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NITROGEN TRANSFORMATION AND FATE IN PRAIRIE WETLANDS

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Abstract. *Agricultural applications of fertilizers and pesticides have increased dramatically in the prairie pothole region since the middle 1960s, and agrochemical contamination of surface and groundwater has become a serious environmental concern. There is growing interest in the potential of prairie wetlands as sinks for excess nutrients in this agricultural landscape. As much as 50% of the fertilizer nitrogen applied to cultivated crops may be lost as nitrate in agricultural drainage water, and prairie wetlands may be especially effective as nitrate sinks. The effectiveness of prairie wetlands as sinks for nonpoint source nitrogen loads is likely to depend on the magnitude of nitrate loads and the capacity of the wetlands to remove nitrate by dissimilatory processes. Performance forecast models are needed to evaluate the effectiveness of prairie wetlands as nitrogen sinks from a watershed scale framework. This will be made significantly more difficult by the spatial and temporal complexity of prairie pothole wetlands and by their hydrologic diversity. Future research should focus on identifying the principal factors controlling nitrogen transformation in prairie wetlands and on developing general predictive tools for modeling nitrogen fate in these systems.*

There is probably no region in North America where understanding and managing agriculture's impacts is more critical than in the prairie pothole region. This region comprises the glaciated prairie, an area of some 300,000

square miles extending northward from central Iowa through western Minnesota and the eastern Dakotas into southern Manitoba, Saskatchewan, and Alberta (see Murkin 1998). The region is characterized by rich, poorly drained soils with numerous depressions of glacial origin forming small wetlands, the prairie potholes for which the region is named. These wetlands are extremely productive, generating between 50% and 75% of North America's waterfowl (Batt et al. 1989). Agriculture is the dominant economic force in the region, and since settlement there have been massive losses of wetlands as more and more were drained for cultivation. Conflict over wetland drainage remains a major issue in the prairie pothole region (Tiner 1984; Stuber 1988), and there is growing concern over the effects of chemical-intensive agriculture.

Agricultural applications of fertilizers and pesticides have more than doubled since the middle 1960s, and impacts of agrochemicals on water quality have come to pose obvious, pressing environmental problems (National Research Council 1986; Keeney 1986). Nitrate (NO_3^-) and pesticides have become the agrochemical contaminants of foremost concern because of their potential adverse impacts on both public health and ecosystem function, because of the widespread use of nitrogen and pesticides in modern agriculture, and because of the potential mobility of nitrate and some pesticides in surface and groundwaters. The total amount of N applied in fertilizers far exceeds that of any other nutrient. In the U.S., the annual application of fertilizer-N (FN) has grown from a negligible amount prior to World War II to approximately ten million metric tons of N per year (Hargett and Berry 1981; Berry 1993). The same trend is found in the states of the prairie pothole region and in the Prairie Provinces of Canada (Figure 1; Hargett and Berry 1981; Berry 1993; Korol and Girard 1996). The impacts of chemical intensive agriculture are a special concern in the prairie pothole region. Non-point loads of nitrogen to surface waters in this region are among the highest in the U.S. (Omernik 1977), and it is clear that many prairie wetlands receive significant agrochemical loads. As much as 50% of the fertilizer nitrogen applied to cultivated crops may be lost in agricultural drainage water, primarily in the form of nitrate (Neely and Baker 1989).

This paper reviews the dominant nitrogen transformations in prairie pothole wetlands, the potential of prairie wetlands as sinks for nonpoint source nitrogen loads, and the likely effects of increased agricultural nitrogen loads on wetland functions. Our goals are to summarize the current state of knowledge on nitrogen transformation and fate in prairie pothole wetlands and to suggest some directions for future research.

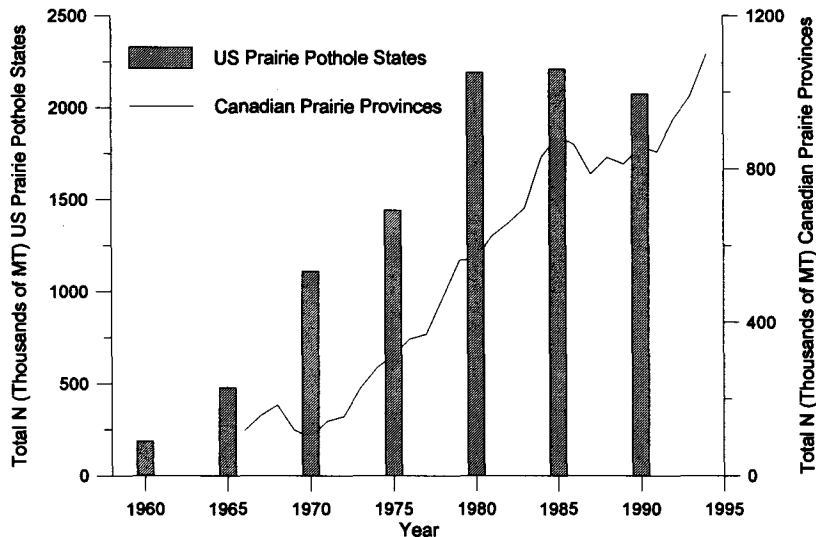


Figure 1. Fertilizer nitrogen use in the Canadian Prairie Provinces (Alberta, Manitoba, and Saskatchewan) and in the prairie pothole states (Iowa, Minnesota, South Dakota, and North Dakota).

Nitrogen Transformations in Wetlands

Nitrogen transformation processes in wetlands are comparable to those in other aquatic systems and in soils (Fig. 2; Kamp-Nielson and Anderson 1977; Painter 1977; Howard-Williams 1985; Bowden 1987; Reddy and Graetz 1988). However, the dynamics and relative importance of various processes are functions of the complex spatial and temporal patterns found in wetlands. Prairie pothole wetlands are complex mosaics of habitats with distinct structural and functional characteristics. Emergent macrophytes comprise the dominant vegetation of prairie potholes, although vegetation structure varies as a function of wetland depth and size. Many small potholes contain only one vegetation type, while larger potholes may have as many as five or six. Vegetation structure is also affected by the characteristic wet-dry cycles of prairie pothole wetlands, where periodic droughts alternate with wet years of above normal precipitation (van der Valk and Davis 1978a, 1978b). Interannual changes in the water levels in these wetlands over a

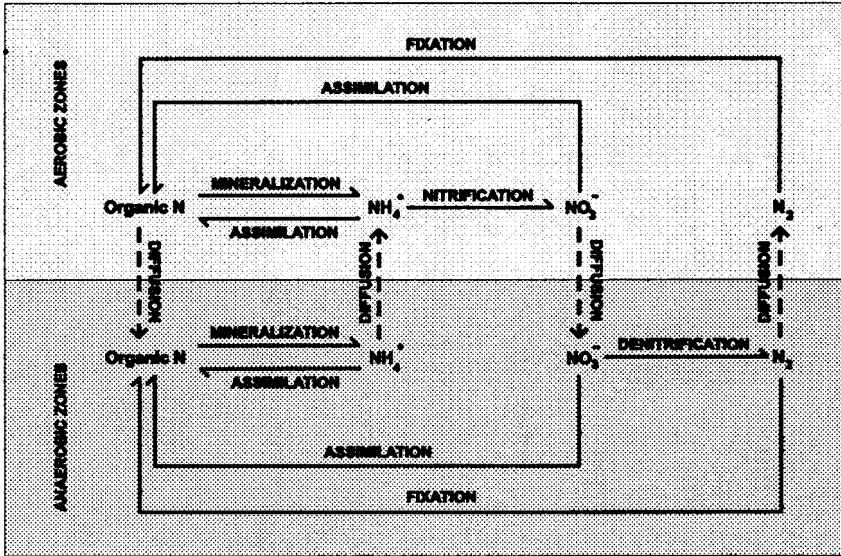


Figure 2. Generalized diagram of nitrogen transformations and compartments in a prairie wetland.

complete wet-dry cycle can exceed 2m in larger wetlands. During high water years, only submersed vegetation and a fringe of emergents and wet meadow species may be found. During droughts when many of these wetlands may go completely dry, seeds of annuals, emergents, and wet meadow species germinate on exposed mudflats, re-establishing the emergent and wet meadow zones eliminated during years of high water (van der Valk and Davis 1978a; Kantrud et al. 1989). In addition to serving as the principal source of organic carbon, wetland vegetation influences the physical and chemical characteristics of wetland sediments and overlying waters, including temperature and oxygen availability (Rose 1996; Rose and Crumpton 1996). Through a combination of direct and indirect effects, vegetation dynamics are likely to control many of the complex biogeochemical transformations involved in the nitrogen cycle of prairie pothole wetlands.

Ammonium can be produced in wetlands through either nitrogen fixation or through mineralization of organic nitrogen. Although there is evidence of nitrogen fixation in similar wetlands (Hooper-Reid and Robinson

1978), nitrogen fixation rates for prairie pothole wetlands are unknown and are likely to vary widely in response to changes in nitrogen availability from external sources or as a result of mineralization. Ammonium release during decomposition of organic material depends on the balance between gross mineralization and assimilation. Emergent macrophyte production in northern prairie wetlands provides large amounts of detrital carbon to decomposers (van der Valk and Davis 1978b; Shay and Shay 1986; Murkin 1989). The high C:N ratio of litter derived from emergent macrophytes in northern prairie wetlands (Neely and Davis 1985) requires microbes to acquire additional nitrogen by assimilation or nitrogen fixation in order to utilize the carbon (Bowden 1987). The net effect during decomposition of such material is to conserve or accumulate nitrogen while carbon is respired away (Bowden 1987; Neely and Baker 1989). However, depressed oxygen levels could limit aerobic decomposition rates in sediments and even in the water column of emergent zones (Rose 1996; Rose and Crumpton 1996). Alternatively, anaerobic decomposition pathways are less efficient than aerobic metabolism, resulting in lower bacterial growth yields and generally lower ratios of nutrient assimilation relative to mineralization. As a result, assimilation is a less effective nitrogen sink under anaerobic conditions and ammonium tends to accumulate, as is illustrated by sediment to water concentration gradients (Fig. 3; Robinson et al. in press). Sediments can serve as an enormous nitrogen pool in prairie wetlands from which the efflux is controlled by the balance between rates of mineralization in underlying anaerobic zones and the competing processes of assimilation and nitrification in the overlying aerobic zones (Fig. 2). Seasonal and even diurnal changes in factors such as temperature, nitrogen concentration gradients, and oxygen concentration can dramatically affect the balance among these processes (Andersen et al. 1984; Christensen et al. 1989; Crumpton et al. 1994; Isenhardt 1992; Phipps 1997).

Under aerobic conditions, ammonium can be converted to NO_3^- by nitrification. Both NH_4^+ and NO_3^- may be assimilated by aquatic organisms and immobilized to organic nitrogen, in the case of NO_3^- requiring assimilatory reduction to ammonium. In addition, NO_3^- may serve as a terminal electron acceptor for the oxidation of organic carbon in sediments having sufficiently low redox potential. This may involve either dissimilatory reduction of nitrate to ammonium, or more commonly denitrification, resulting in gaseous losses of N_2O or N_2 (Bowden 1987). In the absence of external nitrate loading, these dissimilatory processes would be limited by the rate of nitrification, and as a result, relatively low rates of denitrification

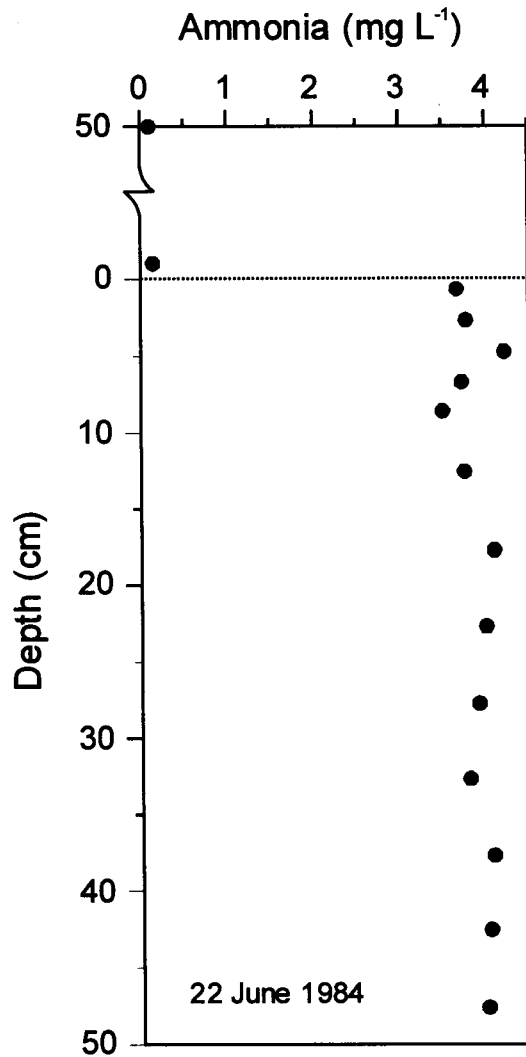


Figure 3. Soil to water concentration gradient of ammonia in a prairie wetland.

are expected in natural, unimpacted wetlands (Seitzinger 1988). Denitrification rates could be enhanced by close coupling of nitrification and denitrification reactions at aerobic-anaerobic interfaces such as at the sediment surface and in the rhizosphere (Reddy et al 1989). In addition, the frequency

and duration of water level fluctuations could significantly affect nitrification and denitrification rates, especially in seasonally flooded wetlands (Neill 1995). In essence, alternately dry and flooded conditions can provide a temporal coupling of aerobic and anaerobic conditions favoring nitrification and denitrification reactions respectively.

Prairie wetlands are rarely unimpacted systems, and many can be expected to receive significant external nitrate loading from surrounding agricultural lands. From a watershed perspective, one can think of this as a coupling of nitrification reactions in aerobic, upland soils and denitrification reactions in anaerobic, wetland soils. Under these conditions, relatively high rates of denitrification can be expected. ^{15}N tracer studies suggest that in excess of 80% of externally loaded nitrate could be lost through denitrification in prairie pothole wetlands receiving sustained nitrate loads (Isenhardt 1992; Moraghan 1993; Crumpton et al. 1994).

Prairie Wetlands as Nitrogen Sinks

Despite conspicuous accumulations of plant biomass and litter in prairie wetlands, and despite the obvious accumulation of organic carbon and nitrogen in wetland sediments, long term rates of organic nitrogen accrual in wetlands are relatively low, a few grams m^{-2} year $^{-1}$ (Johnston 1991; Weisner 1994). The effectiveness of prairie wetlands as sinks for nonpoint source nitrogen loads is likely to depend on the magnitude of nitrate loads and the capacity of the wetlands to remove nitrate by dissimilatory processes. Most of the published papers dealing with wetlands and water quality note the probable importance of denitrification, with resulting gaseous loss of N_2O and N_2 . In fact, with rare exception, denitrification is cited as the primary reason wetlands may serve as nitrogen sinks. It is thus all the more remarkable that so few measurements of denitrification in prairie wetlands have been published (Neely and Baker 1989; Isenhardt 1992; Moraghan 1993; Crumpton et al 1994; Isenhardt and Crumpton In preparation). As Neely and Baker (1989) note, denitrification is assumed to be an important process in northern prairie wetlands based largely on circumstantial evidence, first that conditions in prairie wetlands are suitable for denitrification (anaerobic conditions and a large base of organic carbon) and second that nitrate disappears rapidly from water overlying marsh sediments.

Although no thorough investigation of the mechanisms of nitrogen loss in prairie wetlands has been published, there is no compelling reason to doubt the significance of denitrification as a sink for external nitrate loads in

these systems. In fact, increased nitrate loading to prairie wetlands in agricultural watersheds can be expected to stimulate denitrification (Isenhardt 1992; Moraghan 1993; Crumpton et al. 1994; Phipps 1997). Both Isenhardt (1992) and Moraghan (1993) observed that exposure to nitrate had a stimulatory effect on nitrate loss in experimental wetland enclosures, apparently due to increased denitrification. In addition, despite a relatively low half-saturation constant, denitrification in these systems seems to be transport-limited and as a result can be described as a first order process over a wide range of concentrations (Isenhardt 1992; Crumpton et al. 1994; Kadlec and Knight 1996; Phipps 1997). This is illustrated by the linear relationship between nitrate loss and nitrate concentration in experimental wetland enclosures (Fig. 4). Increases in nitrate load and concentration in prairie pothole wetlands are expected to result in corresponding increases in dissimilatory losses of nitrate.

Effects of nitrogen loads on wetlands

In addition to the obvious effects on nitrogen cycling in wetlands, increased rates of dissimilatory nitrate reduction (to either NH_4^+ or N_2O and N_2) could have profound effects on carbon and energy flow in wetlands. Research on benthic anaerobic metabolism suggests clear distinctions between marine and freshwater systems, differences related primarily to the availability of SO_4^- as a terminal electron acceptor (Ward and Winfrey 1985; Capone and Kiene 1988). In the absence of oxygen, hydrolysis and depolymerization produce fermentable monomers. Fermentation end products are consumed primarily by sulfate reducing bacteria in marine systems and by methanogenic bacteria in freshwater systems (although sulfate reduction does occur in freshwater). Although nitrate is a considerably more favorable electron acceptor, nitrate concentrations in anoxic sediments are generally considered to be too low to support significant carbon metabolism in the absence of significant external loading of nitrate. However, in the presence of external nitrate loading, nitrate respiration can support high rates of metabolism, roughly equivalent to rates supported by aerobic respiration or sulfate reduction in marine systems (Jorgensen and Sorensen 1985).

It is unclear how increased rates of denitrification in response to nitrate loading would affect other anaerobic processes in freshwater wetlands. In slurry experiments, high nitrate concentrations of course favor denitrification over either sulfate reduction or methanogenesis, but this does not mean that denitrification will displace methanogenesis in intact wetland sedi-

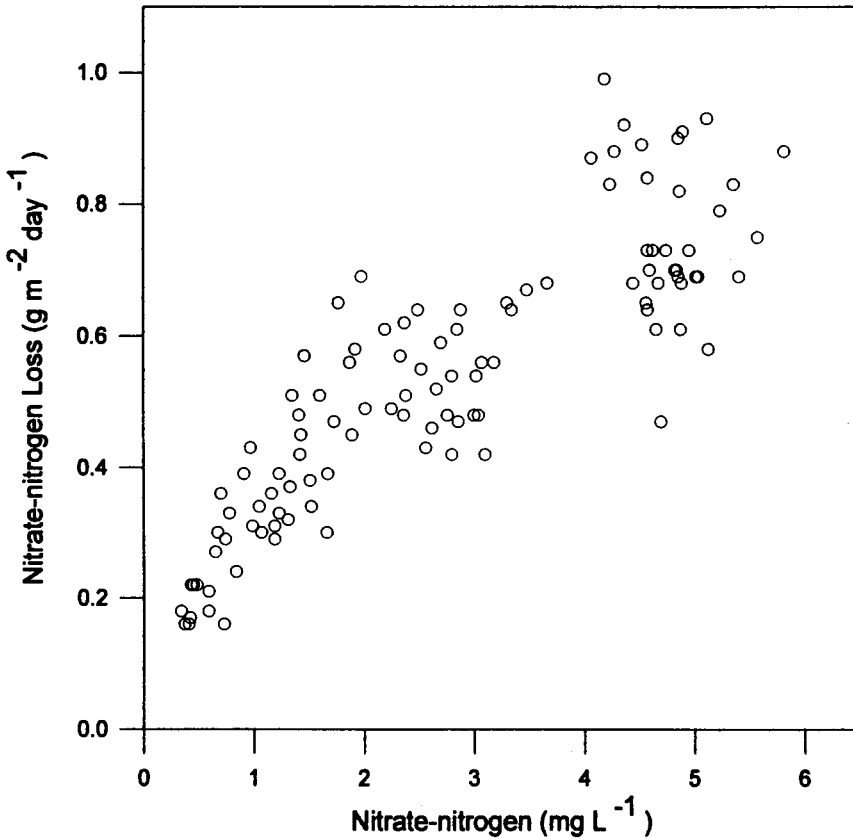


Figure 4. Nitrate-nitrogen loss rates from water overlying wetland sediments versus nitrate concentrations in field enclosures.

ments. As nitrate concentrations in overlying waters increase, denitrification rates increase, as does the depth to which nitrate penetrates, the denitrification "zone" (Christensen et al. 1989). However, in sediments with sufficient organic material, denitrification probably consumes the nitrate within a few mm of the sediment surface, as reported by Christensen et al. (1989) for streams. Methanogenic bacteria in the large volume of underlying, anaerobic sediment would see no nitrate and be largely unaffected. The major effect of external nitrate loads in freshwater wetlands may be increased overall

rates of anaerobic and total carbon metabolism. Agricultural nitrate loads could thus have profound effects on carbon and mineral cycling in prairie pothole wetlands. However, there is virtually no information concerning the effects of elevated denitrification rates on carbon flow through aerobic and various anaerobic pathways in these systems.

There is also relatively little information concerning the effects of agricultural N loads on primary production in prairie wetlands. Although there is considerable descriptive and experimental evidence of P limitation in freshwater systems (e.g., Hecky and Kilham 1988), fragmentary evidence suggests that some prairie wetlands are N limited and, therefore, that their primary production will increase with N enrichment. The few published measurements of the N:P ratio in the water column of prairie wetlands are typically below 25 (Goldsborough and Robinson 1996), and values less than the Redfield (1958) ratio of 16, assumed to indicate N deficiency, have been reported for shallow lakes or wetlands in Iowa (Carper and Bachmann 1984), Manitoba (Kadlec 1986), and Minnesota (Haertel 1976). Metabolic measurements of nutrient deficiency in wetland algae, based on their rate of ammonium uptake and the N:C ratio in their biomass, have shown that unproductive marshes of Manitoba's Interlake region are frequently N limited (Murkin et al. 1991) and nitrogenase activity, indicative of N₂ fixation, has been detected in benthic algae from Delta Marsh, Manitoba (Hooper-Reid and Robinson 1978). N limitation might be more common in systems receiving sustained phosphorous loads, and the development of cyanobacterial blooms in hypertrophic prairie potholes may in part relate to the ability of these organisms to thrive at low ambient N via fixation of atmospheric N₂ (Barica et al. 1980). In the case of macrophytes, a fertilization study in Delta Marsh demonstrated that above-ground biomass of two of the most common species, *Scolochloa festucacea* (whitetop grass) and *Typha glauca* (cattail) increased with N treatment but not with P treatment (Neill 1990). However, there is too little information on the frequency, severity, and spatial occurrence of N limitation in prairie wetlands to estimate the effects of increased N loads on primary production of either algae or macrophytes in these systems.

Research Needs

A better understanding of wetland water quality functions is especially critical in the prairie pothole region at the current time. In 1986, with waterfowl populations at their lowest levels in thirty years, the United States and Canada agreed to the North American Waterfowl Plan, which called for

the protection and restoration of 4.7 million acres of wetlands in the prairie pothole region (Kasten 1988). The National Wetlands Policy Forum (1988) recognized the critical need to restore wetlands in agricultural areas because inadequate opportunities remain to acquire intact wetlands. In recent years, thousands of basins have been restored in the southern prairie pothole region in Iowa and Minnesota through a combination of federal, state and private initiatives (Galatowitsch and van der Valk 1994).

These initiatives in wetland restoration offer a unique opportunity. While wetland restoration has been driven primarily by concerns over waterfowl and habitat loss, wetlands may have great value in agricultural watersheds for their water quality functions. In fact, one of the most promising strategies for reducing nonpoint source contamination of surface and groundwaters in the prairie pothole region is the restoration of wetlands as sinks for agrochemical contaminants (van der Valk and Jolly 1992). However, because so little is known about the water quality functions of prairie wetlands, site selection criteria for wetland restorations have not generally considered water quality functions, and model simulations suggest that commonly used site selection criteria for prairie pothole restorations may be inadequate for water quality purposes (Crumpton et al. 1997). Our understanding of nutrient losses in treatment wetlands has progressed to a point that practical design models are now emerging for performance forecast modeling of nitrogen and phosphorous removal by wetlands (Kadlec and Knight 1996; Kadlec et al. 1997). Future research should seek to extend the application of performance forecast models to prairie wetlands using a watershed scale framework (Crumpton and Baker 1993; Crumpton et al. 1995). This will be made significantly more difficult by the spatial and temporal complexity of prairie pothole wetlands and by their hydrologic diversity. Research should not focus on case studies of specific wetlands but rather on identifying the principal factors controlling nitrogen transformation in prairie wetlands and on developing general predictive tools for modeling nitrogen fate in these systems.

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References

- Andersen, T. K., M. H. Jensen, and J. Sorensen. 1984. Diurnal variation of nitrogen cycling in coastal marine sediments. 1. Denitrification. *Marine Biology* 83:171-76.
- Barica, J., H. Kling, and J. Gibson. 1980. Experimental manipulation of algal bloom composition by nitrogen addition. *Canadian Journal of Fisheries and Aquatic Sciences* 37:1175-83.
- Batt, D. J., M. G. Anderson, and C. D. Anderson. 1989. The continental significance of prairie potholes for waterfowl. In *Northern Prairie Wetlands*, ed. A. G. van der Valk. Ames: Iowa State University Press.
- Berry, J. T. 1993. *Summary data, 1992 U.S. Fertilizer consumption, Tennessee Valley Authority Bulletin Y-232*. Muscle Shoals, AL: National Fertilizer and Environmental Research Center.
- Bowden, W. B. 1987. The biogeochemistry of nitrogen in freshwater wetlands. *Biogeochemistry* 4:313-48.
- Capone, D. G. and R. P. Kiene. 1988. Comparison of microbial dynamics in marine and freshwater sediments: Contrasts in anaerobic carbon metabolism. *Limnology and Oceanography* 33:725-49.
- Carper, G. L. and R. W. Bachmann. 1984. Wind resuspension of sediments in a prairie lake. *Canadian Journal of Fisheries and Aquatic Sciences* 41:1763-67.
- Christensen, P. B., L. P. Nielsen, N. P. Revsbech, and J. Sorensen. 1989. Microzonation of denitrification activity in stream sediments as studied with a combined oxygen and nitrous oxide microsensor. *Applied and Environmental Microbiology* 55:1234-41.
- Crumpton W. G., G. Atchison, C. Rose, E. Seabloom, S. Beauvais, J. Stenback, and S. Brewer. 1997. Experimental determination of ecological fate and effects of agrichemicals in surface waters of the western cornbelt ecoregion of the United States. USEPA Project completion report.
- Crumpton, W. G. and J. L. Baker. 1993. Integrating wetlands into agricultural drainage systems: Predictions of nitrate and loss in wetlands receiving agricultural subsurface drainage. *Proceedings, International Symposium on Integrated Resource Management & Landscape Modification for Environmental Protection*, December 13-14, 118-126. Chicago: American Society of Agricultural Engineers.
- Crumpton, W. G., J. L. Baker, J. Owens, C. Rose, and J. Stenback. 1995. Wetland and streams off-site sinks for agricultural chemicals. In *Clean*

- Water-Clean Environment- 21st Century, Volume I: Pesticides*, 49-53. St. Joseph, MI: American Society of Agricultural Engineers publication 2-95.
- Crumpton, W. G., S. W. Fisher, T. M. Isenhardt, M. P. Matter, and A. G. van der Valk. 1994. Transformations and fate of nitrate and atrazine in freshwater wetlands. Ames, IA: Leopold Center for Sustainable Agriculture, Project completion report.
- Galatowitsch, S. M. and A. G. van der Valk. 1994. *Restoring Prairie Wetlands: An Ecological Approach*. Ames: Iowa State University Press.
- Goldsborough, L. G. and G. G. C. Robinson. 1996. Pattern in wetlands, Chapter 4. In *Algal Ecology in Freshwater Benthic Ecosystems*, ed. R. J. Stevenson, M. L. Bothwell, and R. L. Lowe, 77-177. New York: Academic Press.
- Haertel, L. 1976. Nutrient limitation of algal standing crops in shallow prairie lakes. *Ecology* 57:664-78.
- Hargett, N. L. and J. T. Berry. 1981. *1980 Fertilizer Summary data, Tennessee Valley Authority Bulletin Y-165*. Muscle Shoals, AL: National Fertilizer and Environmental Research Center.
- Hecky, R. E. and P. Kilham. 1988. Nutrient limitation of phytoplankton in freshwater and marine environments: A review of recent evidence on the effects of enrichment. *Limnology and Oceanography* 33:796-822.
- Hooper-Reid, N. M. and G. G. C. Robinson. 1978. Seasonal dynamics of epiphytic algal growth in a marsh pond: Composition, metabolism, and nutrient availability. *Canadian Journal of Botany* 56:2441-48.
- Howard-Williams, C. 1985. Cycling and retention of nitrogen and phosphorus in wetlands: A theoretical and applied perspective. *Freshwater Biology* 15:391-431.
- Isenhardt, T. M. 1992. Transformation and fate of nitrate in northern prairie wetlands. Ph.D. diss., Iowa State University.
- Isenhardt, T. M. and W. G. Crumpton. In preparation. ISN studies of nitrate fate in wetlands receiving different nitrate loads.
- Johnston, C. A. 1991. Sediment and nutrient retention by freshwater wetlands: Effects on surface water quality. *Critical Reviews of Environmental Control* 21:491-565.
- Jorgensen, B. B. and J. Sorensen. 1985. Seasonal cycles of O_2 , NO_3^- , and SO_4^{2-} reduction in estuarine sediments: The significance of an NO_3^- reduction maximum. *Marine Ecology Progress Series* 24:65-74.
- Kadlec, J. A. 1986. Effects of flooding on dissolved and suspended nutrients in small diked marshes. *Canadian Journal of Fisheries and Aquatic Sciences* 43:1999-2008.

- Kadlec, R. H. and R. L. Knight. 1996. *Treatment Wetlands*. Boca Raton, FL: Lewis Publishers.
- Kadlec, R. H., W. G. Crumpton, and C. Rose. 1997. Nutrient dynamics in event driven wetlands. Project completion report submitted to Wetlands Research, Inc. Chicago, IL. and USEPA Region V.
- Kamp-Nielson, L. and J. M. Anderson. 1977. A review of the literature on sediment-water exchange of nitrogen compounds. *Progress in Water Technology* 3:393-418.
- Kantrud, H. A., J. B. Millar, and A. G. van der Valk. 1989. Vegetation of wetlands of the prairie pothole region. In *Northern Prairie Wetlands*, ed. van der Valk, A. G., 132-87. Ames: Iowa State University Press.
- Kasten, R.W. 1988. Legislative perspectives on wetland protection. In *Increasing Our Wetland Resources*, ed. J. Zelazny and J. Feierabend, 32-34. Washington, DC: National Wildlife Federation Corporate Conservation Council.
- Keeney, D. R. 1986. Sources of nitrate to groundwater. *CRC Critical Reviews of Environmental Control* 16:257-303.
- Korol, M. and L. Girard. 1996. Canadian fertilizer consumption, shipments and trade, 1994/1995. Ottawa, ON: Farm Input Markets Unit, Policy Branch, Agriculture and Agri-Food Canada.
- Moraghan, J. T. 1993. Loss and assimilation of ¹⁵N-nitrate added to a North Dakota cattail marsh. *Aquatic Botany*, 46:225-34.
- Murkin, H. R. 1989. The basis for food chains in prairie wetlands. In *Northern Prairie Wetlands*, ed. A. G. van der Valk, 316-39. Ames: Iowa State University Press.
- Murkin, H. R., Stainton, M. P., Boughen, J. A., Pollard, J. B. and Titman, R. D. 1991. Nutrient status of wetlands in the interlake region of Manitoba, Canada. *Wetlands* 11:105-22.
- Murkin, H. R. 1998. Freshwater functions and values of prairie wetlands. *Great Plains Research* 8:3-15.
- National Wetlands Policy Forum. 1988. *Protecting America's Wetlands: An Action Agenda*. Washington, DC: The Conservation Foundation.
- National Research Council. 1986. *Proceedings of the Colloquium on Agrichemical Management and Water Quality*. Washington, DC: National Academic Press.
- Neill, C. 1995. Seasonal flooding, nitrogen mineralization and nitrogen utilization in a prairie marsh. *Biogeochemistry* 30:171-89.
- Neely, R. K. and C. B. Davis. 1985. Nitrogen and phosphorus fertilization of *Sparaganium eurycarpum* Engelm. and *Typha glauca* Godr. stands. II: Emergent plant decomposition. *Aquatic Botany* 22:363-75.

- Neely, R. K. and J. L. Baker. 1989. Nitrogen and phosphorous dynamics and the fate of agricultural runoff. In *Northern Prairie Wetlands*, ed. A.G. van der Valk, 92-131. Ames: Iowa State University Press.
- Neill, C. 1990. Effects of nutrients and water levels on emergent macrophyte biomass in a prairie marsh. *Canadian Journal of Botany* 68:1007-14.
- Omernik, J. M. 1977. Nonpoint source-stream nutrient level relationships: A nationwide study. EPA-60013-77-105. Corvallis, OR: U.S. Environmental Protection Agency.
- Painter, H. A. 1977. Microbial transformations of inorganic nitrogen. *Progress in Water Technology* 8:3-29.
- Phipps, R. G. 1997. Nitrate removal capacity of wetlands. Ph.D. diss., Iowa State University.
- Reddy, K. R. and D. A. Graetz. 1988. Carbon and nitrogen dynamics in wetland soils. In *The Ecology and Management of Wetlands*, ed. D.D. Hook et al., 307-18. Portland, OR: Timber Press.
- Reddy, K. R., W. H. Patrick Jr., and C. W. Lindau. 1989. Nitrification-denitrification at the plant root-sediment interface in wetlands. *Limnology Oceanography* 34:1004-13.
- Redfield, A. C. 1958. The biological control of chemical factors in the environment. *American Scientist* 46:205-21.
- Robinson, G. G. C., S. E. Gurney, and L. G. Goldsborough. In press. Algae in prairie wetlands. In *Prairie Wetland Ecology: The State of Our Understanding and the Contribution of the Marsh Ecology Research Program*, ed. H. Murkin, J. Kadlec and A. van der Valk.
- Rose, C. 1996. Effects of emergent vegetation on wetland microbial processes. Ph.D. diss., Iowa State University.
- Rose, C. and W. G. Crumpton, 1996. Effects of emergent macrophytes on dissolved oxygen dynamics in a prairie pothole wetland. *Wetlands* 16: 495-502.
- Seitzinger, S.P. 1988. Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. *Limnology and Oceanography* 33:702-24.
- Shay, J. M. and C. T. Shay. 1986. Prairie marshes in western Canada with specific reference to the ecology of five emergent macrophytes. *Canadian Journal of Botany* 64:443-54.
- Stuber, P. J. 1988. Proceedings of the National Symposium on Protection of Wetlands from Agricultural Impacts. Washington, DC: U. S. Fish and Wildlife Service, Biol. Rep. 88(16).
- Tiner, J. R. 1984. Wetlands of the United States: Current status and recent trends. Washington, DC: U. S. Fish and Wildlife Service.

- van der Valk, A. G. and C. B. Davis. 1978a. The role of seed banks in the vegetation dynamics of prairie glacial marshes. *Ecology* 59:322-335.
- van der Valk, A. G. and C. B. Davis. 1978b. Primary production in prairie glacial marshes. In *Freshwater Wetlands: Ecological Processes and Management Potential*, ed. R. E. Good et al., 21-37. New York: Academic Press.
- van der Valk, A. G. and R. W. Jolly. 1992. Recommendations for research to develop guidelines for the use of wetlands to control rural NPS pollution. *Ecological Engineering* 1: 115-34.
- Ward, M. D. and M. R. Winfrey. 1985. Interactions between methanogenic and sulfate-reducing bacteria in sediments. In *Advances in Aquatic Microbiology*, Vol. 3, ed. H. W. Jannasch and P. J. LeB. Williams, 141-68. London: Academic Press.
- Weisner, S. E. B., P. G. Eriksson, W. Graneli, and L. Leonardson. 1994. Influence of macrophytes on nitrate removal in wetlands. *Ambio* 23:366.