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Managing denitrification in human-dominated landscapes

1. The N problem

Increases in food supply and fossil fuel consumption are among the hallmarks of the 20th century. These changes share a common characteristic – they both contribute to an excess supply of plant-available (i.e., reactive) nitrogen – with negative consequences to ecosystems and water supplies across the globe. While there are vast quantities of di-nitrogen gas (N_2) in the atmosphere, this form of N is unavailable (termed unreactive N) to the vast majority of biological life (Galloway et al., 2003). Globally, this unreactive N can be converted to reactive N by four major processes: N-fixing microorganisms (often in symbiotic association with plants, 140 Tg N year^{-1}), industrial fertilizer production (125 Tg N year^{-1}), fossil fuel combustion (25 Tg N year^{-1}), and lightning (5 Tg N year^{-1}) (Schlesinger, 2009). The benefits of increased food production by use of N inputs are clear. World economies also continue to rely on fossil fuels for transport and fertilizer production. As with many biogeochemical processes that are manipulated at global scales, increased N inputs has adverse and unintended consequences (Galloway et al., 2008).

Direct losses originate from N applied to land that is in excess of plant and animal requirements or from deposition and runoff from urban lands, while indirect losses result from increasing volumes of animal and human wastes derived from harvested food. Our food production systems are not fully efficient in utilizing added N and there is frequently N in excess of plant demand. The excess N can be sequestered into soil organic matter or denitrified back to unreactive nitrogen gas (N_2), but the remainder is lost from the soil/plant system through product removal, leaching, erosion, and volatilization of ammonia. This leaching of N to groundwater, into surface water and eventually to the oceans creates a number of undesired effects with N functioning as unwanted fertilizer in nutrient-poor ecosystems causing increases in aquatic growth in streams, lakes, estuaries, and the ocean. Groundwater nitrate (NO_3^-) concentrations exceed drinking water standards in many parts of the world (Rupert, 2008). On passage from land to the ocean, nitrous oxide (N_2O) – a potent greenhouse gas – can also be produced. Production of reactive N gases (NH_3, NO_x) from fossil fuel production and agricultural land and subsequent deposition can also increase N inputs to nutrient-poor ecosystems altering plant growth and biodiversity. These multiple effects of excess N moving through ecosystems have been termed the N cascade (Galloway et al., 2003).

The importance of managing the N cycle and cascading effects of excess N has been recognized as one of the 14 Grand Challenges for Engineering in the 21st Century (http://www.engineeringchallenges.org). Excess N is also one of three global planetary boundaries that have been exceeded (Rockstrom et al., 2009). Rockstrom et al. (2009) suggested a planetary boundary of 35 Tg reactive N input per year from anthropogenic activities while current inputs are three to four times higher. Galloway et al. (2008) estimated that a range of practices including managing fossil fuel production of reactive N, improved plant uptake efficiency, and animal management practices coupled with better wastewater treatment systems could reduce reactive N inputs to the environment by about 53 Tg N. This still leaves a substantial gap between current inputs and the proposed planetary boundary that might be addressed through management of denitrification in the environment.

Clearly, matching the inputs of N to plant demands is the most important approach for reducing N losses to watersheds and aquatic resources. But this goal is unlikely to be achieved with 100% efficiency, which leaves "managed" denitrification as an additional viable mitigation tool to reduce unwanted reactive N in the environment. In some ways, denitrification is analogous to the role of C sequestration in mitigating increasing carbon dioxide concentrations in the atmosphere. A difference from the analogy to C sequestration is the importance of spatial separation between where N inputs are made and where this N is converted back to N_2 gas through denitrification. While the N cycle is eventually completed in the oceans via denitrification, before reactive N has reached the ocean it has caused a wide range of the adverse effects on aquatic and terrestrial ecosystems. Consequently, it is important to remove reactive N close to the site of N input either by maximizing plant uptake and removal or by enhancing denitrification. However, many anthropogenic practices have spatially decoupled the sites of N inputs from sites of active denitrification, such as the drainage of land for food production and large losses of wetlands and riparian buffers adjacent to agricultural land.

2. The potential for managing denitrification

In May 2009, a workshop was held at the University of Rhode Island under the auspices of the National Science Foundation's Denitrification Research Coordination Network (RCN), which was established to enhance collaboration between researchers investigating denitrification. Full details of Denitrification RCN can be found at: http://www.denitrification.org. This was the fourth workshop organized by the RCN and this workshop attempted to identify ways in which denitrification could be managed to reduce N movement along the N cascade. Our focus was mainly

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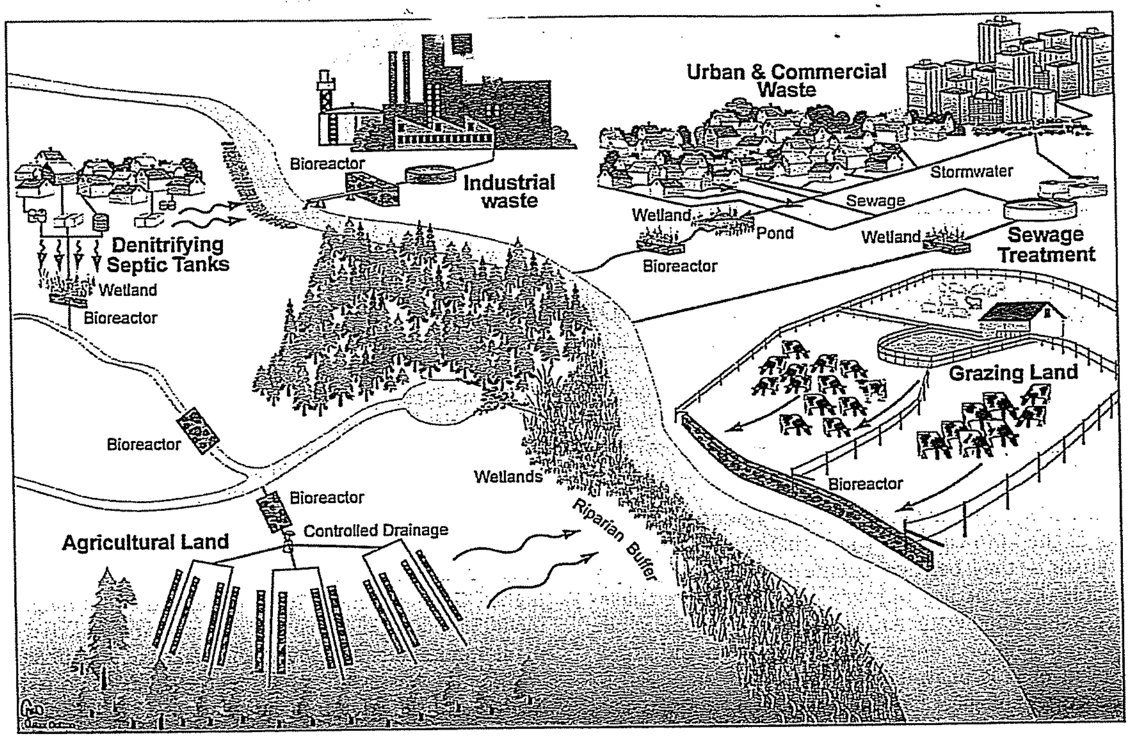

Fig. 1. Schematic of potential sites and approaches for enhancing denitrification to reduce N movement along the N cascade. Many of these approaches are addressed in this Special Issue. Note that there are multiple points along the N cascade where denitrifying bioreactors can be implemented.

on managing diffuse sources (non-point) of N contamination – e.g., from croplands, farms, wastewater from unsewered developments and small industrial systems, and urban runoff. We considered tools and techniques that could complement other management approaches such as denitrification systems for municipal wastewater treatment and nutrient management systems for agriculture. In contrast to point sources such as municipal wastewater treatment plants, the widespread nature of diffuse sources requires low-cost, treatment approaches with less intensive management (although not zero-maintenance). Ecological engineering approaches that rely on self-regulating systems can provide an important foundation for addressing excess N losses at the farm, residential, and small catchment scales associated with diffuse pollution. The workshop included biophysical scientists, engineers, extension personnel, and social scientists because it was recognized that data generated in research projects require transitioning from experimentation to practical implementation. Attendees also crossed a variety of ecosystems and disciplines, ranging from those who work in sewage treatment systems (large and small scale), urban, rural, riparian, aquatic, and wetland environments.

This Special Issue of Ecological Engineering is an output from this workshop, comprising three synthesis papers developed by groups that formed at breakout sessions at the workshop. A number of contributed papers in line with the workshop theme also form part of this Special Issue. The common aim of these papers is to examine strategies to minimize N movement through the cascade often by enhancing denitrification. There are multiple points along the N cascade where intervention is possible (Fig. 1) and papers in this Special Issue have focused on two broad areas; approaches to enhance denitrification in wastewaters (e.g., septic tank discharges and stormwater) and in water derived from agricultural practices (e.g., in groundwater, tile flow, and in streams).

3. Dealing with concentrated discharges

Collins et al. (2010) review the issue of N discharges in urban storm water systems including approaches to manage losses and a survey of the personnel responsible for storm water management. Perhaps surprisingly, N in storm water was not ranked highly as a pollutant by respondents and consequently most of the approaches for storm water treatment utilize structures (e.g., wet and dry ponds) focus on treating other contaminants like suspended solids and pathogens. Collins et al. suggest that educational efforts are needed that document the effects of urban sources of N on aquatic ecosystems to motivate storm water managers to reduce N loading. The authors cite a number of opportunities to promote denitrification within the suite of existing storm water control practices. These include enhancing retention times, increasing the use of vegetation and treatment wetlands, adding denitrifying bioreactors to the treatment train and creating anaerobic, carbon rich zones within rain-gardens and bioretention filters. In addition, they argue for a catchment scale treatment approach where developers and regulators are encouraged to protect and restore natural denitrification sinks downstream to augment onsite controls.

Perhaps better recognized is the issue of N losses from decentralized wastewater treatment. In contrast to the performance of centralized wastewater treatment systems, there has been little comparative research on the wide array of N removal designs for onsite wastewater treatment systems (OWTs). The synthesis paper by Oakley et al. (2010) analyzed data from three major studies that tested the comparative performance of different N removal OWT systems. The types of OWTs differed in their approach to enhancing denitrification for N removal. Oakley et al. demonstrated that most OWT designs cannot meet the rigorous performance targets set by local policy directives; however, most of the systems could achieve at least a 50% reduction in annual N loading. Limitations for N removal by many of the OWTs designs occurred because OWTs were designed as downscaled versions of large wastewater treatment systems but without the same control of effluent loads, volumes, and processes. The small-footprint OWTs do have benefits for retrofitting small, shoreline lots and afford options for sustaining compact village developments. The only OWT system that consistently met policy targets was a simple sand filter coupled to a denitrifying bioreactor. The findings of Oakley et al. suggest that further thought should be given to the integration of ecological engineering principles into the design of new OWTS that require less management and can adapt to fluctuating flows and concentrations of effluent from dwellings.

Both the synthesis papers of Kelley et al. (2010) and Oakley et al. (2010) recognize the potential development and implementation of denitrifying bioreactors as an approach for removing nitrate from water. Schipper et al. (2010b) provide a synthesis of the current understanding of denitrifying bioreactors used for removing nitrate from wastewaters, groundwater, and discharges from tile drains and ditches. The general approach is to add particulate carbon sources (often fragmented wood products) to provide a slowly degrading carbon source to denitrifying organisms. This synthesis draws upon the literature that addresses the functioning of these systems within different hydrological settings and summarizes nitrate removal rates, longevity, N removal processes and potential adverse effects. There are many areas that remain poorly understood and will benefit from future research and the development of design manuals targeted at both engineers and land managers.

Many of the contributed papers in this Special Issue investigate the use of denitrifying bioreactors to treat different municipal and industrial waste streams (Leverenz et al., 2010; Schipper et al., 2010a), tile drainage systems (Woli et al., 2010; Moorman et al., 2010), streams (Elgood et al., 2010), and shallow groundwater (Robertson et al., 2010).

To treat wastewaters, Leverenz et al. (2010) integrated the concept of a denitrification bioreactor with the more established practice of constructed wetlands. Wood chips replaced the more expensive gravel (normally used for constructed wetlands) to provide the additional advantage of supporting denitrification. Plants were established, providing aesthetic benefits without compromising treatment in the medium term. Clearly, the wood chips will degrade and need to be replaced and the frequency of replacement in this study was not known. However, Robertson (2010) demonstrated that after 7 years the nitrate removal rates were still within 75% of those measured after 2 years. This longevity was supported by Moorman et al. (2010), who demonstrated sustained nitrate removal in bioreactors installed on either side of a tile drain for more than 9 years. These authors also showed that wood chips that were permanently water-saturated had a half life of 36.6 years as compared to 4.6 years for wood chips that were only periodically saturated.

Nitrate removal supported by different carbon sources was tested by Cameron and Schipper (2010). Nitrate removal rates were greater with maize cobs and garden waste than wood chip material of a range of sizes, but these substrates will likely degrade more quickly and may need more frequent replacement. Finding carbon sources that support nitrate removal rates greater than wood chips is important to allow construction of bioreactors with small footprints for retrofitting (Oakley et al., 2010). But denitrifying bioreactors do not necessarily need to be small. Schipper et al. (2010a) demonstrated sustained nitrate removal from three different waste waters with high flows and nitrate concentrations. One of the bioreactors studied was approximately 200 m long and contained 1320 m^3 of wood chip material. It may be possible to incorporate denitrifying bioreactors into centralized treatment systems as a final polishing step where large amounts of wood material are available. However, a range of studies of denitrifying bioreactors to date have clearly demonstrated that they only remove nitrate from waste water and not other forms of N compounds (e.g., ammonium, organic N) or other contaminants such as pathogens, phosphorus, and biological oxygen demand (Schipper et al., 2010b).

4. Dealing with agricultural tile- drained systems

Denitrifying bioreactors have also been integrated into agroecosystems to treat water with excess NO_3^- including streams and shallow groundwater moving into tile drains. Moorman et al. (2010) present results from a 9-year study of bioreactors installed on either side of a tile drain under corn soy-bean rotation. They measured average NO_3^- losses in tile drains surrounded by denitrifying bioreactors of 24.5 kg N ha^{-1} $year^{-1}$, which was substantially less than the losses from control plots of 54.5 kg N ha^{-1} $year^{-1}$. Similarly, Woli et al. (2010) studied bioreactors installed at the end of tile drains, and these removed 17 kg N ha^{-1} $year^{-1}$. In this study, the bioreactors were integrated with controlled drainage as a dual approach for nitrate removal. Controlled drainage, unlike bioreactors, is thought to increase denitrification – not by adding a carbon source, but by holding back the groundwater under fields to enhance the time of anoxic conditions and denitrification. Nitrate losses from free drainage plots were 57.2 kg N ha^{-1} $year^{-1}$ whereas losses in controlled drainage were much less at 17 kg N ha^{-1} $year^{-1}$.

5. Potential downside of incorporating denitrification systems throughout the landscape?

While nitrate removal in bioreactors has been clearly demonstrated in previous studies and in this Special Issue, it is widely acknowledged that there may be adverse consequences to inappropriate use of bioreactors. Incomplete denitrification can lead to the production of N_2O and decomposition of wood material might also lead to leaching out of degradable organic matter into receiving waters which can increase consumption of oxygen and anoxic waters. Three of the studies in this Special Issue demonstrated that denitrification in bioreactors appeared to be reasonably complete with less than 0.6% of removed nitrate being converted to N_2O (Moorman et al., 2010; Elgood et al., 2010; Woli et al., 2010). The production of N_2O needs to be quantified for all strategies using denitrification along the N cascade. Cameron and Schipper (2010) measured concentrations of biological oxygen demand (BOD) and ammonium from a range of different carbon substrates that was greater than environmental guidelines for the first two months after setup. Practical approaches to deal with this initial flush of BOD and ammonium need to be developed. Similarly, Leverenz et al. (2010) also measured substantial declines in BOD exiting a denitrification bioreactor during the first two months of operation.

6. Integrating into the bigger picture

The papers in this Special Issue generally describe specific approaches to deal with nitrate moving through the cascade, but regulators of land are often faced with difficult decisions about which approaches to promote and where. What are the overall and comparative benefits of the different approaches? Will these approaches meet the outcomes desired by the community and required by the environmental constraints of receiving ecosystems? Regulators need to make decisions often with limited information about the NO_3^- removal capacity of natural ecosystems and how these capacities might be enhanced utilizing ecological engineering principles. Kellogg et al. (2010) attempt to make progress towards resolving these issues by developing a modeling approach that can target the risk of N delivery from different locations within a small catchment by focusing on N losses within natural N sinks. Thus, local planners, regulators, and resource protection agencies can make informed decisions on where to prioritize the use of source controls and ecological engineering approaches, such as controlled drainage and denitrifying bioreactors that are described in this Special Issue. The model is based on combining widely available geospatial and stream flow data with empirical relationships of N removal within naturally occurring sinks, such as riparian wetlands, lakes, and headwater streams. These sorts of models are critical to improve the placement, selection, and efficiency of denitrification controls at critical source areas and to minimize the effects of the N cascade at the catchment scale.

7. Conclusions

The most important way of decreasing the impacts of N on receiving ecosystems is to better match N inputs to demand to avoid problems from arising. However, given the current N inefficiencies of agro-ecosystems, storm water controls, and waste treatment systems, the need exists to manage denitrification throughout catchments to complete the N cycle. The closer to the source of nitrogen the mitigation strategy is placed, the better. The workshop was successful at initiating cross-fertilization of ideas but much more progress needs to be made in integrating concepts for N removal developed in engineering and ecological realms. Some of these ideas have started to be explored in this Special Issue. It is also important to engage stakeholders in the need for reducing N movement through the cascade and provide them with tools and appropriate design manuals to implement solutions. Often, this is best achieved by involving stakeholders and regulators in the initial design and testing of mitigation strategies to ensure that approaches are practical and fit within policy frameworks.

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