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Review and Interpretation: Nitrogen Management Strategies to Reduce Nitrate Leaching in Tile-Drained Midwestern Soils

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REVIEW AND INTERPRETATION

Nitrogen Management Strategies to Reduce Nitrate Leaching in Tile-Drained Midwestern Soils

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ABSTRACT

Balancing the amount of N needed for optimum plant growth while minimizing the NO_3 that is transported to ground and surface waters remains a major challenge for everyone attempting to understand and improve agricultural nutrient use efficiency. Our objectives for this review are to examine how changes in agricultural management practices during the past century have affected N in midwestern soils and to identify the types of research and management practices needed to reduce the potential for nonpoint NO_3 leakage into water resources. Inherent soil characteristics and management practices contributing to nonpoint NO_3 loss from midwestern soils, the impact of NO_3 loading on surface water quality, improved N management strategies, and research needs are discussed. Artificial drainage systems can have a significant impact on water quality because they behave like shallow, direct conduits to surface waters. Nonpoint loss of NO_3 from fields to water resources, however, is not caused by any single factor. Rather, it is caused by a combination of factors, including tillage, drainage, crop selection, soil organic matter levels, hydrology, and temperature and precipitation patterns. Strategies for reducing NO_3 loss through drainage include improved timing of N application at appropriate rates, using soil tests and plant monitoring, diversifying crop rotations, using cover crops, reducing tillage, optimizing N application techniques, and using nitrification inhibitors. Nitrate can also be removed from water by establishing wetlands or biofilters. Research that is focused on understanding methods to minimize NO_3 contamination of water resources should also be used to educate the public about the complexity of the problem and the need for multiple management strategies to solve the problem across agricultural landscapes.

NITROGEN IS ESSENTIAL for growth and reproduction of all life forms, and except for legume crops and virgin soils with relatively high soil organic matter (SOM), soil N must usually be supplemented to sustain food, feed, and fiber production. During the past 20 yr, public concern regarding N movement from agricultural nonpoint sources into broader water resources has increased as problems such as hypoxia (Rabalais et al., 1996) became more evident. To understand current questions about N management in the U.S. northern Corn and Soybean Belt, it is necessary to examine the changes that have occurred in agriculture during the past century. These changes include the use of less diversified crop rotations, separation of crop production and animal enterprises, changes in tillage intensity, drainage of ag-

ricultural fields, and increased use of manufactured N fertilizers.

Less than 50 yr ago, corn (*Zea mays* L.) was generally grown in rotation with cereal crops and forage legumes such as alfalfa (*Medicago sativa* L.), red clover (*Trifolium pratense* L.), and sweetclover (*Melilotus* spp.). Through biological N fixation, the legumes generally increased the amount of residual N in the soil profile. Cycling of this residual, biologically fixed N along with N mineralized from SOM added with animal manure or deposited through rainfall was the primary process through which corn and other grain crops obtained N.

Following World War II, increased availability of commercial N fertilizer and decreased demand for forage crops led to a significant reduction in crop rotations and a general substitution of purchased N for biological N. In Iowa, forage pasture represented more than 33.6% (3 389 160 ha) of the state's total cropped area at the end of World War II (U.S. Dep. of Commerce, Bureau of the Census, 1945). By 1997, forage pasture area in Iowa comprised only 12.8% (1 393 451 ha) of the state's total cropped area (USDA Natl. Agric. Stat. Serv., 1997). Incorporation of legumes into a crop rotation was no longer needed as commercial N inputs gradually replaced biological N fixation.

The increased availability of commercial N also facilitated specialization and a national trend for separating crop and animal production enterprises. Animal manure no longer served as an important crop nutrient resource, and meadow legumes were not required on farms that began to specialize in corn and soybean [*Glycine max* (L.) Merr.] production. Although there is considerable variation among years and regions in N fertilizer usage, the net result of this farming-system change was a national average increase in commercial N fertilizer use of $2.4 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (Fig. 1) between the mid-1960s and the late 1990s. The rise in commercial N fertilizer usage within Midwest states (Illinois, Iowa, Indiana, Michigan, Minnesota, Missouri, Nebraska, Ohio, and Wisconsin) has slowed, with average use of commercial N fertilizer increasing by 4 kg ha^{-1} ($0.5 \text{ kg ha}^{-1} \text{ yr}^{-1}$) from 1991 to 1999 (Fig. 2). However, the trend of steadily increased usage of commercial N fertilizer may change if fossil fuel prices increase substantially and the fertilizer industry is

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Abbreviations: LCD, localized compaction and doming; LSNT, late-spring nitrate test; NIR, near infrared; PSNT, presidedress soil nitrate test; SI–CD, subirrigation with controlled drainage; SOM, soil organic matter; WTM, water table management.

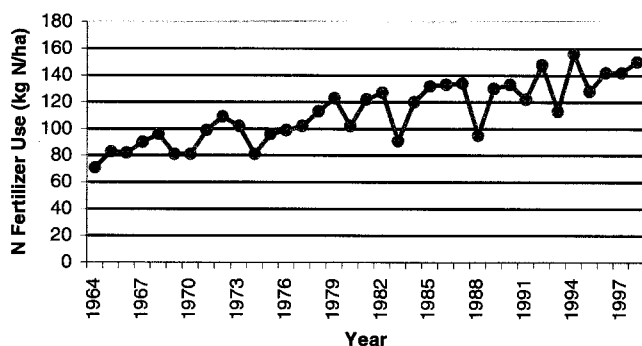


Fig. 1. Average U.S. commercial N fertilizer use for corn production for the years 1964–1998.

unable to develop lower-cost energy sources or new strategies for fertilizer N production. An alternative to commercial N fertilizer inputs would be to reconsider some of the traditional N management practices, especially if they can be made more efficient and predictable by applying 21st century technologies.

In addition to decreased diversity in crop rotation, separation of crop and animal production enterprises, and use of more commercial N fertilizers, an even more dramatic change affecting N cycling within many midwestern soils was the installation of artificial subsurface drainage lines. Subsurface drainage lines began to be installed in the late 1800s in the Midwest, intensified in the next century (Hewes and Frandson, 1952), and continues today. By 1987, more than 20.8×10^6 ha in the states of Illinois, Indiana, Iowa, Ohio, Minnesota, Michigan, Missouri, and Wisconsin had been artificially drained (Zucker and Brown, 1998) in contrast to approximately 2×10^6 ha being irrigated in the same states (USDA Natl. Agric. Stat. Serv., 1999). Subsurface artificial drainage thus preceded and occurred in coincidence with other management changes in midwestern agriculture, most likely catalyzing other crop production and field management changes.

Drainage was deemed necessary and strongly promoted for the betterment of farmers, communities, and the nation as a whole because natural surface drainage was limited throughout much of the region. As a result, the landscape was significantly changed, and many areas that were previously classified as *prairie potholes* could now be converted to row crop production. Artificial drainage and an increased availability of N fertilizers are two of the most prominent practices that facilitated a tremendous increase in the intensity of agricultural production throughout the midwestern USA. These changes had enormous positive impact on the agricultural economy and the expansion of agricultural exports. However, they drastically altered the farming practices that had evolved within the midwestern landscape and contributed to changes in soil and water quality that had an impact far beyond farm fields.

The ecosystem effects of artificial subsurface drainage in the Midwest were compounded because many of the soils that had developed under a subhumid climate, in areas of low relief and poor surface drainage, had high (>5–6%) levels of organic matter. The predominately

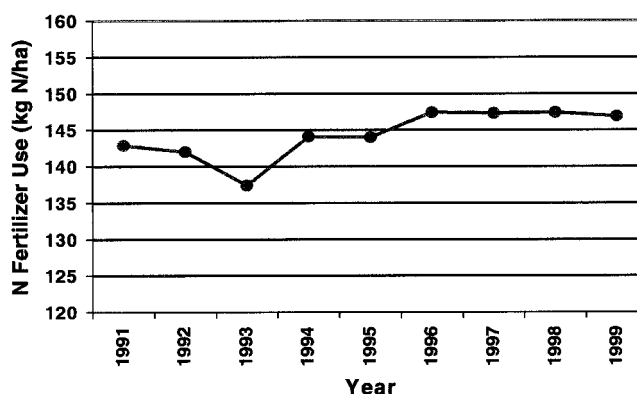


Fig. 2. Average commercial N fertilizer use in the Midwest for corn production for the years 1991–1997.

wet condition of these soils was modified by installation of artificial drainage, tillage to prepare a seedbed for agricultural crops, and a change from perennial to seasonal vegetation (Hewes and Frandson, 1952). With these changes, the potential for mineralization of N from the stored organic matter and N loading of surface waters increased significantly. As stated by Randall (1997), “Soils high in organic matter can mineralize a substantial amount of nitrate-N which is susceptible to loss in subsurface tile drainage, especially when wet years follow very dry years.” Collectively, the unintended side effect of these changes in cultural practices and N management strategies has been an apparent reduction in N-cycling efficiency compared with natural ecosystems or crop production systems that have little to no reliance on commercial N inputs.

Each agricultural region has its own specific environmental challenges or imbalances associated with land use decisions and crop production practices. In the Midwest, artificially drained areas, increased use of synthetic fertilizers, and decreased diversity in crop rotation are among the most notable causes of this region’s problem with N contamination of water resources. The objectives of this review are to (i) examine the scope of the NO_3 leaching problems associated with artificially drained midwestern soils, (ii) discuss current strategies for reducing NO_3 losses from these soils, and (iii) identify future research needs to develop new strategies for reducing NO_3 leaching losses. The desired impact of research on field- and watershed-scale NO_3 loss and opportunities for improving overall water quality by increasing N use efficiency are discussed.

NITRATE LEACHING LOSSES FROM AGRICULTURAL SYSTEMS

We will focus the attention of this paper on the NO_3 form of N because a number of leaching and drainage studies have consistently found that NO_3 is the dominant form of N present in the soil water (Willrich, 1969; Baker et al., 1975; Kladvko et al., 1991; Jacinthe et al., 1999). Soil NO_3 can be derived from both organic and inorganic N. Whether the N source is animal manure or commercial N fertilizer, overapplication or ill-timed application of either source can provide too much plant-available

N and increase the potential for NO_3 leaching (Hatfield and Cambardella, 2001). Animal manure N sources provide many physical, chemical, and biological benefits to the soil environment beyond that of inorganic commercial N sources (Hatfield and Cambardella, 2001), and a thorough discussion of these aspects and the N dynamics of manures would be very extensive. Therefore, within this review, we will not specifically discuss manure management issues. Instead, we will focus on commercial fertilizer and soil or crop management impacts on NO_3 leaching.

Nitrate contributes to surface water degradation when it flows into subsurface drainage lines that discharge into streams and lakes or when it leaches below the active plant-root zone and into shallow ground water resources that connect to surface water bodies through natural processes such as baseflow. The intensification of row crop production as a whole, not just the increased use of N fertilizers, has been identified as the primary cause for increased NO_3 contamination of surface waters during the past several decades (Keeney and DeLuca, 1993). This is especially true for continuous corn production, which has repeatedly been identified as providing the greatest amount of NO_3 to streams through subsurface drainage (Kanwar et al., 1993; Weed and Kanwar, 1996; Randall et al., 1997a).

Field drainage systems can have a significant impact on water quality because they behave like shallow, direct pipelines or conduits to surface waters. They increase the speed with which water moves off the landscape by short-circuiting natural water flow into shallow ground water. Nutrients and pesticides are often transported with subsurface drainage water directly into streams or lakes (Baker and Johnson, 1981; Buhler et al., 1993; Jayachandran et al., 1994; Kladienko et al., 1991; Randall et al., 1997a). Numerous studies have shown significant edge-of-field losses of NO_3 (Hanway and Laflen, 1974; Gast et al., 1978; Miller, 1979; Benoit, 1973; Logan et al., 1980; Baker and Johnson, 1981; Bergström, 1987; Kanwar et al., 1988; Drury et al., 1996). One example is a study reported by Baker et al. (1975) where they found average $\text{NO}_3\text{-N}$ concentrations of 21 mg L^{-1} in subsurface drainage water leaving fields planted to corn-soybean or corn-oat (*Avena sativa* L.) rotations. Similarly, for a 5130-ha watershed located on the Des Moines lobe in central Iowa, Jaynes et al. (1999) reported flow-weighted $\text{NO}_3\text{-N}$ concentrations in field and county agricultural drainage lines that were often greater than the USEPA maximum contamination level (MCL) for drinking water of 10 mg L^{-1} , especially from April through July. Combining stream flow and $\text{NO}_3\text{-N}$ concentration data for this intensively row-cropped agricultural watershed showed that total $\text{NO}_3\text{-N}$ losses ranged from 4 to $66 \text{ kg ha}^{-1} \text{ yr}^{-1}$. The variation in $\text{NO}_3\text{-N}$ losses among years was directly linked to variation in annual precipitation (Hatfield et al., 1998). Those watershed-scale measurements also showed that 45% of the average annual precipitation was drained from the soil profile through the subsurface drainage lines. This emphasizes the importance of these drainage lines as a primary pathway for herbicide and NO_3 movement to

surface waters (Hatfield et al., 1998; Hatfield et al., 1999).

Current practices of N fertilizer management are often very inefficient compared with natural systems, thus increasing the potential for contamination of water resources (Sanchez and Blackmer, 1988; Kanwar et al., 1993, 1996; Randall, 1997; Randall et al., 1997a; Cambardella et al., 1999). As a result, first-year recoveries of fertilizer N by corn have been reported to be 35% (Bijeriego et al., 1979), 14 to 65% (Meisinger et al., 1985), 23 to 45% (Kitur et al., 1984), 24 to 26% (Olson, 1980), 15 to 33% (Sanchez and Blackmer, 1988), and 45 to 59% (Reddy and Reddy, 1993). Based on these N fertilizer recovery studies, one factor contributing to low fertilizer N recovery is that in any given year, corn plants can obtain up to 85% of their N from mineralized SOM.

The potential impact of soil-derived N can be illustrated by estimating the amount of plant-available N associated with the SOM in many midwestern soils. Assuming there is approximately $4.5 \times 10^6 \text{ kg}$ of soil per hectare in the surface 30 cm and an average 10:1 C/N ratio for the SOM, there would be $4500 \text{ kg N ha}^{-1}$ for 1% SOM. Therefore, crops grown on a soil with 3% SOM, which would be about average for the subhumid Midwest, would have the potential to obtain $13\,500 \text{ kg N ha}^{-1}$ just from the SOM. Obviously, only a small percentage of this soil-derived N would be available at any given point in time because it must first be mineralized by microbes to a plant-available form such as NH_4 or NO_3 and the amount mineralized may not be enough to meet crop N needs during rapid growth periods.

Schepers and Mosier (1991) stated that for a given climatic region, a general estimate of N mineralization could be made based on SOM content. They estimated that assuming 2% of the total organic N in the surface 30 cm is mineralized annually, a soil with 1% SOM content could be expected to mineralize approximately $45 \text{ kg N ha}^{-1} \text{ yr}^{-1}$. It is important to remember that these are general estimates because the amount of organic N made available through mineralization processes will vary greatly over time due to factors such as temperature, precipitation, and tillage (Doran, 1980; Franzluebbers et al., 1995; Wienhold and Halvorson, 1999). However, because of their high SOM levels, these estimates illustrate that midwestern soils have a high potential for providing N throughout the entire growing season.

With regard to N management, including both fertilizer and manure application, the SOM-N pool is extremely important for midwestern soils. Routine soil and crop management decisions not only affect the mineralization process per se, but also the rates and timing of N release through interactions between the microbial communities and practices such as drainage or tillage and/or residue management. As a result, fertilizer N that is not taken up by the crop to which it was applied can have many different fates. If rainfall is below normal for midwestern conditions, NO_3 can accumulate within the soil profile. It can also be leached below the active plant root zone; lost through denitrification; or incorporated, cycled, and stabilized within many different SOM-N pools. These factors significantly affect N man-

agement in midwestern soils. For example, in Iowa alone, Swoboda (1990) reported that failure to account for residual N caused farmers to annually spend approximately \$100 million on excess N fertilizer applications. This is not only an on-farm economic waste, but also an environmental waste with regard to both water quality and the fossil energy needed to manufacture N fertilizer.

Nonpoint contamination is a major water quality concern throughout the Midwest (Humenik et al., 1987), and use of N fertilizers on artificially drained agricultural land has been implicated as a major contributor to this problem (USEPA, 1992; U.S. Geological Survey, 1995; CAST, 1999). The 1992 national water quality inventory (USEPA, 1992) notes that in the rivers studied, 72% of the water quality problems were attributed to agriculture. Plant nutrients have been identified as surface-water contaminants throughout the Midwest (Baker, 1988; Thurman et al., 1992; USEPA, 1992; Goolsby and Battaglin, 1993). Nitrate N concentrations in excess of 10 mg L⁻¹ in drinking water may pose risks to humans and livestock (USDA, 1991; Tyson et al., 1992) and have cost some communities millions of dollars for their removal or to provide alternate drinking water sources. For example, Des Moines, IA alone has spent in excess of \$4.8 million for NO₃ removal from drinking waters between 1991 and 1999 (G. Benjamin, unpublished data, 2000).

Nitrogen loadings into the Mississippi River and its tributaries have also been identified as a potential cause for degradation of freshwater and marine ecosystems. Elevated N concentrations have altered natural aquatic floral and faunal population dynamics, exacerbated occurrences of hypoxia and anoxia, and sped the process of eutrophication in the Gulf of Mexico (Alexander et al., 1995; Rabalais et al., 1996). Turner and Rabalais (1991) reported that increased levels of NO₃ in the Mississippi River have paralleled increased use of commercial fertilizer throughout the river basin and increased severity of hypoxic events in the Gulf of Mexico since the 1950s. Alexander et al. (1995) estimated that drainage from the Upper and Central Mississippi Basins (including portions of Minnesota, Wisconsin, Iowa, Missouri, and Illinois) accounts for 39% of the N delivered to the Gulf. This is the largest estimated source fraction among the Mississippi River's various drainage basins.

STRATEGIES FOR IMPROVED NITROGEN MANAGEMENT

Timing of Application and Nitrogen Rates

Many attempts to reduce NO₃ concentrations in shallow ground-water draining to tiles have focused on timing and N application rate. Randall (1997) stated that "... fertilizer N management, particularly rate and time of application, plays a dominant role in the loss of nitrate to surface waters." The challenge is to manipulate N availability before, during, and after peak crop demand. Nitrogen molecules are susceptible to leaching, denitrification, volatilization, and immobilization processes within the soil environment. The risk of N losses due

to these processes increases as the time between N application and crop uptake increases. This is true for residual N as well as applied N (Magdoff, 1991; Karlen et al., 1998), especially in years that do not produce optimal yields (Power et al., 1998). Limiting the amount of inorganic N within the soil at the end of a crop's growing season and before the next crop has established an extensive root system is a key factor for reducing N losses. Therefore, although timing, method of N application, and accounting for mineralizable soil N are important for reducing potential NO₃ leaching, Power and Schepers (1989) concluded that the most important factor was to apply the correct amount of N fertilizer.

Typical N fertilizer management for corn production in the subhumid Midwest currently consists of a single preplant application, usually in autumn before the year that corn is grown. This practice was promoted by agricultural experts because the potential for soil compaction following harvest is generally less, labor is often more available, weather and soil conditions are generally more favorable, and fertilizer prices are frequently lower than in the spring. However, fall application places the applied N in the soil several months before the crop needs it, and thus increases the potential for leaching or other losses. Sanchez and Blackmer (1988) reported that 49 to 64% of the fall-applied fertilizer N was lost from the upper 1.5 m of the soil profile through pathways other than plant uptake.

Changing the timing of a single preplant fertilizer application from fall to spring could significantly decrease N loss and increase fertilizer use efficiency. This was demonstrated for southern Minnesota (Randall et al., 1992; Randall, 1997) where spring application increased N use efficiency by more than 20% compared with fall N application. In addition, annual NO₃ losses from tile drainage were reduced by an average of 36%. Despite the opportunities for increased nutrient use efficiency and decreased loss of N through drainage waters, many farmers continue fall fertilizer applications to minimize real and perceived risk. Spring rainfall patterns can result in very wet soils and prevent or delay N fertilizer applications. This risk is very real because there are few options in most rainfed farming operations to compensate the farmer for yield losses and reduced income associated with an inadequate N supply. Therefore, to achieve significant farmer adoption of N management practices other than fall fertilization, concepts such as insurance policies against N deficiencies are needed along with more flexible and efficient application methods.

Power et al. (2000) reported that midwestern fields frequently have a high degree of variability in soil NO₃ content from site to site within a single field. They stated that "... soils are seldom uniform throughout a field, so applying sufficient fertilizer N to assure high yields for more productive areas of the field often results in over-fertilization of the less productive areas. This may lead to greater nitrate leaching, particularly in those areas of the field that are more susceptible to leaching." Kranz and Kanwar (1995) estimated that within a given field, 70% of the NO₃ leached typically comes from

<30% of its area. To be effective, variable-rate technology must accurately identify within-field spatial and temporal variability and reliably interpret these patterns of variability (Sawyer, 1994). If this is accomplished, a reliable variable-rate or site-specific N fertilizer application program could reduce application on areas requiring little to no additional N to optimize yield (Dinnes et al., 1998) and could limit application on areas sensitive to leaching and surface runoff (Larson and Robert, 1991). Methodologies to accurately assess within-field spatial and temporal variability for subsequent N application, however, have yet to be fully developed.

Most conventional methods of N fertilizer recommendation were developed on a state or regional scale, so it is questionable whether these methods can reasonably be used for variable-rate N management that attempts to account for within-field spatial and temporal variability (Hergert et al., 1997). Several research studies have found large differences in crop yield and crop N response within individual fields (Ferguson et al., 1995; Kitchen et al., 1995; Vetch et al., 1995), confirming the need for reliable methods to generate site-specific N recommendations (Hergert et al., 1997).

Pierce and Nowak (1999) discussed three basic management approaches currently being tested for variable-rate N application. The first involves determining plant-available N levels from field grid sampling and interpreting N rates based on current recommendation methods (i.e., a N balance equation). The second approach bases N rates on observed crop N responses using replicated strips with varying N rates across the landscape. The third approach involves determining crop N status by monitoring (i.e., light reflectance or chlorophyll content). Usually this intervention-type approach uses a portion of the crop that is well fertilized as a standard for comparison. The best approach for a particular field, be it conventional or site-specific, will depend on the amount of spatial and temporal N variability (Pierce and Nowak, 1999). Pan et al. (1997) reported that temporal N variability could frequently exceed spatial variability. While highly predictable spatial variability may be amenable to multiple approaches of site-specific N management, strong temporal variability is much more difficult to manage (Pierce and Nowak, 1999). In this case, Pierce and Nowak (1999) concluded that an intervention strategy would be the most appropriate option.

Crediting Nitrogen Mineralization

Nitrate, as an end product of mineralization and subsequent nitrification of SOM, manure, crop residue, or previously applied fertilizer N that has cycled through soil organic N pools, can make significant contributions toward meeting crop N requirements (Oberle and Keeney, 1990; Kanwar et al., 1996). However, to ensure NO_3 is used efficiently, it is important to quantify the amount being produced and, where possible, minimize its accumulation during noncropping periods. This strategy is important when crop uptake is minimal or nonexistent because much of the NO_3 produced during those periods is susceptible to leaching (Fig. 3). Strategies

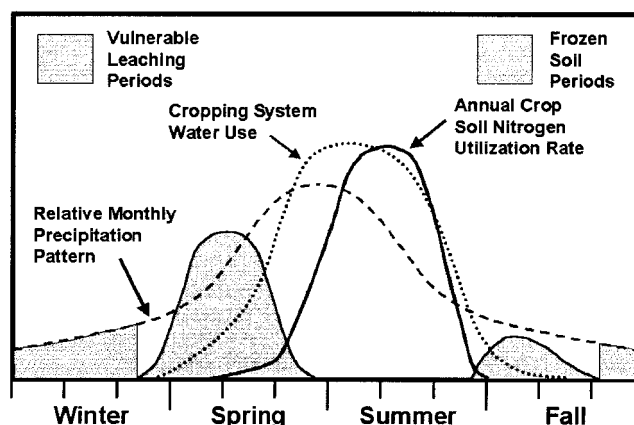


Fig. 3. General seasonal patterns for precipitation, N uptake rate by a corn crop, cropping system water use, and periods potentially favorable for NO_3 leaching from midwestern corn production (adapted from Fig. 4 of Power et al., 1998).

to slow NO_3 accumulation include use of nitrification inhibitors (Kidwaro and Kephart, 1998) or temporary immobilization by cycling N through cover crops (Doran and Smith, 1991; Staver and Brinsfield, 1998; Waggoner et al., 1998). Minimizing NO_3 accumulation until the primary crop is actively growing can generally reduce leaching loss because, by that time, plant water use will generally exceed available precipitation (Hatfield et al., 1998; Power et al., 1998).

Monitoring N mineralization to better match the required amount of available N with crop needs is one strategy for reducing NO_3 leaching potential. To accomplish this task, several versions of a presidedress soil nitrate test (PSNT) (Magdoff et al., 1984; Fox et al., 1989; Magdoff et al., 1990) or modifications such as the late-spring nitrate test (LSNT) (Blackmer et al., 1997) have been developed. These tests generally involve sampling the soil approximately 6 wk after planting. The philosophy behind the tests is that by late spring, the net effects of mineralization, leaching, and other potential losses that may have occurred since the last crop was harvested can be accurately assessed. Soil test results can then be used to determine the appropriate amount of additional N fertilizer to apply.

Plot-scale studies using PSNT or LSNT strategies to determine fertilizer N rates have generally shown reductions in measured or potential NO_3 leaching. In Iowa, these procedures resulted in fertilizer N applications ranging from 50 to 168 kg N ha⁻¹ and significantly reduced NO_3 loss to subsurface drainage tiles compared with single preplant applications of only 112 kg N ha⁻¹ (Kanwar et al., 1996). A study in Vermont (Durieux et al., 1995) also reduced leaching potential compared with a yield-goal N management strategy. In this 3-yr study, the PSNT-LSNT N management program resulted in fertilization rates of 112, 123, and 123 kg N ha⁻¹ and corresponding residual soil NO_3 levels in the upper 1.2 m of the soil profile of 87, 68, and 44 kg N ha⁻¹ in the fall. In contrast, the yield-goal management program required 168 kg N ha⁻¹ and resulted in residual soil NO_3 levels in the upper 1.2 m of the soil of 138, 160, and 156 kg N ha⁻¹, respectively. These plot-scale results suggest

that the PSNT-LSNT approach for N management has the potential for avoiding excess N application compared with yield-goal approaches (Magdoff, 1991; Durieux et al., 1995; Kanwar et al., 1996; Randall, 1997; Karlen et al., 1998). However, while designed to provide an optimum N fertilizer rate at an appropriate time to improve crop N use efficiency, the impacts on water quality and risk to the farmer for adopting this approach have been poorly quantified at the whole-field scale and have only begun to be evaluated at the watershed scale.

A regional study (Bundy et al., 1999) determined that the PSNT-LSNT program at the watershed scale failed to identify some field sites where a yield response to N fertilizer was not achieved. Their follow-up investigation found that such errors were reduced by increasing the soil-sampling depth from 30 to 60 cm (Bundy et al., 1999). The end-of-season basal stalk NO_3 test (Blackmer and Mallarino, 1997) has also been used to verify the accuracy of the PSNT-LSNT program. Stalk NO_3 concentrations are categorized into four groups: low ($<250 \text{ mg kg}^{-1} \text{ NO}_3$), marginal ($250\text{--}700 \text{ mg kg}^{-1} \text{ NO}_3$), optimal ($700\text{--}2000 \text{ mg kg}^{-1} \text{ NO}_3$), and excess ($>2000 \text{ mg kg}^{-1} \text{ NO}_3$). Although the information does not assist in correcting in-season deficiencies, it is useful for guiding future operations to improve N use efficiency. This test is helpful in assessing the performance of any N management program for corn production (Magdoff, 1991; Blackmer and Mallarino, 1997).

Fertilizer Application Equipment

When measuring the efficiency of any system, the variation or error of each step is additive. Therefore, the overall efficiency is only as good as the summed margins of error for each step. For site-specific N management, attempting to apply variable N rates will do little to improve overall N fertilizer use efficiency if the application equipment has a high degree of error. This problem was confronted in studies examining the performance of conventional anhydrous NH_3 manifolds. Weber et al. (1993) found appreciable degrees of error over entire fields, and other investigators documented large degrees of error across the individual outlets of the equipments' manifolds (Reichenberger, 1994; Fee, 1997; Schrock et al., 1999).

Research efforts have concentrated on evaluating new technology to reduce equipment application error. Boyd et al. (2000) conducted an evaluation of anhydrous NH_3 manifolds under field conditions at two target application rates (84 and 168 kg N ha^{-1}). Several anhydrous NH_3 manifold entry methods were used for conventional manifolds, and although each method had differing coefficients of variation at both N rates, all had similar trends, with greater coefficients of variation at 84 kg N ha^{-1} ($12\text{--}80\%$) than at 168 kg N ha^{-1} ($10\text{--}66\%$). Alternative manifold designs typically performed better than conventional manifolds at both rates, with less variability at the high rate. Rotaflo and small-housing Vertical Dam manifolds produced the least amount of variability across all outlets at both the low and high N rates. A John Blue FD-1200 manifold had a low variability at the higher rate.

Hanna et al. (1999) measured anhydrous NH_3 distribution for several manifold designs during field application at three flow rates (56 , 112 , and 168 kg N ha^{-1}). This experiment accounted for dynamic vibration of the applicator's distribution system, which can affect the anhydrous NH_3 vapor-liquid phase separation within the manifold. They found that a Vertical Dam manifold had less variability than a conventional manifold at the 56 kg N ha^{-1} rate. At the two higher N rates, they detected little difference in variability between the two designs. When examining flow rates by manifold outlet location in relation to incoming flow, Hanna et al. (1999) found that the highest flow rates came from outlet ports located directly across from the incoming flow port. Next highest flow rates came from outlet ports near the incoming flow port. They theorized that this was due to flow reflected from the opposite side of the manifold chamber. The least amount of flow came from outlet ports located on both sides perpendicular to the incoming flow path. The conventional manifold exhibited these traits to a stronger degree than alternative designs, but all manifolds showed similar patterns.

Liquid N fertilizer applicators tend to have less variation in fertilizer distribution across all outlets along the implements' toolbars than conventional anhydrous NH_3 applicators. Recently, liquid N fertilizer applicators have become even more accurate with the use of hydraulic flow rate control devices. With careful installation and calibration, applied rates can easily be within 1 to 4% of the target rate (Dinnes et al., unpublished data, 2000).

Some recent innovations in the design and function of N fertilizer application equipment have focused on strategies to limit N losses from leaching, volatilization, and runoff. One, a localized compaction and doming (LCD) applicator (Ressler et al., 1997), was developed to alter soil physical properties immediately above the soil volume where knife-injected liquid N fertilizer was placed. A compacted zone with a higher bulk density is created above the injection zone so that water infiltration through that area would be reduced compared with a conventional knife-injection applicator. When compared with either conventional knife or broadcast methods of N application, the LCD design reduced the amount of NO_3 leaching following an intense rainfall that occurred soon after fertilizer application (Ressler et al., 1998).

Another application technique developed to improve N management was the point injector, which under no-till conditions, was demonstrated to have the potential to reduce NH_3 volatilization and N immobilization at the surface without destroying surface residues or adversely affecting corn yield (Baker et al., 1985). In Minnesota (Randall et al., 1997b), point injection of urea ammonium nitrate resulted in greater corn grain yield and total N uptake compared with surface-broadcast or banded urea ammonium nitrate in a ridge tillage system. Additional agricultural drainage studies in Iowa from 1995 through 1997 (Iowa Dep. of Agric. and Land Stewardship, 1997) showed that both LCD and point injection methods consistently reduced NO_3 concentrations in tile drainage water compared with conventional

knife-injection methods. In addition to having the lowest NO_3 concentrations in the drainage water, the LCD application method also resulted in the highest corn grain yield.

Nitrification Inhibitors

Nitrification inhibitors for N fertilizers have met with varying success, generally depending on soil type and weather pattern under which they were used. These inhibitors function by limiting the activity and population of *Nitrosomonas* bacteria that convert NH_4 to NO_2 . The primary use for nitrification inhibitors in the Upper Midwest is to slow the conversion of fall-applied anhydrous NH_3 fertilizer to the more leachable NO_3 form, thus potentially reducing N fertilizer losses before peak N demand by subsequent corn crops.

An 8-yr project in Ohio (Stehouwer and Johnson, 1990) examined different application timings with and without a nitrification inhibitor. The results showed that at similar N rates, spring preplant application of urea $[(\text{NH}_2)_2\text{CO}]$ or anhydrous NH_3 produced higher yields than fall applications. Addition of nitrapyrin [2-chloro-6-(trichloromethyl)-pyridine] as a nitrification inhibitor with spring-applied N had no effect on grain yield but did increase yields associated with fall N applications. Similarly, in Minnesota, Randall et al. (1992) found that corn yield and N use efficiency were lowest with fall N applications without nitrapyrin, highest with spring N applications, and intermediate with fall N plus nitrapyrin. They reported N use efficiency of fall-applied N fertilizer was 16%, whereas with nitrapyrin, it improved to 26%. Spring applications, with or without nitrapyrin, had fertilizer N use efficiencies ranging from 42 to 48%. The highest NO_3 losses associated with corn production occurred with fall N applications, with or without nitrapyrin.

The economics of nitrapyrin use have shown mixed results. In a 7-yr study by Christensen and Huffman (1992), nitrapyrin in spring-applied N fertilizer significantly increased corn grain yield, with the increase more than compensating for the added cost of the nitrapyrin. In Minnesota, 7-yr averages of corn fertilized with 150 kg N ha^{-1} showed that yield was increased 5% by fall-applied N with nitrapyrin, 5% by spring preplant N, and 10% by split-applied N (40% preplant and 60% sidedressed at V8 corn growth stage) compared with fall-applied N without nitrapyrin (G.W. Randall, personal communication, 2000). Economic returns for the 7-yr study were split-applied N ($\$239.40 \text{ ha}^{-1} \text{ yr}^{-1}$) > spring preplant N ($\$210.90 \text{ ha}^{-1} \text{ yr}^{-1}$) > fall-applied N with nitrapyrin ($\$192.40 \text{ ha}^{-1} \text{ yr}^{-1}$) > fall-applied N without nitrapyrin ($\$166.70 \text{ ha}^{-1} \text{ yr}^{-1}$). Although the nitrapyrin treatment did not result in an economic advantage for the spring and split N application treatments, the results indicate that for farmers who continue to fall-apply N, there is a positive economic return for using nitrapyrin. However, a number of other studies have found no significant yield effect nor economic advantage with the use of nitrapyrin (Hendrickson et al., 1978; Touchton et al., 1979; Blackmer and Sanchez,

1988). The variable results with nitrapyrin use are probably due to differences among factors affecting the microbial process of nitrification (e.g., climate and soil type). Further research should focus on quantifying those factors to determine when and where the use of nitrapyrin will be most beneficial.

Chlorophyll Monitoring

Other methods of monitoring N availability for corn have also been used to guide in-season fertilizer applications. In some studies, chlorophyll measurements have been shown to correlate well with N concentrations in the plant tissue and to have the ability to predict grain yield (Wood et al., 1992; Siambi et al., 1999). In Pennsylvania, Piekielek and Fox (1992) showed that the chlorophyll meter measures were as accurate as several soil N availability tests (NO_3 concentration of the surface 20 cm of soil at planting, ultraviolet absorbance at 200 nm of a 0.01 M NaHCO_3 extract of the surface 20 cm of soil at planting, and PSNT) in identifying N responsive and nonresponsive sites. However, the chlorophyll meter did not correlate well enough with soil N-supplying capacity (in-season mineralization) to accurately make N sidedress recommendations. Chlorophyll meter in combination with a sufficiency index was very successful as an in-season N management strategy for irrigated corn in Nebraska (Varvel et al., 1997a). The chlorophyll meter identified when additional N fertilizer was required, but it did not estimate the amount needed. A sufficiency index was calculated $[(\text{treatment/well-fertilized control}) \times 100]$, and in-season N fertilizer applications were made when index values were below 95%. Their N adjustment was to hand-apply 30 kg N ha^{-1} and water it within 24 h of when the chlorophyll meter measurements were taken and the sufficiency index fell below 95%.

Currently, it is unlikely that chlorophyll meter data alone will be sufficient to guide N fertilizer applications in subhumid regions. Factors other than N availability affect leaf chlorophyll content and can confound chlorophyll meter data, thus limiting its applicability as a N test (Piekielek and Fox, 1992). Piekielek and Fox (1992) pointed out that leaf chlorophyll content could be affected by any of the following: plant leaf chlorosis due to nutrient deficiencies other than N (i.e., K and Mg), disease, insect damage, cold temperatures, too high or too low plant populations on soils with marginal N availability, and recent N fertilizer application. These authors also stated that for soils with large organic N pools in areas of highly variable weather conditions, any test that measures plant N content at the six-leaf stage will not always be indicative of soil N availability for the remainder of the growing season nor always relate to final yield.

Excess soil water content is another major factor in subhumid regions with high organic matter soils that can confound developing relationships between chlorophyll meter data and plant N status. In the Upper Midwest, poorly drained prairie potholes comprise an appreciable amount of the total cropland area. Even with surface inlets to subsurface tile drainage lines, these areas often have periods during spring and early summer when wa-

ter is ponded on the surface. Under such conditions, many plant species often become chlorotic because of aeration stress rather than insufficient plant-available N (Hocking et al., 1987). Lizaso and Ritchie (1997) reported a three to seven times greater rate of senescence and loss of green leaf area in corn under ponded conditions compared with a control at the 12-leaf stage. Undoubtedly, denitrification and leaching can deplete NO_3 when water is ponded on these areas, but most of the pothole soils contain very high levels of organic N, which can become available to the plant through mineralization later in the growing season (Cambardella et al., 1994). This situation has been observed in the preliminary results from our watershed-scale N management research project (Walnut Creek N Initiative) where the prairie pothole soils have been unresponsive to additional N fertilizer (Dinnes et al., 1998) despite having lower corn leaf chlorophyll levels when measured at V9 to V12 growth stages (Dinnes et al., unpublished data, 2000). It also suggests that one reason for the success of the chlorophyll meter method under irrigated conditions is the absence of chlorosis from excess soil water and the fact that the soils are generally well drained and have relatively low SOM levels.

Insufficient soil water content can also confound interpretation of chlorophyll meter measurements. Leaves under water stress can have changes in the red and near-infrared (NIR) reflectances and because the chlorophyll meter computes the reading as the ratio of red/NIR, deviations away from normal will affect this ratio in the absence of N stress. Red and NIR reflectances change with age of the leaf, with the greater change in the NIR (Gausman et al., 1970). Observations of leaf reflectance from corn grown under N and water stress showed that NIR was more affected by water stress (Hatfield and Prueger, unpublished data, 2000). Changes in leaf reflectance with water stress have been known for several years, as shown by Thomas et al. (1971), who found leaf water content could be estimated through reflectance measurements. Chlorophyll meter readings can be affected by leaf water status, but the change induced would predict lower N requirements because the meter readings would increase under long-term water stress.

The chlorophyll meter has been successfully used in conjunction with the end-of-season basal corn stalk NO_3 test (Blackmer and Mallarino, 1997) to guide and improve future soil sampling for site-specific N fertilizer recommendations for irrigated corn (Varvel et al., 1997b). By combining both monitoring techniques, field areas with different soil N pools and N mineralization potential can be identified and sampled separately. The PSNT–LSNT program or any future soil tests developed to better predict N mineralization potentials could then be used to determine optimum fertilizer N rates for those areas. This approach could assist in creating a reliable variable N rate protocol for areas where field-scale variability in soil N content is very high (Cambardella et al., 1994).

Diversified Crop Rotations

Changing from continuous corn to a corn–soybean rotation has been shown to reduce NO_3 leaching al-

though the amount of reduction can be minimal, depending on climatic conditions (Randall et al., 1997a). With soybean, the leaching potential is reduced most when it is between growth stages V4 and R5, but leaching can be quite high during the early spring if large quantities of residual NO_3 remain following the corn crop. Significant NO_3 leaching can also occur following soybean (Randall et al., 1992; Baker and Melvin, 1994; Kanwar et al., 1996; Jaynes et al., 1999), especially if significant mineralization occurs when soils are not frozen during the late fall, winter, or early spring.

Including perennial legume or nonlegume crops in rotations has also been shown to decrease NO_3 losses. In Iowa, Baker and Melvin (1994) documented much lower NO_3 –N concentrations beneath alfalfa than for corn or soybean. Also, in Minnesota, Randall et al. (1997a) measured NO_3 –N concentrations in drainage water from alfalfa fields and Conservation Reserve Program (CRP) lands cropped to a mixture of alfalfa and perennial grasses and found they were 37 and 35 times lower than in drainage water from corn and soybean fields, respectively. They attributed the differences to longer growing seasons and greater annual evapotranspiration in fields with perennials because both of these processes contribute to greater N uptake and less drainage than in fields with only annual crops. Differences in fertilizer management between annual and perennial cropping systems also impact their relative NO_3 –leaching potentials. Typically, perennial cropping systems receive less tillage and N fertilizer than do annual cropping systems.

Factors contributing to differences in NO_3 –leaching potential for various crop rotations extend beyond fertilizer practices. Interactions between hydrology and tillage are very important because any residual NO_3 that accumulates in the soil profile, whether from N fertilizer or microbial processes, can be leached if it is not assimilated by microbes decomposing crop residue or taken up by another plant. When a crop such as alfalfa depletes profile water content and the amount of precipitation is not sufficient to fully recharge the profile, the leaching potential will be minimal and very little water will be moving into subsurface drainage lines. Differences in residue and root decomposition relationships, as well as soil–plant–water dynamics (i.e., soil water extraction capacity), among various plant species also influence the leaching potential (Baker and Melvin, 1994; Randall et al., 1997a; Malpassi et al., 2000). The rate of N cycling is important because although N-fixing legumes can release large quantities of N to soils over time, organic N derived from plant and microbial residues is not as rapidly available to plants as inorganic N provided by most commercial fertilizers. Additionally, the gradual release of organic N is often better synchronized with subsequent plant needs and microbial population dynamics than point-in-time applications of N fertilizers. The large flush of available N following an inorganic fertilizer N application can often supply more N than can be assimilated by plants and microbes. When this pool is nitrified, large amounts of NO_3 are susceptible to leaching and can potentially contaminate surface and ground

water resources. These processes are not limited to row crop areas because research also suggests that if excessive N is applied to either annual or perennial forage crops, water quality can be degraded by a resultant increase in soil NO₃ concentrations (Anderson et al., 1997).

Cover Crops

Cover crops have been shown to reduce the potential for NO₃ leaching from farm fields (Magdoff, 1991; Staver and Brinsfield, 1998) by mimicking natural ecosystems such as prairies where some plant species are growing whenever the ground is not frozen. They function by accumulating the inorganic soil N between main-crop seasons and holding it in an organic form, thus preventing it from leaching (Magdoff, 1991; Staver and Brinsfield, 1998). The N is subsequently released to the next crop as the cover crop residue decomposes. Cover crops also protect against soil erosion (Dabney, 1998; Kaspar et al., 2001), increase SOM (Reicosky and Forcella, 1998), and suppress weed growth (Buhler et al., 1998; Lal et al., 1991).

Desirable attributes for midwestern cover crops include the ability to establish rapidly under less-than-ideal conditions, grow vigorously despite cool temperatures and decreasing daylength, and not inhibit the growth of subsequent row crops. The biggest problem facing cover crops in this region is the short and generally cool growing season between harvest and planting of the subsequent row crop. In studies reviewed by Meisinger et al. (1991), cover crops reduced both the mass of N leached and NO₃ concentration of leachate 20 to 80% compared with no cover crop control. They also determined that grasses and brassicas were two to three times more effective than legumes in reducing NO₃ leaching. Rye (*Secale cereale* L.) has been used successfully as a cover crop in the northern Corn and Soybean Belt. But because rye overwinters, it must be killed or it can reduce the yield of subsequent corn crops by using too much water in the spring or immobilizing soil N (Munawar et al., 1990; Karlen and Doran, 1991; Tollenaar et al., 1993; Johnson et al., 1998). Other evidence suggests that the corn yield depression caused by the rye cover crop may be due to allelochemicals (Tollenaar et al., 1993). Further research may identify rye genotypes that do not release these compounds. Oat has been demonstrated to be an effective cover crop for this region because seed is easy to obtain and inexpensive, fall growth is more vigorous than rye, and it winter-kills, thus eliminating the need for herbicide or tillage in spring (Johnson et al., 1998).

Other plant species, including legumes, cereals, grasses, and brassicas, have been evaluated and used as cover crops, but these species have generally not been successful as true cover crops in the northern Corn and Soybean Belt. One exception was a study done in Wisconsin where hairy vetch (*Vicia villosa* Roth) and red clover grown in an oat-corn rotation showed a favorable economic comparison with continuous corn fertilized with 180 kg N ha⁻¹ commercial N fertilizer (Stute and Posner, 1995).

Tillage Impact on Nitrogen Cycling

Although tillage affects N cycling within soils, midwestern farmers generally do not intentionally use tillage to manage N. In more arid regions of the USA where fallow was traditionally used every second or third year, increased N availability following the fallow period was often more important than the perceived water conservation associated with that soil management practice (Haas et al., 1974). Tillage alters the soil environment by aerating the zone of disturbance and increasing the availability of O₂ to soil microorganisms. This favors different microbial species, populations, and processes than a nontilled soil (Doran, 1987). The net result of tillage is increased aerobic microbial activity, leading to elevated oxidation of SOM and mineralization of soil N (Randall et al., 1997a). This N mineralization response, often associated with preplant tillage, is also a benefit associated with using cultivation for weed control during the growing season.

Depending on tillage to release N for crop production is generally not a wise soil management practice. From a soil quality perspective, it reduces the benefits of SOM such as cation exchange capacity, soil structure, and water retention capacity merely for the release of plant-available N. It also exposes the soil surface to wind and water forces, thus increasing the potential for increased erosion (Reicosky et al., 1995). Furthermore, depending on seasonal weather patterns, temperature, and rainfall, tillage during autumn or early spring can cause N mineralization too early and increase the potential for NO₃ leaching before subsequent crops have an opportunity to assimilate the N provided by these processes.

Effects of tillage on N management have been demonstrated in studies comparing no-till with conventional tillage at several midwestern locations. In a long-term Minnesota study (Randall and Iragavarapu, 1995), residual soil NO₃ contents in the 0- to 1.5-m soil profile were significantly higher with conventional tillage than no-till for 5 out of 11 yr and were not significantly different for the other 6 yr. Average flow-weighted NO₃-N concentrations were 13.4 and 12.0 mg L⁻¹ for conventional and no-till corn production treatments, respectively. Furthermore, while the no-till treatment had 12% greater subsurface drainage flow than the conventional treatment, NO₃ losses were marginally greater (about 5%) with conventional tillage. Although insignificant, these results suggest a minimal trend toward greater NO₃ losses with conventional tillage in this study. The authors concluded that NO₃ losses through tile drainage depend more on growing-season precipitation than on tillage. Recently, Randall and Mulla (2001) concluded that NO₃ losses from agricultural fields are minimally affected by differences in tillage systems compared with N management practices.

In Iowa, Kanwar et al. (1993) monitored NO₃ leaching beneath both continuous corn and corn-soybean rotations managed using moldboard plowing, chisel plowing, ridge tillage, and no-tillage practices. The 3-yr average NO₃-N concentration in drainage water from continuous corn plots receiving moldboard tillage was signifi-

cantly greater (35.8 mg L^{-1}) than for either no-till or ridge tillage treatments (22.2 and 21.8 mg L^{-1} , respectively). Similar trends were observed for the corn-soybean rotation where the 3-yr average $\text{NO}_3\text{-N}$ concentrations in drainage water were 19.0 , 13.4 , and 13.9 mg L^{-1} for moldboard plow, no-tillage, and ridge tillage treatments, respectively. However, due to a higher volume of water moving through the soil with no-till and chisel plow, the 3-yr average $\text{NO}_3\text{-N}$ losses in subsurface drain flow were greater with no-till and chisel plow systems (61.2 and 64.3 kg ha^{-1} , respectively) than with moldboard plow (45.8 kg ha^{-1}) under continuous corn. Nitrate N load losses by tillage method for the corn-soybean rotation produced trends that differed from those of continuous corn. Under the corn-soybean rotation, greatest $\text{NO}_3\text{-N}$ losses were observed with chisel plow (32.1 kg ha^{-1}), followed by moldboard plow (27.5 kg ha^{-1}), no-till (23.9 kg ha^{-1}), and ridge tillage (23.7 kg ha^{-1}). These results led Weed and Kanwar (1996) to conclude that despite having an influence on N mineralization, tillage had less impact than crop rotation.

Plant Residue Management

Management of plant residues is directly related to tillage and affects N cycling in soils because plant tissue is a primary source and sink for C and N. Normally, when plant residues with C/N ratios greater than approximately 20:1 are added to the soil, available N is immobilized during the first few weeks of decomposition (Sinha et al., 1977; Doran and Smith, 1991; Somda et al., 1991; Green and Blackmer, 1995). Green et al. (1995) observed that incorporation of corn stover into soil resulted in rapid immobilization of all available inorganic N during the decomposition period. This occurred because the microbial population decomposing the plant residue had increased its biomass in response to the C source. If such immobilization occurs when plants need N for growth and development, the availability of NO_3 for the plants may be reduced. When this involves the primary crop, the immobilization process can have a negative economic impact by ultimately limiting yield. Eventually, as residue decomposition proceeds, the C/N ratio will begin to approach that of SOM ($\approx 10:1$ or $12:1$), microbial biomass will decrease, and N from plant residues that was incorporated in the microbial biomass will once again be released into the soil. However, if temperature, water content, or other factors slow the residue decomposition process, N may not be released from the plant residue or microbial biomass until the primary crop has matured and stopped assimilating N. This was hypothesized by Varvel and Peterson (1990) as a factor affecting continuous corn production when they estimated that 80% of the applied fertilizer N was still immobilized in crop residues, SOM, and microbial biomass at the end of the growing season. This emphasizes the importance of understanding all of the factors affecting plant residue decomposition and how they might be manipulated to reduce the potential for NO_3 leaching without decreasing availability of N to the primary crop or adversely affecting the soil C and N pools.

Disturbance of plant residue through tillage can significantly increase the decomposition rates (Douglas et al., 1980; Doran, 1987; Holland and Coleman, 1987; Aulakh et al., 1991). Additionally, root residues may respond differently to tillage or disturbance than shoot residues. For example, Martin (1989) observed that decomposition of root residues was more rapid and more complete when they were left undisturbed in the soil than when air-dried roots were mixed with moist or air-dried soil.

Even in the absence of disturbance, decomposition rates of root and shoot residue may differ. In a laboratory simulated no-till experiment, Gale and Cambardeila (2000) found that 75% of the new C inputs into soil after 1 yr of decomposition were root derived and 25% were shoot derived. They concluded that accrual of soil organic C associated with no-till is primarily due to the greater retention of root-derived C in the soil.

NITROGEN REMOVAL STRATEGIES

If agricultural management of N by all in-field means (e.g., crop rotations, cover crops, fertilizer application best management practices, and tillage) cannot satisfactorily reduce NO_3 concentrations, alternative strategies may be needed to remove NO_3 from subsurface drainage, shallow ground water, and/or surface water. Numerous methods for removing NO_3 from water have been identified. These include ion exchange, biological denitrification and assimilation, chemical denitrification, reverse osmosis, electrodialysis, and catalytic denitrification (Kapoor and Viraraghavan, 1997). Of these, only biological denitrification and assimilation seem suitable and adaptable for removing NO_3 from subsurface drainage in agricultural fields and watersheds. Multiple basic strategies that rely on biological denitrification and/or assimilation as either primary or secondary mechanisms for reducing NO_3 concentrations in shallow subsurface waters have been and continue to be researched. Several are discussed below.

Buffers

Research on effects of riparian buffers on shallow ground-water quality has been conducted in many areas across the USA and overseas. A consistent conclusion of research projects on riparian buffers is that the buffers decrease the NO_3 concentration of the shallow ground water flowing through them, in many cases to a dramatic degree, ranging from 48 to 100% (Peterjohn and Correll, 1983; Lowrance et al., 1984; Jacobs and Gilliam, 1985; Haycock and Pinay, 1993; Jordan et al., 1993; Hubbard and Lowrance, 1997; Verchot et al., 1997; Snyder et al., 1998; Addy et al., 1999; Rickerl et al., 2000; Spruill, 2000). However, there are differences among these studies as to the reported primary mechanisms responsible for these reductions.

One potential contributing factor to reduced NO_3 concentrations in shallow ground water is dilution. Precipitation that infiltrates in the buffer area can contain low NO_3 concentrations compared with the shallow ground water originating from the adjacent cropland.

Most studies have reported insignificant effects of dilution on the observed NO_3 concentration reductions, but there have been a couple of exceptions. In a study of various riparian forest management techniques on NO_3 reduction in the Georgia coastal plain, Hubbard and Lowrance (1997) found that dilution did play a role in reducing NO_3 concentration of the shallow ground water from a clear-cutting management treatment. Spruill (2000) found a 95% reduction in NO_3 concentration of young ground water in riparian buffer areas and estimated that 30 to 35% of the reduction was due to dilution effects. The remaining 65 to 70% reduction of NO_3 concentration was attributed to other processes.

Vegetative assimilation is one potential biological fate of NO_3 in shallow ground water as it passes through a riparian buffer. Forest trees, and the grass strips that are frequently placed between the trees and cropland, do assimilate NO_3 from the shallow ground water. It is the degree to which each vegetative group assimilates NO_3 that varies among the research studies. Lowrance (1992) measured a seven- to ninefold decrease in the shallow ground water NO_3 and NO_3/Cl ratio within the first 10 m of riparian forest buffer. He attributed this change in NO_3 concentration primarily to forest vegetation assimilation because the denitrification potentials of the soils at the typical depths of the water table were too limited to account for the dramatic losses of NO_3 . Hubbard and Lowrance (1997) also determined that vegetative assimilation was a significant sink of shallow subsurface-flow NO_3 . They reported that a grass buffer assimilated some of the NO_3 but that a downslope riparian forest was more efficient at assimilating NO_3 from shallow ground water. Other studies (Jacobs and Gilliam, 1985; Verchot et al., 1997; Addy et al., 1999) reported that vegetative assimilation did not play a significant role in reducing NO_3 concentration of shallow ground water. In these reports, denitrification was identified as the main loss mechanism.

Riparian buffers adjacent to cropland and placed in wet soils can provide several of the environmental factors that are essential to drive the processes of denitrification. Riparian buffers can supply the C required by the denitrifying bacteria, helping to fuel a biologically active zone within the soil profile. Denitrification rates have been found to be much greater within riparian buffers placed in soils with high water tables and organic C than in well-drained soils that are more aerobic (Jacobs and Gilliam, 1985; Ambus and Lowrance, 1991; Gold and Groffman, 1995; Addy et al., 1999).

An additional factor that influences denitrification is the residence time of the shallow ground water as it passes through the soil. Areas where flow of shallow ground water is relatively slow are the most effective at removing NO_3 because of increased residence time in anaerobic and biologically active zones (Lowrance et al., 2000). Snyder et al. (1998) observed the smallest degree of NO_3 reduction in a riparian buffer where the terrain was steep and ground water flow was rapid. They found the greatest NO_3 reductions in wet soils where the topographic gradient was very low and ground water movement was low.

Although riparian buffers have repeatedly been proven very effective at removing NO_3 from shallow ground water, any field tile drainage line that passes through a buffer and empties directly into a surface water stream will bypass all NO_3 remediation benefits of a riparian buffer. In order to utilize riparian buffer technology in tile-drained areas, alterations would be required. One option would be to terminate tile lines before entering a buffer zone. The tile flow would then seep through the buffer to the stream. Though effective, this strategy would severely limit the drainage capacity of the tile. A more appropriate option may be to discharge the tile effluent to a wetland retention area adjacent to the riparian buffer to treat the tile drainage before it enters the stream.

Wetlands

Natural and constructed wetlands have been used successfully as biological treatment systems for NO_3 removal (Gersberg et al., 1983; Crumpton et al., 1995; Romero et al., 1999). Eriksson and Weisner (1997) confirmed that epiphytic biofilms on submerged vegetation could remove NO_3 by denitrification. In a laboratory wetland microcosm experiment, Ingersoll and Baker (1998) determined NO_3 removal efficiencies at two temperatures (28 and 35°C), varying hydraulic-loading rates (5–20 cm d^{-1}) and C additions (1–6 g wk^{-1} dried plant residue) with NO_3 -contaminated water (30 mg L^{-1} $\text{NO}_3\text{-N}$). They measured NO_3 removal efficiencies from 8% to >95%, which decreased with increasing hydraulic-loading rates and increased with increasing C addition rates. Nitrate removal efficiencies increased as the C/N ratio increased to 5:1, and at C/N ratios > 5:1, NO_3 loss was nearly complete. One conclusion was that the denitrification rate constant directly depended on the C addition rate. Therefore, the authors suggested that a wetland could become more efficient at NO_3 removal if its plant growth were to be increased or if the plants were cut and the residue was left in place.

Crumpton et al. (1995) studied the efficacy of wetlands for removing NO_3 by assimilation and denitrification. They estimated that NO_3 draining from approximately 100 ha of land producing corn could potentially be removed by a 1-ha wetland. However, given an approximate 100:1 ratio of cropland to wetland area suggested by Crumpton et al. (1995), such a wetland may not provide sufficient residence time to remove an appreciable amount of NO_3 from drainage water during high rainfall events, which are typical for the Midwest during spring and early summer. Xue et al. (1999) came to similar conclusions in their study of the denitrification capacity of three wetlands in Illinois. Mean monthly NO_3 -loading rates to the wetlands were highly variable (4–451 kg N) due to a wide range in amounts of monthly precipitation. The ratios of denitrification capacity and mean NO_3 load ranged from 19 to 59% for the three wetlands, with an average of 33%. The authors estimated that for months of low inputs, almost all of the NO_3 could be denitrified, but for months of high inputs, only a small percentage would be removed. A conclu-

sion from their studies was, "The ultimate wetland denitrifying efficiency depends on both wetland capacity and the water residence time in each wetland."

Bioreactors

Substantial research has been done on designing bioreactors for denitrification (McCleaf and Schroeder, 1995; Reising and Schroeder, 1996; Shanableh et al., 1997). Most designs require a supplemental C source such as sucrose (Sison et al., 1995), ethanol [C_2H_5OH] or acetic acid [$C_2H_4O_2$] (Constantin and Fick, 1997), or methane [CH_4] (Thalasso et al., 1997) to be effective. They also require a high level of management for in-field or edge-of-field treatment of subsurface drainage water. Solid C sources have also been tested and would appear to be more amenable to field application. Volokita et al. (1996) used shredded newspaper as a C source in laboratory columns to obtain N removal rates ranging from 0.056 to 0.875 mg N g⁻¹ newspaper d⁻¹. Blowes et al. (1994) used a fixed-bed bioreactor filled with a compost mixture of sand, tree bark, wood chips, and leaves to treat drainage water from a farm field. Over a year, a 200-L bioreactor was able to remove nearly all NO₃ from a 10 to 60 L d⁻¹ discharge of field drainage water containing 3 to 6 mg L⁻¹ NO₃-N. Neither study differentiated between assimilation and denitrification as the NO₃ loss mechanism.

Robertson and Cherry (1995) demonstrated the NO₃ removal potential of a bioreactor constructed in situ. They filled a 0.6-m-wide trench that extended 0.75 m below a shallow water table with sand containing 20% (v/v) coarse sawdust and measured the concentration of NO₃ in ground water before and after flowing through the mixture. Very high NO₃-N concentrations (57–62 mg L⁻¹) were reduced to much lower concentrations (2–25 mg L⁻¹) in ground water passing laterally through the bioreactor. They attributed the removal of NO₃ to heterotrophic denitrification, with the sawdust serving as a labile C source, and estimated that this denitrification wall would have an effective lifetime of 20 to 200 yr. However, they offered only indirect evidence that denitrification was the primary removal mechanism, citing only reduced O₂ and SO₄ concentrations in water passing through the wall. In a similar study with a constructed denitrification wall, Schipper and Vojvodic-Vukovic (1998) found NO₃-N concentrations to be reduced from between 5 and 16 mg L⁻¹ to <2 mg L⁻¹ in shallow ground water passing through a wall. They attributed the NO₃ removal to denitrification and reported that denitrifying enzyme activity reached a plateau of 906 ng g⁻¹ h⁻¹ after 6 mo of operation. Placing C source denitrification walls around or near tile lines may help reduce NO₃-N concentrations of shallow ground water before it enters the tiles and discharges into surface waters.

Drainage Control Strategies

Other investigations have focused on methods to manipulate water table depth by altering the conventional designs of artificial drainage lines, using structures to

change the level of the drainage line outlet, or both. Such drainage control studies have used one or more of three basic strategies to reduce NO₃ contamination of surface waters. One strategy is to increase the anaerobic volume of the soil profile to enhance denitrification. A second is to decrease the amount of drainage water exiting the artificial drainage system. The third is to decrease the depth of the soil profile through which water infiltrates to reduce the leaching potential of soil NO₃. All three of these strategies can be accomplished by creating a shallower depth to the water table than exists with conventional, uncontrolled artificial drainage designs.

In Indiana, Kladviko et al. (1999) conducted a subsurface drain-spacing study where they installed drain lines at spacings of 5, 10, and 20 m, all at an average depth of 0.75 m. They found a consistent trend of greater loads of NO₃ removed with increasingly narrow drain line spacing during the 3 yr of study. Skaggs and Chescheir (1999) ran a simulation model for drain depths of 0.75 and 1.5 m and varied spacing for corn production on a sandy loam soil near Plymouth, NC. Their predictions showed that NO₃ losses could be reduced by a factor of more than 2.5 by placing drainage lines relatively shallow and close together. These predicted results are contrary to spacing effects reported by Kladviko et al. (1999), which may likely be due to the differing climates and soils of Indiana and North Carolina.

Jacinthe et al. (1999) used soil columns with added NO₃-N fertilizer (2.11 g column⁻¹ KNO₃-N) to simulate two water table management (WTM) techniques. The first WTM treatment (WTM1) had a static water table maintained at 0.5 m below the soil surface for 92 d, after which the water table was raised to 0.1 m below the soil surface for the next 18 d. The other WTM treatment (WTM2) simulated a dynamic water table where the water table was held at 0.5 m below the soil surface for 7 d and raised to 0.1 m for the next 4 d. The water table was then lowered gradually to the 0.7-m depth, held there for the following 4 d, and then raised back to 0.5 m and maintained at that level for the next 43 d. Before termination of the experiment, the researchers raised the water table back to the 0.1-m depth below the soil surface, holding it at that level for another 18 d. The WTM1 treatment removed 9 to 14% of the added NO₃-N during the 130-d simulation. The WTM2 treatment removed 24 to 43% of the added NO₃-N, and the researchers detected a faster rate of NO₃ removal when the water table was perched near the soil surface. Their conclusion was that NO₃ removal can be stimulated and enhanced by raising a water table into the upper soil profile layers, but the need for doing so during time periods conducive for denitrification—coinciding with a summer annual crop's growing season—could limit application of WTM practices due to possible damage to the crop.

An Iowa research project examined NO₃ transport in shallow ground water under two different WTM schemes (Kalita and Kanwar, 1993). One scheme maintained water table depths at 0.2, 0.3, 0.6, 0.9, and 1.1 m for time periods of 53 to 96 d after planting of corn for

1989 and 1990 and 45 to 97 d after planting in 1991. Samples of shallow ground water were taken from piezometers installed at depths of 1.2, 1.8, and 2.4 m for each water table depth treatment. In general, there was a trend of increasing $\text{NO}_3\text{-N}$ concentrations in ground water with increasing depth to the water table. The 1.2-m piezometer yielded average $\text{NO}_3\text{-N}$ concentrations varying from 7 to 2.5, 14.7 to 8.2, and 20.3 to 17 mg L^{-1} under shallow, medium, and deep water table depths, respectively. The second WTM scheme had water tables maintained at 0.3, 0.6, and 0.9 m from 50 to 53 d after planting to the time of harvest during the 3-yr study. Piezometers were again located at the 1.2-, 1.8-, and 2.4-m depths. The researchers found a trend of decreasing NO_3 concentrations in ground water with time during the growing season at all three sampling depths, with the lowest average NO_3 concentrations observed under the 0.3-m water table depth. Corn grain yields were negatively affected by increasingly higher water table levels at both sites for each year of the project. The researchers concluded that ground water NO_3 concentrations could be reduced by maintaining shallow water table depths and that the 0.3-m water table depth would provide the most beneficial water quality results. However, maintaining a water table at such a shallow depth would restrict the ability to produce high corn grain yields.

Fisher et al. (1999) compared a subirrigation with controlled drainage (SI-CD) treatment with a subsurface drainage treatment with no drainage control in a field-plot scale corn-soybean production system. The SI-CD treatment had a water table maintained at 0.4 m. The authors found that mean soil NO_3 concentrations were not affected by the SI-CD treatment at the 0- to 15- and 15- to 30-cm depths, but at the 30- to 75-cm depth, the SI-CD treatment reduced the 2-yr mean soil NO_3 concentration by 46% compared with the subsurface drainage treatment. Compared with the subsurface drainage treatment, the SI-CD treatment increased average corn N uptake by 13% and yield by 19% and increased average soybean N uptake by 62% and yield by 64%. The researchers concluded that proper implementation of a SI-CD management system could stabilize crop yield and N use efficiency and significantly reduce soil NO_3 concentrations deeper in the soil profile compared with subsurface drainage management, resulting in an overall reduction in NO_3 leaching.

Although drainage control methods have shown positive results in reducing NO_3 loading to surface waters from tile drainage, these technologies currently have substantial limitations. Drainage control is typically limited to landscapes with 1% slope or less due to the costs of drainage control structures (Evans et al., 1992; Shirmohammadi et al., 1992; Skaggs and Chescheir, 1999). At a 1% slope, there would be a 1-m depth to water table difference from the height of the drainage control structure to an upslope point 100 m along the drainage path. This may necessitate the need for multiple drainage control structures, even within a moderately sized field of 32 ha, to maintain a uniform water table depth. Even if structure costs were reduced, the

increased management time may make drainage control impractical on landscapes of steeper slope (Skaggs and Chescheir, 1999).

FUTURE RESEARCH NEEDS

Several methods for improving N management and reducing NO_3 contamination of water resources have previously been proposed by others (Kanwar et al., 1996; Randall, 1997). Among them are recommendations for (i) better use of soil tests to properly credit N sources other than commercial N fertilizers, (ii) abandoning tile and surface drainage systems, (iii) installing constructed wetlands or denitrifying ponds, (iv) implementing crop rotations that include perennial and cover crops, (v) improving the timing of N fertilizer application, and (vi) applying the proper rate of N fertilizer. The remainder of this review will focus on our perspectives of these suggestions and on some potential new directions for research and development.

Improved Monitoring of Soil Nitrogen for Split Nitrogen Application Programs

The logistics and time required for soil sampling and analysis in relation to the window of opportunity for fertilizer application has prevented widespread adoption of the PSNT-LSNT N management approach. A test is needed that can be performed earlier in the season and that relies on real-time meteorological and soil data to predict plant-available soil N status in late spring. This would allow for more timely soil sampling and would increase the window of opportunity for N application. One approach to developing a model to predict soil N mineralization from soil and meteorological data is to use soil respiration measurements as a surrogate for measurements of soil N mineralization (Parkin et al., 1996). Because soil N mineralization is the direct result of microbial activity, measurements of soil respiration without actively growing plants present are an integrated measure of soil microbial activity and are likely correlated with soil N mineralization. The advantage of using soil respiration measures as a surrogate for soil N mineralization measurements is that soil respiration can be measured almost continuously, responds quickly to changes in meteorological conditions, and can be used to determine cumulative activity of microorganisms over very short or very long time periods. This approach should be more amenable to determining the effect of meteorological conditions on soil N mineralization than direct measurements of changes in soil NO_3 concentrations, which must be taken at discrete points in time. Coupling respiration measurements with meteorological data and estimates of soil water movement should allow the prediction of soil NO_3 accumulation.

A model that predicts N mineralization based on temperature (growing degree days) has been developed (Honeycutt et al., 1988). However, application of the model requires a calibration procedure for the soil of interest. This calibration involves lengthy incubations of soil samples (30 to 60 d) to determine N mineralization potential, and thus precludes widespread adoption and

use. However, N mineralization potential is often not fully realized due to variations in climatic conditions. Soil respiration measurements may be a means of in situ calibration. Improving our understanding of N immobilization and nitrification processes in soils and N assimilation processes in plants would also facilitate development of predictive models based on interactions between local soils and weather patterns, and thus reduce or eliminate the need for PSNT-LSNT sampling and soil analyses.

Variable-Rate Nitrogen Application Models and Methodologies

Pierce and Nowak (1999) state that, "There appear to be no standards regarding the underlying agronomic principles that should be guiding the development and application of precision agriculture." At this time, there appears to have been little advancement in regard to this problem, which indicates that much research is needed in this area if precision farming techniques are to become adopted and successfully practiced by farmers. The nature and predictability of spatial and temporal variability within farm fields, at least at the local level, need to be identified to guide farmers as to which precision farming techniques may improve their profitability and protect the environment.

Improvements in long-range weather prediction models and incorporation of soil drainage and organic matter decomposition modules for various soil types are needed to create more reliable guidelines for variable-rate N management. Remote-sensing technologies that replace chlorophyll meters may play an integral role in establishing variable-rate strategies, but further research is needed to refine the technologies and their application for agricultural processes. To be effective, remote sensing must be able to correctly differentiate crop areas that are N deficient due to low soil N from all other conditions that may lead to chlorosis in plants, such as water-saturated soils, Mg and K deficiencies, or disease and insect infestations. Currently, it remains to be seen whether sensing across multiple wavelengths (hyperspectral remote sensing) can separate these factors and correctly identify those areas where soils are N deficient. If these areas cannot be accurately identified, simply using remote sensing to determine variable rates of N application is no guarantee against overapplication and, thus, an increased potential for NO_3 leaching and contamination of water resources.

If these challenges are met by science and technology in developing reliable variable-rate application methodologies, they should play a significant role in improving water quality, protecting the environment, and improving crop production efficiencies. Whether farmer economics are improved will depend on how much it will cost the farmer to adopt and learn the new methodologies.

Cover Crop Options and Management Strategies

Presently, there are no cover crop species or genotypes that integrate well with corn and soybean produc-

tion practices in the northern Corn and Soybean Belt. Cultural and plant-breeding research is needed to improve the compatibility of cover crops with current cropping systems. For example, rye cover crops grow well in the northern Corn and Soybean Belt, but rye genotypes that do not reduce corn yields need to be found or developed. A better understanding of the timing of cover crop N release from shoot and root residues as well as their interactions with tillage and surface conditions is needed. Agronomic research focusing on improved establishment of cover crops, reduced management costs, and decreased negative impacts on subsequent row crops is also needed. Achievements in these areas may be especially important as N management tools for both conventional and organic production systems.

Developing Perennial Cash-Cropping Systems

Based on the native prairie model, incorporation of more perennial crops into midwestern crop rotations could greatly reduce the amount of NO_3 that leaches into subsurface drainage lines. However, farmers and landowners will not adopt these crops until they have been proven to maintain or improve net economic returns. Research is desperately lacking in this area despite evidence that native prairies, forage crops, and Conservation Reserve Program land can substantially improve the quality of subsurface drainage water.

Public funding and use of government resources are vital to this area of research because currently there is very little economic incentive for private industry or commodity groups to support such efforts. Development of alternative cropping strategies, however, would be of very great benefit to the environment, farmers, and rural communities. Adoption of alternative and/or perennial crops could reduce farmer reliance on continuous corn and corn-soybean rotations, offer an opportunity for improved environmental quality, bolster prices of current commodities by reducing their overproduction, and reduce government spending on farm support and emergency relief programs.

Aside from the current social and environmental concerns of genetic modification, this recent technology could serve to vastly accelerate the development of both more suitable cover crops and perennial cash crops. Genetic modification could enhance nutrient contents of perennial crops or add pharmaceutical traits, making them more valuable commodities. Also, genetic modification could improve other traits of perennial and cover crops that would make it easier for farmers to incorporate them into their operations.

Nitrate Removal Strategies

If field-based N management fails to provide adequate protection against NO_3 contamination of subsurface drainage water, alternative strategies will have to be developed for treating the water before it enters surface waters. Unless subsurface drainage lines are blocked or cut, riparian buffers will not provide the protection needed because these drains often bypass

buffer areas and discharge directly into streams. Research will be needed to address a variety of limitations for wetlands, buffers, and bioreactors, including residence time for remediation and assimilation, available C sources, land area limitations, and costs of implementation. Also, because the predominant types of limitations will differ geographically, a suite of options will need to be developed rather than a single solution.

Research on drainage control or WTM strategies has produced some favorable water quality and crop yield results. The most promising appears to be systems that allow for dynamic control of the water table during the growing season to change the drainage level in response to precipitation events. However, these technologies are economically limited at this time to lands with little to no slope. Future research in this area needs to examine designs that are low cost and will allow for easy retrofitting of existing field tile drainage lines.

SUMMARY AND CONCLUSIONS

The most desirable outcome associated with research and technology development to minimize NO_3 contamination of water resources throughout the Midwest is a better public recognition that several different management practices will be required to ensure that surface and ground water resources achieve a quality acceptable to society as a whole. The problem is not solely caused by N fertilizer management nor any other single factor but is a combination of the soil management practices and inherent physical, chemical, and biological characteristics of the soil.

Several current and potential N management practices are being evaluated and used to reduce the NO_3 loading of subsurface drainage water, ground water, and surface water resources. Based on current experiences, a combination of two or more management practices may work in harmony to reduce NO_3 loss to field drainage, and thus make implementation of the practices more cost effective. One example would be to establish a perennial cash crop that could also function as a tile drainage line biofilter by growing the crop directly above subsurface drainage lines. The perennial crop may be able to remove NO_3 from the water as it flows toward the drainage lines. An income-producing biofilter such as this would reduce the farmer's costs for achieving compliance and accelerate adoption of management practices that more effectively mitigate NO_3 contamination of water resources.

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LETTER FROM THE EDITOR

As of 1 Jan. 2002, I succeeded Dr. Ken Barbarick as Editor of *Agronomy Journal* (AJ). During the 6 yr that Dr. Barbarick was Editor, AJ experienced many significant changes, possibly more so than at any time in the history of the journal. Dr. Barbarick's excellent leadership during this unprecedented period of change is noteworthy. These changes were necessary to address current demands and to prepare for anticipated needs.

While AJ has experienced many changes in recent years, more changes are on the horizon. Sometime this year, we expect to begin using Manuscript Tracker. This web-based software will allow authors, editors, and headquarters staff to access and input information appropriate to their responsibilities. With the use of Manuscript Tracker, manuscripts will be submitted and moved electronically through the steps of the review process. This will greatly speed the transfer of manuscripts from one person to another. Ultimately, everything else staying the same, the use of Manuscript Tracker will reduce the time it takes for manuscripts to go from submission to publication. Obviously, Manuscript Tracker does not have any control over manuscripts during the time when authors are making revisions on their papers. Typically, the longest time spent in the review and publication process is when authors are revising their manuscripts.

Manuscript Tracker will give authors new freedom in that they can follow the review of their manuscripts using the internet and they can check on the status of their manuscripts whenever they want. Editors will also be able to conveniently enter and maintain up-to-date review information for manu-

scripts assigned to them. Manuscript Tracker will be a major change in how we review and handle manuscripts, and we anxiously anticipate this exciting, new development. We readily see how valuable this software will be to our authors and editors. We do, however, anticipate a startup phase with Manuscript Tracker, and we ask for your indulgence during this time.

As I begin as Editor of AJ, I welcome your questions and comments about how we can better serve our authors and readers. You may communicate with me by telephone, internet, or letter, whichever is most convenient for you.

We encourage authors to submit manuscripts electronically. International authors are well served with electronic submissions by the savings they obtain in time and mailing costs. Authors who wish to submit manuscripts electronically should continue to send them to Dr. Robert Lascano at r-lascano@tamu.edu.

As of January 2002, paper-copy manuscripts should be submitted to me at the address below.

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