow through a catchment.

Predictive geospatial tools, such as the USGS SPARROW model (Preston et al. 2009), have targeted source locations within a growing number of watersheds that have high potential for delivery of waterborne N to coastal estuaries (e.g., the Mississippi Basin, Alexander et al. 2008; Chesapeake Bay, Preston and Brakebill 1999; the Southeast U.S., Hoos and McMahon 2009). These tools recognize that certain areas of the landscape function as removal sites (i.e., sinks) for waterborne N (National Research Council 1993). In these N sink areas, denitrification converts soluble nitrate to N gas, and plant and microbial biomass retains N. Nitrogen sinks include riparian wetlands, reservoirs and lower-order (headwater) stream reaches. These locations are characterized by extended retention times and flow paths that enhance interaction of nitrate-enriched waters with labile organic matter (Groffman et al. 2009). However, in many areas of the Mississippi River Basin nitrate sink areas have been removed and/or bypassed by tile drainage, which is important in transporting nitrate from fields to streams (David et al. 2010).

Where natural sinks are absent, artificial sinks, such as constructed wetlands (Tanner et al. 2005) or carbon bioreactors – simple, wood-chip filled trenches (Schipper et al. 2010a) – hold great promise for reducing edge-of-field N losses. These artificial sinks are positioned to intercept and promote denitrification in drainage waters or N-rich ground water, particularly in settings where edge-of-field N losses are dominated by nitrate-N (Gentry et al. 1998; Goolsby et al. 1999; Nolan 2001). Artificial sinks are now starting to be employed in an array of climatic, geophysical and agricultural settings. Some practices are eligible for USDA EQIP support in select states (e.g., Iowa, Arkansas, and Illinois), and more widespread adoption could occur as research advances on seasonal performance and design criteria.

Field testing of bioreactors and constructed wetlands has shown that these systems can remove nitrate-N in a range of climatic conditions and be integrated into different land uses if designed to meet site-specific conditions. There are emerging general design principles for both systems that can foster the development of best management practice (BMP) guidelines for specific spatial or geographic conditions, climate regimes, and agricultural practices. Research will play a critical role for these guidelines to address the current level of uncertainty in the N removal efficacy linked to site variables.

Artificial sinks afford additional treatment options for reducing N contamination from agricultural lands, but their success requires informed adoption and placement. In this paper we (1) summarize the current state of knowledge on bioreactors and constructed wetlands; (2) highlight research needed to address uncertainties; (3) discuss the potential for geospatial tools to guide managers in siting these artificial N sinks; and (4) explore opportunities and challenges for implementation of these systems.

Bioreactors for Offsite Nitrogen Removal

Overview

Denitrifying bioreactors are a relatively recent approach for reducing N loads into receiving waters (Schipper et al. 2010a). They were designed with an understanding of how natural systems (e.g., wetlands, soils, sediments) remove nitrate from water through denitrification, the microbial conversion of nitrate to N gases under anaerobic conditions with a carbon (C) energy source (Seitzinger et al. 2006). In denitrifying bioreactors, the energy source is a solid C substrate, often fragmented plant material, such as wood chips although other materials (such as corn cobs) have also been utilized (Cameron and Schipper 2010, 2011). Under water-saturated conditions, such as found in ground water or from pipe flow and channelized discharges, this plant material degrades slowly, creating anaerobic conditions and labile carbon (C) that can foster microbial denitrification.

Two basic forms of denitrifying bioreactors have been tested – denitrification beds and denitrification walls (Schipper et al. 2010a). A denitrification bed (Figure 1) is usually a large lined container, filled with a particulate C source, with nitrate-bearing water fed in at one end and discharged at the other. In general, these beds are used to treat nitrate-rich discharges from conveyance systems, e.g., tile drainage or effluents (e.g., Woli et al. 2010; Schipper et al. 2010b), but have also been tested

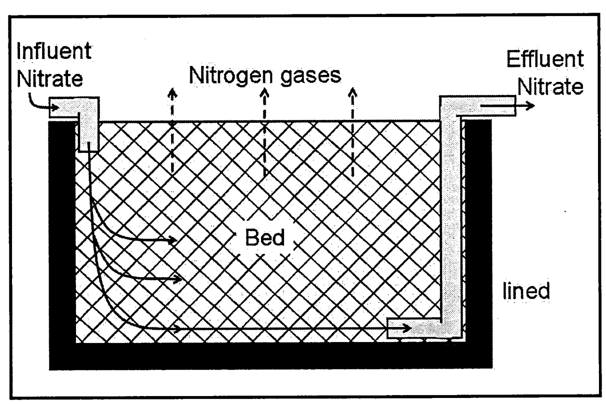

Figure 1. Schematic of denitrification bed in cross section (from Schipper et al. 2010a).

in the base of streams (Robertson and Merkely 2009). Denitrification walls (Figure 2) generally treat nonpoint discharges of nitrate-rich ground water before it reaches surface water or tile drains (e.g., Jaynes et al. 2008; Moorman et al. 2010; Schipper et al. 2005). A trench of soil is excavated perpendicular to ground water flow and back-filled with a wood chip or soil mix. Bioreactors are not simple "cookie-cutter" designs that can be added to any site generating high nitrate losses. Rather, the design of bioreactors needs to be tailored to accommodate site-specific conditions that recognize the range of temperatures, flow rates, discharge points, nitrate concentrations and nitrate removal goals found at sites in different settings and regions.

Performance

The performance of denitrifying bioreactors has been tested in laboratory, pilot scale and field-scale experiments. Lab studies have generally focused on comparisons of different potential C sources to optimize hydraulic performance and N removal (e.g., Gibert et al. 2008; Greenan et al. 2006; Cameron and Schipper 2010). The majority of field testing has been for "proof of concept" within agricultural landscapes or for treating wastewaters and has been focused in Canada (Robertson et al. 2008; Robertson and Merkley 2009), New Zealand (Schipper et al. 2004, 2005, 2010b), Australia (Schipper et al. 2010a) and in several of the States, including Iowa (Moorman et al. 2010), Illinois (Woli et al. 2010), Rhode Island (Addy and Gold 2008), and California (Leverenz et al. 2010). A wide range of nitrate removal rates (2 to 22 g N m^{-3} d^{-1} in beds; 0.014 to 3.6 g N m^{-3} d^{-1} in walls) have been reported in field-based bioreactor studies which generally reflect differences in C substrates,

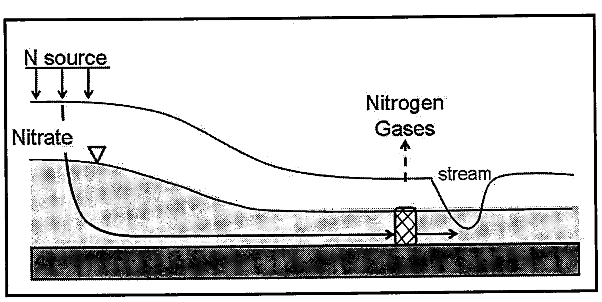

Figure 2. Schematic of denitrification wall in cross section (from Schipper et al. 2010a).

hydrologic setting, temperature, seasonal or site variation in N loading and hydraulic residence time (Schipper et al. 2010a). Reported rates of nitrate removal can also be conservative (less than the potential rate) in situations where the bioreactor removes all incoming nitrate such that the bioreactor is underutilized due to N limitation.

Until recently, it was not clear how long denitrifying bioreactors would continue to remove nitrate but three studies have now demonstrated that bioreactors constructed with wood chips or sawdust will remove nitrate for nine years or more (Long et al. 2011; Moorman et al. 2010; Roberston et al. 2008). These studies also estimated the future performance of denitrifying bioreactors by examining decay rates of wood material in denitrification walls and estimated bioreactor material half life as between 4.6 and 37 years in Iowa depending on sample depth (Moorman et al. 2010) and as 11 years in New Zealand (Long et al. 2011). Warneke et al. (2011) measured total losses of C from a large denitrification bed and estimated a life time of up to 39 years. These studies suggest that nitrate removal will be sustained for decades once the bioreactors are constructed, although with a likely declining efficiency through time. Based on a functional life time of 20 years, Schipper et al. (2010a) estimated a removal cost of between US$2.39-15.17 per kg of N, which compared well with other agricultural management techniques, such as controlled drainage, soil testing, wetlands, and fall cover crops. The lower cost estimate was when the landowner constructed the bioreactors using materials on site and demonstrates the need for robust design criteria that are usable by the land manager.

Hydrologic Considerations

Hydrologic site conditions are critical in designing a bioreactor. Wall bioreactors are passive systems, restricted by construction practicalities to the upper 1-2 meters of ground water. They can only treat ground water that is intercepted. Figure 2 displays optimal treatment conditions in an aquifer with a shallow confining layer allowing for ground water flow to intersect the denitrification wall. In deep aquifers, where the denitrification wall is not installed to the confining layer depth, flow paths may go below the wall (Figure 3) restricting the extent of treatment. Saturated hydraulic conductivity (K) in a bioreactor wall versus the surrounding media may control site flow paths and affect performance. If the denitrification wall's K is lower than the surrounding media – which may occur if labile C structures in the wall break down – flow will bypass the wall. Recent research has demonstrated that hydraulic efficiency can be maintained by using large wood chip size without a decline in nitrate removal performance (Cameron and Schipper 2010) due to the large secondary porosity of wood chip material. Alternative C sources, such as maize cobs, can provide greater N removal, but there is concern that they may not support nitrate removal for as long and decline in hydraulic performance more quickly. Alternatively, denitrification walls with high K relative to the surrounding media have been found to induce ground water upwelling (Figure 4), extending the functional treatment depth (Robertson et al. 2005). Matching the site-specific hydraulic conditions to bioreactor design is critical.

Because bioreactor beds (Figure 1) are positioned to intercept channelized flow or tile drainage, the extent of removal within these designs can be limited by hydraulic residence time and nitrate loading. Sizing bioreactor beds warrants careful understanding of void space volume within the space as well as the temporal and spatial variation of inputs. Design criteria for beds need to optimize costs vs. performance when considering seasonal variation and storm generated pulses of hydrologic input.

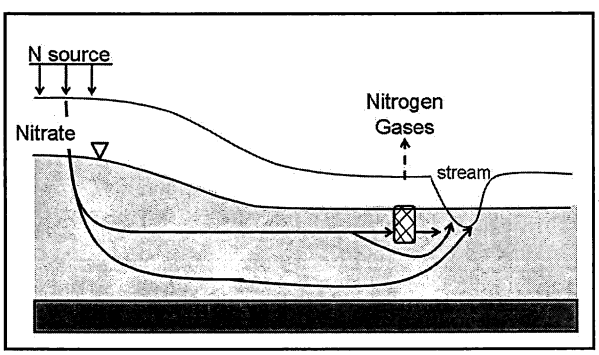

Figure 3. Schematic of denitrification wall in cross section where the ground water flow paths may bypass the wall where saturated hydraulic conductivity in the deep aquifer is higher than denitrification wall (from Schipper et al. 2010a).

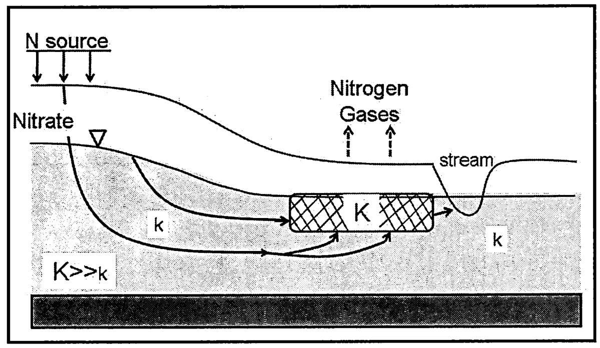

Figure 4. Cross-section of a denitrification wall where the wall's K is higher than the surrounding media resulting in upwelling of ground water into the wall (Schipper et al. 2010a).

Uncertainty and Variability

Seasonal and annual temperature differences between regions are likely to account for some of the variability observed in bioreactor performance. In general, biological reaction rates positively correlate with temperature. In the Schipper et al. (2010a) review of non-nitrate limiting bioreactors, a general trend of increasing denitrification rates with increasing average annual temperature was noted. However, some studies suggest that long-term N removal may be lowered in warmer climates since the C substrate may decompose more rapidly (Cameron and Schipper 2010).

Uncertainty also surrounds the extent of unintended environmental trade-offs. Bioreactors might produce other unwanted pollutants, such as N_2O and CH_4 (both potent greenhouse gases) from either incomplete denitrification or prolonged retention times that promote highly reducing conditions within the bioreactor. Controlling greenhouse gas emissions will likely depend on scaling bioreactors to ensure that the N concentrations within the bioreactor are poised to ensure complete denitrification while not allowing methanogenesis (Warneke et al. 2011). A further reason for optimizing size of denitrification bioreactors is management of the production of methyl mercury which can be produced under sulfate-reducing conditions (Shih et al. 2011). Leaching of dissolved labile C from plant material to water ways can affect dissolved oxygen concentrations of receiving waters. Careful start-up procedures may be able to minimize losses of dissolved C, such as pre-leaching of wood chips.

Constructed Wetlands for Offsite Nitrogen Removal

Overview

Constructed wetlands have long been used to treat wastewater, particularly sewage effluent, with extensive knowledge summarized on their design and operation (Kadlec and Knight 1996). With wastewater, inputs and retention times can be carefully controlled and optimized to provide maximum reductions in nitrate.

For nonpoint nitrate sources, such as tile drainage or surface runoff, tight control of constructed wetlands is generally not possible, and the literature is not as well summarized. For croplands, constructed wetlands are often small basins, typically 0.2 to 2 ha in size, created – often where natural wetlands were once located – to intercept nonpoint source runoff from agricultural fields. Osborne and Kovacic (1993) proposed using constructed wetlands when they observed ineffective nitrate removal in riparian buffer zones in a tile drained landscape since the tile lines short-circuited the buffer. Wetlands have been created to intercept surface runoff or tile drainage lines before entering ditches or streams. Berms are created, and the tile line broken so that the drainage enters the wetland. They are typically located alongside streams, with a buffer strip between the wetland berm and the stream. A few wetlands have been constructed in the middle of agricultural fields away from the stream, but these are not favored by landowners. Wetlands have also been created to receive surface runoff from agricultural fields (e.g., Jordan et al. 2003, Chesapeake Bay watershed), although these systems typically have much lower nitrate concentrations than tile drainage.

Performance

Several recent papers have summarized our knowledge of constructed wetlands when used to treat nonpoint nitrate inputs (Figure 5; Crumpton et al. 2008; Mitsch et al. 2005; O'Geen et al. 2010; U.S. EPA 2008). The major process that removes nitrate from wetlands is denitrification in shallow sediments following diffusion of the nitrate downward a few mm to cm. Denitrification in wetlands typically produces little N_2O (O'Geen et al. 2010; Xue et al. 1999), although there are limited measurements. Plant uptake is typically not an important long-term removal process (Hoagland et al. 2001). Most constructed wetlands receiving agricultural runoff have been built and evaluated in Iowa, Illinois, and Ohio, although there are studies of wetlands in many other areas of the U.S.

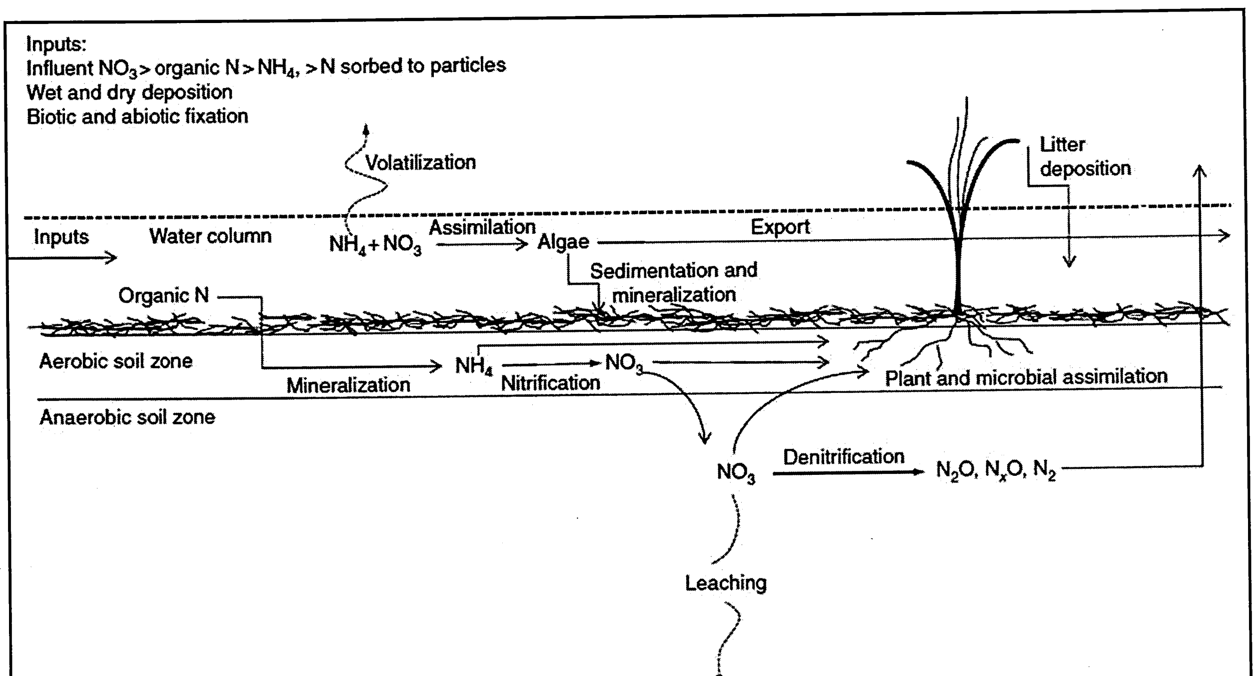

Figure 5. A general conceptual diagram of the N cycle within a constructed wetland (from O'Geen et al. 2010).

and around the world as well (primarily northern Europe and New Zealand) as well.

Only a few studies have made direct measurements of denitrification in wetlands receiving agricultural drainage, including Fleischer et al. (1994), Xue et al. (1999), Poe et al. (2003) and Hernandez and Mitsch (2007). Denitrification rates from these studies varied from 0.02 to 11.8 mg N m^{-2} hr^{1}, with an average rate of 2 mg N m^{-2} hr^{-1} reported by O'Geen et al. (2010) in their review.

Uncertainty and Variability

Much of the variation reported in denitrification rates across and within studies was due to seasonal temperature effects, with rates greatly reduced during winter and early spring periods, and maximized in mid-summer. In areas receiving extensive inputs during the colder periods of the year, the ability of constructed wetlands to greatly reduce nitrate loads is compromised (O'Geen et al. 2010). A careful compilation of all available data by season, rather than annual data, is needed to more carefully evaluate this component. There are many sources of variation, including hydraulic loading, temperature, nitrate concentration, soil C availability to drive denitrification, and vegetation (O'Geen et al. 2010).

More information is needed regarding the long-term performance of a constructed wetland in removing N via denitrification – as all studies that we are aware of now have evaluated wetlands during the first few years of construction. Studies are critically needed to evaluate wetlands many years after construction. In addition, long-term studies can provide insight into the potential for these artificial wetlands to release organic N, potentially making them a net source of N when there is little input of N (O'Geen et al. 2010). We suspect that in wetlands receiving high N loads dominated by nitrate, N will be transformed to N gas rather than stored within the plant and microbial biomass, minimizing release of organic N.

The amount of nitrate that is removed through denitrification is also a function of the hydraulic load and the surface area of the wetland. The longer nitrate-laden water is in the wetland (residence time), the greater the nitrate removal (Kovacic et al. 2000). Crumpton et al. (2008) illustrated a strong response curve of percent mass nitrate-N removed to the hydraulic loading rate but also found it best to predict mass removal using both loading and nitrate concentration. These studies suggest that wetlands are best located in areas with the greatest nitrate concentrations, to maximize mass removal of N. A related factor is the appropriate watershed to wetland ratio. In some studies, this ratio is presented as wetland area to the effective tile drained area (Kovacic et al. 2000). But as pointed out by Crumpton et al. (2008), this ratio is often not known and does not reflect the landscape level removal.

The Role of Geospatial Tools for Siting Artificial Sinks

Geospatial analysis has been used to integrate data layers at various scales to allow for the spatial targeting and interpretation of BMP implementation. The efficacy of bioreactors and constructed wetlands is dependent on spatially and temporally variable factors such as physiography, aquifer properties, soils, precipitation, temperature, and drainage networks. At regional scales, geospatial analysis can therefore provide broad guidelines for bioreactor and constructed wetland BMP implementation, based on variations due to temperature differences or generalized physiographic characteristics. This type of regional analysis has been applied to demonstrate differences in the efficacy of riparian buffers for pollution abatement across nine physiographic provinces in the Chesapeake Bay watershed (Lowrance et al. 1997). In that analysis, the effectiveness, optimum design, and management of buffer systems was linked to differences in hydrologic connections associated with different physiographic provinces. Similarly, there would be benefits for regional geospatial analysis that provide broad interpretations for bioreactor and constructed wetland BMP implementation, based on variations due to temperature, seasonal runoff patterns, and farming practices. Physiographic provinces are also important considerations for these artificial sinks; for example, in karst terrain improperly constructed wetlands can increase the risk of sinkhole formation and associated ground water contamination.

Geospatial data is also useful for design and siting at more localized scales. County-level soil survey data and associated interpretations have a long history of guiding conservation practices. Many phosphorus indices draw upon soil survey data to estimate such site-specific properties as soil erodibility and runoff. Design and placement of artificial sinks would benefit from soil survey data on depth to restrictive layers, seasonal water tables elevations, and hydraulic conductivity. The USGS real time stream gauging network affords an extensive spatially explicit dataset on area-normalized flow that has been used to provide insight into expected magnitudes and seasonality of flow (Armstrong et al. 2004). County-scale geospatial data are often best used as a guide for site-specific investigations with users made aware of the appropriateness and resolution limitations of the geospatial data sets. The fine resolution data necessary for direct site-specific analyses and design of specific artificial sinks will most likely be derived from either remotely sensed or in situ data, such as in precision agriculture applications.

Given our understanding of artificial sinks, such as constructed wetlands and bioreactors, to optimize nitrate removal they should be located in areas with the greatest nitrate loads (Crumpton et al. 2008). David et al. (2010) showed locations within the Mississippi River basin where artificial sinks (wetlands) would be strongly beneficial, based on the co-location of tile drainage combined with intensive agriculture in producing high winter and spring nitrate loads (Figure 6). There is a need to refine this information to be tailored for use at the state and county scale. This refinement will require the development of place-based information that includes site constraints based on soils, hydrology, and climate for suitability and optimal performance.

Opportunities and Challenges for Implementation

To advance the adoption and strategic placement of appropriate bioreactor and constructed wetland designs to remove offsite losses of N from agricultural lands, there is a need to increase the capacity of public, private, and university sectors to incorporate

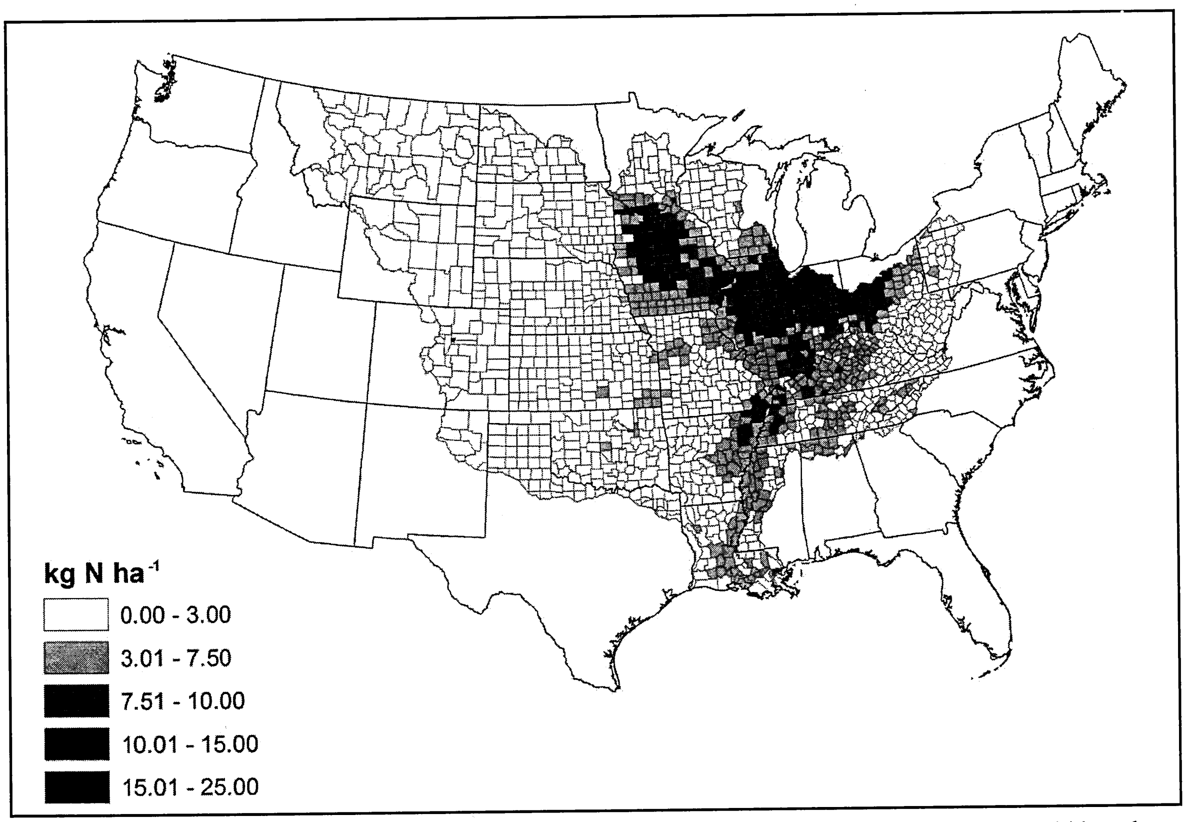

Figure 6. Estimated January to June nitrate N yield (from David et al. 2010). The black areas would be where tile-fed wetlands would be best located for maximum nitrate load reductions.

placed-based factors into the siting, design, and construction of these BMPs. While there is a strong research base available on these systems, experts (i.e., researchers, private sector, and Extension) need to come together to synthesize this information and generate region-specific guidance documents that Natural Resources Conservation Service, state agencies, and the private sector can use. A variety of groups across the U.S. (e.g., University of Illinois Extension and the Iowa Soybean Association) are already promoting the use of artificial sinks for N control through the establishment of demonstration sites. To bring these demonstration sites and research sites visibility and make them accessible to multiple audiences, we are exploring the creation of a place-based National Atlas of artificial N sinks through the use of Google MapsTM Mashup, similar to the Low Impact Development (LID) National Atlas (Dickson et al. 2011) that locates and showcases stormwater treatment practices. By sharing this information in such a widely-used format, local awareness of the practicalities of artificial N sinks for a given type of setting will be improved. Additionally, the synthesis of research data and demonstration sites should be incorporated into course curricula, perhaps including online training modules or podcasts by experts, to enhance student knowledge and technical literacy of the mechanisms, assessment techniques, design factors, and challenges presented by bioreactors and constructed wetlands for N removal from agricultural lands.

Conclusions

Where natural sinks are absent, carbon bioreactors and constructed wetlands hold great promise for reducing edge-of-field N losses. Current research needs to be further synthesized into guidance and training materials. Further research is needed to elucidate the types of designs best suited for different locales based on climate, seasonality, and physiography. Geospatial tools will be helpful in guiding the placement of these systems on regional and local scales. To facilitate widespread adoption of these artificial N sinks, guidance documents, training materials, demonstration sites, and web-based resources will help to increase local awareness of the practicalities of these BMPs for given settings.

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References

Addy, K. and A.J. Gold. 2008. *Groundwater nitrogen removal estimates for nitrogen barriers: Green Hill Pond, RI.* Project Report for Town of South Kingstown, RI.

Alexander, R.B., R.A. Smith, G.E. Schwarz, E.W. Boyer, J.V. Nolan, and J.W. Brakebill. 2008. Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River Basin. *Environmental Science and Technology* 42: 822-831.

Armstrong, D.S., G.W. Parker, and T.A. Richards. 2004. Evaluation of streamflow requirements for habitat protection by comparison to streamflow characteristics at index streamflow-gaging stations in Southern New England. *Water-Resources Investigations Report 03-4332.* USGS. Reston, VA. 1-101.

Cameron, S.G. and L.A. Schipper. 2010. Nitrate removal and hydraulic performance of carbon substrates for potential use in denitrification beds. *Ecological Engineering* 36: 1588-1595.

Cameron, S.G. and L.A. Schipper. 2011. Evaluation of passive solar heating and alternative flow regimes on nitrate removal in denitrification beds. *Ecological Engineering* 37: 1195-1204.

Cassman, K.G., A. Dobermann, and D.T. Walters. 2002. Agroecosystems, nitrogen-use efficiency, and nitrogen management. *Ambio* 31: 132-140.

Conley, D.J, H.W. Paerl, R.W. Howarth, D.F. Boesch, S.P. Seitzinger, K.E. Havens, C. Lancelot, and G.E. Liken. 2009. Controlling eutrophication: Nitrogen and phosphorus. *Science* 323: 1014-1015.

Crumpton, W.G., D.A. Kovacic, D.L. Hey, and J.A. Kostel. 2008. Potential of restored and constructed wetlands to reduce nutrient export from agricultural watershed in the corn belt. In *Upper Mississippi River Sub-basin Hypoxia Nutrient Committee. Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop.* American Society of Agricultural and Biological Engineers, St. Joseph, MI. 29-42.

David, M.B., L.E. Drinkwater, and G.F. McIsaac. 2010. Sources of nitrate yields in the Mississippi River basin. *Journal of Environmental Quality* 39: 1657-1667.

Dickson, D.W., C.B. Chadwick, and C.L. Arnold. 2011. National LID Atlas: A collaborative online database of innovative stormwater management practices. *Marine Technology Society Journal* 45: 59-64.

Fleischer, S., A. Gustafson, J. Joelsson, J. Pansar, and L. Stibe. 1994. Nitrogen removal in created ponds. *Ambio* 23: 349-357.

Gentry, L.E., M.B. David, K.M. Smith, and D.A. Kovacic. 1998. Nitrogen cycling and tile drainage nitrate loss in a corn/soybean watershed. *Agriculture, Ecosystems and Environment* 68: 85-97.

Gibert, O., S. Pomierny, I. Rowe, and R.M. Kalin. 2008. Selection of organic substrates as potential reactive materials for use in a denitrification permeable reactive barrier (PRB). *Bioresource Technology* 99: 7587-7596.

Goolsby, D.A., W.A. Battaglin, G.B. Lawrence, R.S. Artz, B.T. Aulenbach, R.P. Hooper, D.R. Keeney, and G.J. Stensland. 1999. *Flux and sources of nutrients in the Mississippi-Atchafalaya River Basin: Topic 3 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico.* National Oceanic and Atmospheric Administration, Silver Spring, MD. NOAA Coastal Ocean Program Decision Analysis Series no. 17.

Greenan, C.M., T.B. Moorman, T.C. Kaspar, T.B. Parkin, and D.B. Jaynes. 2006. Comparing carbon substrates for denitrification of subsurface drainage water. *Journal of Environmental Quality* 35: 824-829.

Groffman, P.M., K. Butterbach-Bahl, R.W. Fulweiler, A.J. Gold, J.L. Morse, E.K. Stander, C. Tague, C. Tonitto, and P. Vidon. 2009. Challenges to incorporating spatially and temporally explicit phenomena (hotspots and hot moments) in denitrification models. *Biogeochemistry* 93: 49-77.

Hernandez, M.E. and W.J. Mitsch. 2007. Denitrification potential and organic matter as affected by vegetation community, wetland age, and plant introduction in created wetlands. *Journal of Environmental Quality* 36: 333-342.

Hoagland, C.R., L.E. Gentry, M.B. David, and D.A. Kovacic. 2001. Plant nutrient uptake and biomass accumulation in a constructed wetland. *Journal of Freshwater Ecology* 16: 527-540.

Hoos, A.B. and G. McMahon. 2009. Spatial analysis of instream nitrogen loads and factors controlling nitrogen delivery to streams in the southeastern United States using spatially referenced regression on watershed attributes (SPARROW) and regional classification frameworks. *Hydrologic Processes* 23: 2275-2294.

Howarth, R.W., D.M. Anderson, T.M. Church, H. Greening, C.S. Hopkinson, W.C. Huber, N. Marcus, R.J. Naiman, K. Segerson, A. Sharpley, and W.J. Wiseman. 2000. *Clean coastal waters: Understanding and Reducing The Effects of Nutrient Pollution.* National Academy Press, Washington, D.C.

Jaynes, D.B., T.C. Kaspar, T.B. Moorman, and T.B. Parkin. 2008. *In situ* bioreactors and deep drain-pipe installation to reduce nitrate losses in artificially drained fields. *Journal of Environmental Quality* 37: 429-436.

Jordan, T.E., D.F. Whigham, K.H. Hofmockel, and M.A. Pittek. 2003. Nutrient and sediment removal by a restored wetland receiving agricultural runoff. *Journal of Environmental Quality* 32: 1534-1547.

Kadlec, R.H. and R.L. Knight. 1996. *Treatment Wetlands.* CRC Press, Boca Raton, FL.

Kovacic, D.A., M.B. David, L.E. Gentry, K.M. Starks, and R.A. Cooke. 2000. Effectiveness of constructed wetlands in reducing nitrogen and phosphorus export from agricultural tile drainage. *Journal of Environmental Quality* 29: 1262-1274.

Leverenz, H.L., K. Haunschild, G. Hopes, G. Tchobanoglous, and J. Darby. 2010. Anoxic treatment wetlands for denitrification. *Ecological Engineering* 36: 1544-1551.

Long, L., L.A. Schipper, and D.A. Bruesewitz. 2011. Long-term nitrate removal in a denitrification wall. *Agricultural Ecosystems and Environment* 140: 514-520.

Lowrance, R., L.S. Altier, J.D. Newbold, R.R. Schnabel, P.M. Groffman, J.M. Denver, D.L. Correll, J.W. Gilliam, J.L. Robinson, R.B. Brinsfield, K.W. Staver, W. Lucas, and A.H. Todd. 1997. Water quality functions of riparian forest buffers in Chesapeake Bay Watersheds. *Environmental Management* 21: 687-712.

Mitsch, W.J., J.W. Day Jr., L. Zhang, and R.R. Lane. 2005. Nitrate-nitrogen retention in wetlands in the Mississippi River Basin. *Ecological Engineering* 24: 267-278.

Moorman, T.B., T.B. Parkin, T.C. Kaspar, and D.B. Jaynes. 2010. Denitrification activity and wood loss over nine years of a wood chip bioreactor for treatment of nitrate in drainage water. *Ecological Engineering* 36: 1567-1575.

National Research Council. 1993. *Managing Wastewater in Coastal Urban Areas.* National Research Council, Washington, D.C., USA.

Nolan, B.T. 2001. Relating nitrogen sources and aquifer susceptibility to nitrate in shallow ground waters of the United States. *Groundwater* 39: 290-299.

O'Geen, A.T., R. Budd, J. Gan, J.J. Maynard, S.J. Parikh, and R.A. Dahlgren. 2010. Mitigating nonpoint source pollution in agriculture with constructed and restored wetlands. *Advances in Agronomy* 108: 1-76.

Osborne, L.L. and D.A. Kovacic. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology* 29: 243-258.

Poe, A.C., M.F. Piehler, S.P. Thompson, and H.W. Paerl. 2003. Denitrification in a constructed wetland receiving agricultural runoff. *Wetlands* 23: 817-826.

Preston, S.D., R.B. Alexander, M.D. Woodside, and P.A. Hamilton. 2009. SPARROW MODELING—Enhancing Understanding of the Nation's Water Quality. *U.S. Geological Survey Fact Sheet* 2009-3019, 6.

Preston, S.D. and J.W. Brakebill. 1999. *Application of Spatially Referenced Regression Modeling for the Evaluation of Total Nitrogen Loading in the Chesapeake Bay Watershed.* USGS Report WRIR 99-4054, Baltimore MD.

Robertson, W.D. and L.C. Merkley. 2009. In-stream bioreactor for agricultural nitrate treatment. *Journal of Environmental Quality* 38: 230-237.

Robertson, W.D., J.L. Vogan, and P.S. Lombardo. 2008. Nitrate removal rates in a 15-year old permeable reactive barrier treating septic system nitrate. *Ground Water Monitoring and Remediation* 28: 65-72.

Robertson, W.D., N. Yeung, P.W. van Driel, and P.S. Lombardo. 2005. High-permieability layers for remediation of ground water: Go wide, not deep. *Ground Water* 43: 574-581.

Rockstrom, J., W. Steffen, K. Noone, A. Persson, F.S. Chapin III, E.F. Lambin, T.M. Lenton, M. Scheffer, C. Folke, H.J. Schellnhuber, C.A. de Wit, T. Hughes, S. van der Leeuw, H. Rodhe, S. Sorlin, P.K. Snyder, R. Costanza, U. Svedin, M. Falkenmark, L. Karlberg, R.W. Corell, V.J. Fabry, J. Hansen, B. Walker, D. Liverman, K. Richardson, P. Crutzen, and J.A. Foley. 2009. A safe operating space for humanity. *Nature* 461: 472-475.

Schipper, L.A., G.F. Barkle, and M. Vojvodic-Vukovic. 2005. Maximum rates of nitrate removal in a denitrification wall. *Journal of Environmental Quality* 34: 1270-1276.

Schipper, L.A., G.F. Barkle, M. Vojvodic-Vukovic, J.C. Hadfield, and C.P. Burgess. 2004. Hydraulic constraints on the performance of a groundwater denitrification wall for nitrate removal from shallow groundwater. *Journal of Contaminant Hydrology* 69: 263-279.

Schipper, L.A., W.D. Robertson, A.J. Gold, D.B. Jaynes, and S.G. Cameron. 2010a. Denitrifying bioreactors – an approach for reducing nitrate loads to receiving waters. *Ecological Engineering* 36: 1532-1543.

Schipper, L.A., S.G. Cameron, and S. Warneke. 2010b. Nitrate removal from three different effluents using large-scale denitrification beds. *Ecological Engineering* 36: 1552-1557.

Seitzinger, S., J.A. Harrison, J.K. Bohlke, A.F. Bouwman, R. Lowrance, B. Peterson, C. Tobias, and G. Van Drecht. 2006. Denitrification across landscapes and waterscapes: A synthesis. *Ecological Applications* 16: 2064-2090.

Shih, R., W.D. Robertson, S.L. Schiff, and D.L. Rudolph. 2011. Nitrate controls methy mercury production in a streambed bioreactor. *Journal of Environmental Quality* 40: 1586-1592.

Tanner, C.C., M.L. Nguyen, and J.P.S. Sukias. 2005. Nutrient removal by a constructed wetland treating subsurface drainage from grazed dairy pasture. *Agriculture, Ecosystems and Environment* 105: 145-162.

United Nations Population Division. 2007. *World Population Prospects: The 2006 Revision*, Executive Summary.

U.S. EPA, U.S. Environmental Protection Agency. 2008. Hypoxia in the northern Gulf of Mexico: An update by the EPA Science Advisory Board. EPA-SAB-08-004. U.S. EPA, Washington, DC.

Warneke, S., L.A. Schipper, D.A. Brusewitz, I.R. McDonald, and S.G. Cameron. 2011. Rates, controls and potential adverse effects of nitrate removal in a denitrification bed. *Ecological Engineering* 37: 511-522.

Woli, K.P., M.B. David, R.A.Cooke, G.F. McIsaac, and C.A. Mitchell. 2010. Nitrogen balance in and export from agricultural fields associated with controlled drainage and denitrifying bioreactors. *Ecological Engineering* 36: 1558-1566.

Xue, Y., D.A. Kovacic, M.B. David, L.E. Gentry, R.L. Mulvaney, and C.W. Lindau. 1999. *In situ* measurements of denitrification in constructed wetlands. *Journal of Environmental Quality* 28: 263-269.