REVIEWS AND ANALYSES

Managing Farming Systems for Nitrate Control: A Research Review from Management Systems Evaluation Areas

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ABSTRACT

The U.S. Department of Agriculture funded the Management Systems Evaluation Area (MSEA) research project in 1990 to evaluate effectiveness of present farming systems in controlling nitrate N in water resources and to develop improved technologies for farming systems. This paper summarizes published research results of a five-year effort. Most research is focused on evaluating the effectiveness of farming system components (fertilizer, tillage, water control, cropping systems, and soil and weather variability). The research results show that current soil nitrate tests reliably predict fertilizer N needed to control environmental and economic risks for crop production. A corn (*Zea mays* L.)–soybean [*Glycine max* (L.) Merr.] rotation usually controls risk better than continuous corn, but both may result in unacceptable nitrate leaching. Reduced tillage, especially ridge-till, is better than clean tillage in reducing risk. Tile drainage controls nitrate in ground water, but discharge may increase nitrate in surface waters. Sprinkler irrigation systems provide better water control than furrow irrigation because quantity and spatial variability of applied water is reduced. Present farming systems have two major deficiencies: (i) entire fields are managed uniformly, ignoring inherent soil variability within a field; and (ii) N fertilizer rates and many field practices are selected assuming normal weather for the coming season. Both deficiencies can contribute to nitrate leaching in parts of most fields.

A FARMING system is defined here as an integrated set of farm management practices used for crop and livestock production. Generally, a farmer chooses a farming system based on the question, With the resources I have available, which farming system best controls risk, especially environmental and economic risk? Unfortunately, there is no one answer to the question because even adjacent farmers with similar soils and climates vary in the resources available for input into a farming system. Farmers vary in the availability of labor, capital, equipment, knowledge, production goals, managerial skills, landlord–tenant relationships, and perceived social pressures. Add to this the fact that the effectiveness of most farming practices is affected by numerous weather-related factors such as rainfall patterns, frost dates, growing season temperatures, flooding and droughts, and wind and hail storms. Weather factors influencing soil temperature and moisture greatly complicate nitrogen management because they affect nitrogen cycling, transformations, and movement, as well as final crop yield and nitrogen requirements (Westerman et al., 1999). Consequently, there are thousands of possible combinations of inputs into farming systems from which to choose. For these reasons, a farming system suitable for one farmer may not fit into the operation of an adjacent farmer.

The dominant basis upon which to select and develop a farming system should be how the farming system affects risks associated with net income and environmental quality (soil, water, and air quality in particular). Numerous research reports have indicated that current agricultural practices can result in nitrate pollution of water resources. Such pollution may be reduced or alleviated by changing to a more appropriate farming system that offers better water quality protection. Improved farming systems to protect and enhance environmental quality will become a greater need in the decades ahead. Most forecasts suggest that global population will increase and crop yields will have to increase correspondingly to meet food demands. Available land for a global food base is being offset by urbanization, transportation, erosion, and salinization. In order to achieve these future yield goals on a limited land area, the use of N fertilizers and animal manures will increase, compounding our present water quality problems. Increased N demand for higher yielding crops cannot be met by greater use of legumes because this would take land out of cereal production (Power and Papendick, 1985). Thus, there is a pressing need to improve our farming systems to alleviate further degradation of water quality.

To address these problems, the U.S. Department of Agriculture (USDA) initiated the Management Systems Evaluation Area (MSEA) research project in 1990. The purpose of the MSEA project was to evaluate the effectiveness of currently popular farming systems for managing nitrate pollution of surface and ground water resources and to develop new technologies that can be incorporated into farming systems to improve water quality (Onstad et al., 1991). The MSEA research project was the largest agricultural water quality research and demonstration project ever undertaken in the USA. It was led by the USDA Agricultural Research Service (ARS) and the Cooperative States Research, Extension, and Education Service (CSREES), in cooperation with

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Abbreviations: MSEA, Management Systems Evaluation Area.

various other federal agencies and some of their state and local counterparts. Major research sites for the project were located in Iowa, Minnesota, Missouri, Ohio, and Nebraska, with satellite locations in Kansas, North Dakota, South Dakota, and Wisconsin. These Midwestern sites were selected because this region produces 80% of the corn and soybean in the United States and uses more than 50% of the fertilizer nitrogen for corn in this country. A few research projects were funded in other states to support research on specific problems or unique situations. The general research approach used was outlined by Ward et al. (1994).

In the MSEA research, it was possible to compare only a very limited number of farming system components because of the large number of inherent variables. Consequently, there were comparisons at several locations of tillage methods, fertilizer practices, crop rotations (especially continuous corn versus corn–soybean rotations), irrigation and water management practices, and soil and water conservation practices. Likewise, many problems associated with variability in soils and weather were investigated. However, a discussion of wetlands as a practice to reduce nitrates is beyond the scope of this research review. Primary data collection focused on the effects of these variables on crop production and management of nitrates in surface and ground water. Most data were collected from 1991 through 1995. The purpose of this paper is to summarize MSEA research results on the effects of farming system components on agricultural water quality as documented in published results. The MSEA contribution provides insights into the potential positive and negative effects that farming system components may have on water quality and economic returns.

RESULTS OF MSEA RESEARCH

In order for nitrate leaching to occur, water-soluble nitrates must move with the water that percolates through the soil beyond the rooting depth. Consequently, keys to controlling nitrate leaching are to select a package of farming practices that best controls both nitrate accumulation within the soil and the movement of water through the soil. Both nitrate accumulation and water movement need to be synchronized with crop growth and crop N uptake in order to assure an adequate supply of both water and nitrates without excess accumulation of either. A large number of MSEA research projects were designed to assess the effectiveness of present farming practices and to develop management alternatives that provide better control of water and nitrates in soils.

Nitrogen Inputs into Farming Systems

This section includes MSEA research results on use of N fertilizer, animal manures, legumes, and atmospheric N inputs into farming systems.

Soil Testing

Numerous studies on N fertilizer application rate usually showed that both residual soil nitrates and crop yields could be maintained within desirable ranges by applying only the amount of fertilizer N needed. This amount can best be determined by implementing a good soil testing program (Bundy and Andraski, 1995; Ferguson et al., 1998; Kanwar et al., 1997; Rice et al., 1995a; Schepers et al., 1991a; Steinhilber and Meisinger, 1995). A good soil testing program must be calibrated to local soil and weather conditions (which differ even within a state) and must give proper credits for other N sources such as legumes and manures (Rice and Havlin, 1994). Soil N tests divided into preplant nitrogen tests (PPNT) and presidedress nitrogen tests (PSNT) have been reviewed previously (Power et al., 2000). At several locations it was observed that producers have reduced the average rate of N fertilizer application substantially during the last 10 yr (Hatfield et al., 1999a; Schepers et al., 1991b), due to the influence of effective educational and demonstration programs (Table 1).

Table 1. Frequency distribution of N fertilizer application rates onto field corn for the 1991 to 1994 period in the Walnut Creek watershed (central Iowa) (Hatfield et al., 1999a).†

	Year			
Rate	1991	1992	1993	1994
kg ha^{-1}				
0–112	0.08	0.11	0.16	0.15
113–168	0.49	0.42	0.34	0.35
>169	0.36	0.42	0.37	0.34

† Fractions do not total 1.00 because seed corn and popcorn fields were excluded.

Nitrogen Fertilizer Management

No research was conducted to directly compare N fertilizer products, although several studies examined forms of fertilizers used. Vigil et al. (1993) compared banded urea with broadcast urea and urea *supergranules*, with and without several nitrification inhibitors. (Supergranules have a spherical diameter of approximately 5–7 mm, in contrast to normal urea with a spherical diameter of approximately 2–3 mm.) They found that broadcast urea produced less corn grain than banded or supergranule urea. When nitrification inhibitors were used with the urea, results were not consistently positive. However, supergranules often slowed nitrification rates and reduced soil nitrate accumulations, while maintaining crop yields. Hughes and Kitchen (1993) investigated the effectiveness of slow release (sulfur-coated) forms of urea on corn production and soil nitrate accumulation. They found that slow-release urea was not effective in reducing residual soil nitrates or enhancing corn yields. Nitrification inhibitors (nitrapyrin) likewise failed to consistently enhance corn yield, water quality, or economic return (Randall et al., 1993c), but did increase nitrogen fertilizer use efficiency.

At several MSEA research locations it was demonstrated that placing the N fertilizer band on the shoulder of the ridge in a ridge-till system also reduced water percolation through the band, reducing nitrate movement below the root zone (Dolan et al., 1993; Lowery et al., 1995; Bargar et al., 1999; Jaynes and Swan, 1999),

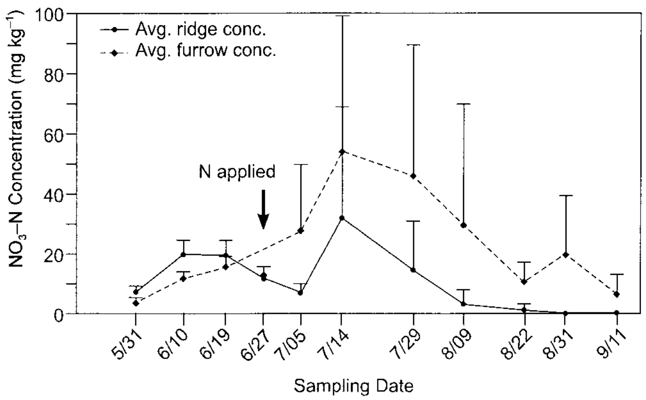

Fig. 1. Average nitrogen concentration below the root zone comparing nitrogen fertilizer placement in the shoulder and in the furrow of a ridge-till system (Dolan et al., 1993).

as illustrated in Fig. 1. With furrow irrigation, running water through every other furrow while applying N fertilizers in the nonwatered furrows reduced nitrate leaching while maintaining corn yields (Martin et al., 1995; Watts and Schepers, 1995).

Several investigators studied time of N fertilizer application. Normally, about 60% of the fertilizer N applied to corn in the Corn Belt is applied in the fall as anhydrous ammonia. In those years when wet weather is encountered during fall months or if harvest is unusually late, more anhydrous ammonia is applied in the spring. In some regions where ground water nitrate concentrations are relatively high, state or local regulations may require that any anhydrous ammonia applied before soil temperatures fall below 10°C must be applied with a nitrification inhibitor. Fall-applied N fertilizer can result in more leaching than spring applications (Olson et al., 1964; Aldrich, 1984). Effects of preplant versus side-dressing N fertilizers were also investigated. Generally, if adequate N was applied, there was little difference between these two times of application (Kitchen et al., 1998; Randall et al., 1993a,b). However, Kanwar and Baker (1993) found more nitrates within the upper 1.5 m of soil for preplant application compared with split applications of N.

Crop Monitoring

The MSEA research resulted in the development of a new technology for determining proper N fertilizer rate, based on greenness of the corn crop. This approach was investigated extensively (Blackmer et al., 1994; Blackmer et al., 1996a,b; Schepers et al., 1996; Varvel et al., 1997b). In this procedure, only a fraction of the anticipated fertilizer need is applied before planting, except for a few strips through the field that receive adequate fertilizer N. A review of the crop greenness technology from MSEA research is covered in a previous review paper (Power et al., 2000).

Animal Manure Nitrogen

There were many MSEA investigations of the effects of animal manures as a source of N for corn production. Animal manures were an effective source of N for crop production (Francis et al., 1995; Kanwar et al., 1995; Randall et al., 1993a). However, because the N in manures must first be mineralized, a process that is dependent on soil and weather conditions, it is more difficult to control soil nitrate levels in manured soils than in those receiving fertilizer N (Kanwar et al., 1995). Consequently, controlling potential nitrate pollution of water is much more difficult when using manures as a lone N source. A significant finding of this research was that high rates of manure applications could markedly increase soil nitrate concentrations for 10 yr or longer (Kitchen et al., 1995a; Pierce et al., 1995). This illustrates the prolonged effects of high manure application rates on potential degradation of water quality. Another reason that leaching potential is greater in manured soils is because manuring tends to enhance earthworm populations and subsequently increase soil hydraulic conductivity (Gupta et al., 1995).

Legumes as a Nitrogen Source

In the MSEA research, three legume sources of N were investigated. These were soybean, alfalfa (*Medicago sativa* L.), and various legume cover crops, most notably hairy vetch (*Vivica villosa* L.). Klocke et al. (1999) concluded that present soybean N credits (approximately 40 kg N ha^{-1}) used in the algorithms developed for soil testing in Nebraska may underestimate the potential N contribution of a soybean crop to the following corn crop. As a result, in their lysimeter experiments, annual nitrate leaching from the corn–soybean rotation averaged 96 kg N ha^{-1}, compared with 59 kg N ha^{-1} for continuous corn. However, they found that the soybean crop was effective in reducing residual nitrates. Similar results were reported by Iragavarapu et al. (1993) and Albus and Knighton (1998). Kanwar and Colvin (1995) and Kanwar et al. (1995) measured almost twice as much nitrate N removed in tile drain discharge under corn compared with soybean. However, Varvel et al. (1997b) found greater nitrate N concentrations under soybean than under corn, whereas Omay et al. (1997) obtained variable results with several different soil types. Climate and soil type where the research was conducted can be used to explain apparent outcome differences.

Alfalfa was an excellent scavenger crop, capable of removing almost all nitrates to several-meter soil depths within a year or two (Watts et al., 1997). Crops grown after alfalfa need little or no additional N for at least a year afterward. Residual effects of alfalfa can be measured for at least 5 yr (Lamb et al., 1998). Likewise, red clover (*Trifolium pratense* L.) in a corn–soybean–wheat (*Triticum vulgare* L.)–red clover rotation provided sufficient N for the following corn crop and helped reduce soil nitrate levels (Subler et al., 1995b).

Atmospheric Nitrogen Depositions

Nitrogen is also added to farming systems in precipitation. Hatfield et al. (1996) conducted extensive studies on this process in central Iowa. They found that N deposition as nitrates in precipitation was uniformly distributed over a watershed, averaging about 12 kg N ha^{-1} yr^{-1}. Atmospheric N deposition as nitrate was closely related to precipitation amount, both on a monthly and annual basis. Thus, in central Iowa, nitrate N deposition was greatest during the summer months. On a whole-watershed basis, accounting for land used for purposes other than corn production, N in precipitation amounted to about 25% of all forms of N added to the watershed. Average nitrate N concentration in precipitation was 1.5 mg L^{-1}, whereas average nitrate N concentration measured in precipitation in Missouri was 0.63 mg L^{-1} (Kelly and Blevins, 1993).

Another potential source of atmospheric N is deposition as ammonia. Harper and Sharpe (1993) showed that the quantity of atmospheric ammonia N absorbed by a corn crop varied greatly, depending on plant growth stage, soil N availability, weather, and possibly other factors. More than 40 kg ammonia N ha^{-1} may be volatilized into the atmosphere from corn leaves during the maturation process (Francis et al., 1993, 1997).

Effects of Production Inputs on Nitrogen Transformations and Movement

Included in this section are the effects of the soil management and crop production practices used in farming systems on N transformations and movement and their effects on crop yields and water quality.

Tillage Effects

At most MSEA research locations, studies were conducted to compare the effects of different tillage practices on N transformations, crop production, and nitrate leaching potential. As a result of tillage, early season N mineralization was hastened, nitrates frequently accumulated in the soil, and essentially all the carbon contained in residues from the previous crop was lost as carbon dioxide. In contrast, for mulch and no-till practices, soil temperatures and water evaporation rates were reduced and the lack of intimate contact between residues and the soil slowed the decomposition rate of residues. These changes (Table 2) resulted in slower N mineralization and nitrification and greater retention of soil C, N, and water for reduced and no-till soils (Karlen et al., 1998a; Katupitiya et al., 1997; Mankin et al., 1996; McCarty and Meisinger, 1995; Pierce et al., 1995). As a consequence, residual nitrates in the soil after harvest were often greater for clean tillage than for reduced or no-till methods, increasing the potential for nitrate leaching into ground water. In contrast, on deep loess soils in Iowa, Kramer et al. (1990) measured greater nitrate discharge in base flow to streams draining a ridge-till watershed than for one clean-tilled. They attributed this difference to greater water infiltration and less runoff over a 20-yr period for ridge-till compared with clean tillage.

Effects of tillage system on corn grain yield were variable (Kanwar and Baker, 1993; Kanwar et al., 1997; Katupitiya et al., 1997; Lowery et al., 1998; Randall et al., 1993a). Reduced and no-till methods often showed an advantage when crops experienced heat or water stress.

At a number of locations the effects of ridge tillage were compared with those of other tillage methods in respect to N transformations and nitrate leaching potential. Generally, ridge-till systems were effective in controlling soil erosion (especially when practiced on the contour) and, when combined with other best management practices, slowed the rate of nitrification and nitrate movement through the soil (Karlen et al., 1998a; Lowery et al., 1998). Kanwar et al. (1997) showed that nitrate concentrations in tile drainage were generally less for ridge-till and no-till methods than for clean tillage, indicating that nitrates moved through reduced-tillage soils at a slower rate. Clay et al. (1994) measured less nitrate leaching when anhydrous ammonia was banded in the ridge shoulder that when placed in the valley between ridges. If the injector knife slot in the valley position remained open, more water would infiltrate through the fertilizer band, increasing nitrate leaching potential.

Hatfield et al. (1998a) concluded that in general, ridge-till systems reduced agrichemical leaching and ad-

Table 2. Changes in surface C and N content after 11 and 15 yr of moldboard plow, chisel plow, ridge tillage, or no tillage for continuous corn production (Karlen et al., 1998b).

	Soil organic carbon			Δ Carbon		C to N ratio		Δ Nitrogen	
Tillage	1977	1988	1992	1977–1988	1977–1992	1988	1992	1977–1988	1977–1988
	g kg^{-1}							kg ha^{-1}	
0–5 cm									
Moldboard	15.5a†	19.2b	23.7c	3.6b	8.2c	12.4b	12.3a	182b‡	416c
Chisel	17.4a	23.2ab	29.1bc	5.8ab	11.7bc	13.2ab	13.2a	275ab	565bc
Ridge tillage	17.3a	24.9a	32.9ab	7.6a	15.6ab	13.8a	12.6a	345a	771ab
No tillage	17.6a	24.0a	37.3a	6.4ab	19.7a	13.4ab	13.2a	300ab	932a
5–10 cm									
Moldboard	18.0a	19.2b	23.6a	1.2a	5.5a	13.4a	11.9a	58a	302a
Chisel	19.6a	22.3a	28.9a	2.7a	9.3a	13.1a	12.8a	136a	487a
Ridge tillage	18.8a	20.6ab	25.5a	1.8a	6.7a	13.6a	12.7a	85a	352a
No tillage	18.0a	19.6ab	26.0a	1.6a	8.0a	13.4a	12.0a	77a	433a
10–20 cm									
Moldboard	16.0a	19.9a	22.3a	3.9a	6.3a	13.3a	12.1a	394a	702a
Chisel	16.9a	20.9a	26.0a	4.1a	9.2a	13.4a	12.7a	410a	1000a
Ridge tillage	16.1a	18.6a	23.3a	2.5a	7.2a	13.0a	12.4a	263a	808a
No tillage	16.2a	18.2a	22.9a	2.1a	6.7a	12.9a	12.1a	220a	751a

† Values for each depth and year followed by the same letter are not different at the 0.05 probability level.
‡ Values computed using soil bulk densities of 1.25, 1.30, and 1.35 Mg m^{-3} for the 0- to 5-, 5- to 10-, and 10- to 20-cm sampling depths, respectively, based on Logsdon et al. (1993) and USDA-NRCS (1995).

verse effects on water quality. Lowery et al. (1998) found that the amount of nitrate leached was reduced with increased time between fertilizer application and the next rainfall or irrigation event. However, Karlen et al. (1998a) stated that ridge tillage with excess N application and for certain cropping systems will not reduce potential for water degradation.

Reduced and No-Till Systems

The MSEA research studies showed that no-till systems reduced runoff and increased preferential flow (due to old root channels and increased earthworm activity), resulting in greater water infiltration from rains and irrigation compared with other tillage systems. This caused greater percolation of water through the soil and consequently greater nitrate movement to deeper soil depths (Eisenhauer et al., 1993; Kanwar et al., 1997; Kranz and Kanwar, 1995; Pierce et al., 1995; Subler et al., 1996). Kanwar and Baker (1993) found fewer nitrates for no-till compared with plowed soils to the 1.5-m depth, and Eisenhauer et al. (1993) found similar results for the 2- to 12-m depth. Kelly and Blevins (1993) also found fewer nitrates in no-till than in plowed soils. From soil samples collected in May from plots that had been no-tilled, chiseled, or plowed for 12 yr, Karlen et al. (1994) demonstrated greater nitrate concentrations in the upper 30 cm of no-till soils, but considerably lower nitrate concentrations below 180 cm. Kanwar and Colvin (1995) showed that nitrate N concentrations in tile discharge from no-till soils were less than those from chisel-plowed soils. However, others found no consistent effects of tillage on nitrate leaching (Kanwar et al., 1995; Lamb et al., 1998; McCracken et al., 1995; Pierce et al., 1995), probably because tillage affected both nitrate concentrations and quantity of leachate. Kitchen et al. (1998) concluded that variability in soil properties and weather conditions had a much larger effect on nitrate leaching than did tillage system.

Water Management

A number of farming practices can be implemented to achieve control over availability and quantity of water running off or leached, even though the producer has no control over precipitation. The practices include tillage method (affecting runoff and evaporation rates), tile drainage (preventing excess water from saturating the soil), water table control (affecting evaporation rate, leaching, and rooting depth), and irrigation practices (reducing plant water stress).

Tillage Effects

Tillage practices can affect soil nitrate concentrations in surface and ground waters by many means, both directly and indirectly. The direct effects of tillage on soil nitrate accumulations and movement will be discussed in more detail later in this paper. Indirect effects of tillage practices are caused primarily by the effects of tillage on the hydrological cycle in a field. In general, reduced and no-till practices, compared with clean tillage, reduce rate of water evaporation from a soil, reducing runoff and increase soil water storage. Future precipitation events often result in greater water percolation and potential nitrate leaching for reduced and no-till fields. However, nitrate concentrations in soil water for reduced and no-till systems are often less than for clean tillage. Total nitrate N leached may be greater, less, or the same when comparing tillage systems. Frequently, however, nitrate concentrations in base flow from reduced and no-tilled fields are usually less than from clean tilled fields. This sequence of events was documented in several MSEA projects (Karlen et al., 1998b; Kramer et al., 1990; Steinheimer et al., 1998). For claypan soils in Missouri, Blanchard et al. (1995) reported that little nitrate N was found in baseflow for Goodwater Creek because it appeared that most of the nitrates in ground water feeding springs for base flow were denitrified before entering the creek. Generally, direct nitrate loss from a field by runoff is relatively minor for all systems (Alberts et al., 1993). In western Iowa, Steinheimer et al. (1998) calculated that less that 1% of the fertilizer N applied could be accounted for in runoff, particularly during snowmelt. Soenksen et al. (1994) measured annual nitrate N losses of 8.0, 0.06, and 7.9 kg N ha^{-1} for streamflow (largely tile drain discharge), stormflow (runoff), and base flow, respectively, for the Walnut Creek watershed in Iowa.

Tile Drainage

More than 30% of the cropland in the Midwest is tile drained (Hatfield et al., 1998b), a practice that has profound effects on soil hydrology and nitrate removal because most of the nitrates passing through the root zone are intercepted and moved eventually as discharge into surface waters. A large part of the nitrates found in surface waters in that region originate from tile drain discharge. It is estimated that at least 95% of the percolating nitrates are intercepted, usually preventing a large accumulation of nitrates in ground water beneath tile drained fields (Hatfield et al., 1999a). Kanwar and Colvin (1995), Kanwar et al. (1997) and Karlen et al. (1998a) found that the quantity of nitrates removed by tile drainage in Iowa was affected by cropping system and tillage practices. Over 3 yr, nitrate N removed in tile drains averaged from 24 to 63 kg ha^{-1} yr^{-1}, depending on what production practices were used (Kanwar and Baker, 1993). Randall et al. (1993b) showed that quantity of nitrate N removed in tile discharge was also closely related to annual rainfall, ranging from less that 10 kg N ha^{-1} in one year up to 170 kg N ha^{-1} the next year. Their data clearly demonstrate much lower losses of nitrate with perennial cropping systems (Table 3). Over a 4-yr period in central Iowa, Jaynes and Swan (1999) provided information on nitrate discharge from tiled watersheds by measuring nitrate discharge from tile drains, county drains, and streams in the 5130-ha Walnut Creek watershed. They found average annual nitrate N discharges of 24 kg N ha^{-1} for tile lines, 35 kg N ha^{-1} for county drains, and 30 kg N ha^{-1} for streams, compared with 30 kg N ha^{-1} for the river into which the streams discharged (Table 4). In this same watershed, Eidem et al. (1999) estimated that annual recharge of

Table 3. Nitrate N concentration and loss in tile drainage as affected by cropping system (Randall et al., 1993b).

Year	CC	C-Sb	Sb-C	Alf	CRP
	Cropping system†				
	NO_3–N concentration, mg L^{-1}				
1990	30	22	26	–	–
1991	39	29	38	4.1	3.9
	NO_3–N loss, kg ha^{-1}				
1990	6	4	7	0	0
1991	170	79	81	1.6	1.7

† CC, continuous corn; C-Sb or Sb-C, corn–soybean rotation with current crop underlined; Alf, alfalfa; CRP, Conservation Reserve Program, unharvested grass crop.

the water table required 5 to 34% of the annual precipitation received and that nitrate N concentrations in ground water below the tile drains were often more than 10 mg L^{-1}, especially in the spring. Cambardella et al. (1999) found little relationship between N fertilizer rate and nitrate removal in tile drainage. Large differences in rainfall amounts could be used to explain differences in nitrate losses (Table 5). Cambardella et al. (1999) concluded that nitrate N losses to subsurface drainage occur primarily because of asynchronous production and uptake of nitrates in the soil and the presence of large quantities of mineralizable N in soil organic matter.

Drainage and Subsurface Irrigation

In Ohio and elsewhere, it has been demonstrated that by using a combination tile drain–subsurface irrigation system, it is possible to control water table depth throughout the season (Fausey et al., 1995). With such a system, Cooper and coworkers (Cooper et al., 1991, 1992) obtained 3-yr average soybean yields for five cultivars of 5390 kg ha^{-1} when water tables were maintained at an average depth of 39 cm and with soybeans planted in 18-cm rows. Controlling water table depth resulted in 58% greater yields, compared with no irrigation. Few nitrates were discharged from the tile lines. Desmond et al. (1996) concluded that the computer model ADAPT adequately simulated soybean production and hydrology under such conditions. In soil columns, Jiang et al. (1997) demonstrated similar effects of controlled water tables on nitrate movement following simulated rains.

Claypan Soils

The unique hydrology of the paleosols (claypan) soils presents special challenges in maintaining water quality

Table 4. Total loss of NO_3–N in drainage water from tiled watersheds and streams located within a 5130-ha watershed, 1992 to 1995 (Jaynes and Swan, 1999).

Year	110 Tile	210 Drain	220 Drain	230 Drain	310 Stream	320 Stream	330 Stream	440 River
	NO_3–N loss — Site # and type of drainage							
	kg ha^{-1}							
1992	26	8	26	17	28	30	28	5
1993	51	52	57	59	67	66	66	83
1994	5	6	4	6	6	5	4	12
1995	13	22	23	23	21	24	19	19

(Kitchen et al., 1998; Blanchard et al., 1995). These soils crack extensively (2 to 6% by volume) upon drying (Baer et al., 1993), creating ample opportunity for preferential flow when precipitation events occur. By this means, nitrate N on or near the soil surface can be rapidly moved into and through the subsoil. However, once in the subsoil, these nitrates are trapped in a very high clay environment with extremely low hydraulic conductivity. Also, because soil cracking is more or less random, spatial variability in nitrate movement is great.

Irrigation

Use of irrigation also offers special challenges in controlling water quality. Gravity or furrow irrigation is commonly employed because of the low cost of the equipment required. Lack of uniform gradients down a furrow as well as natural variability in water infiltration rates result in spatial variability in water infiltration. In addition, length of time during which water runs down the furrow is usually greater for the upper than the lower ends of the furrow, causing further variation in water infiltration. If the lower end of furrows is diked to prevent runoff water, distribution in the soil is further complicated. As a consequence, water and nitrate distribution from one end of the furrow to the other is highly variable (Eisenhauer et al., 1993; Katupitiya et al., 1997; Watts and Schepers, 1995). Use of recently developed surge irrigation techniques somewhat reduced the quantity of water applied and variability in infiltration rates (Eisenhauer et al., 1993; Schepers et al., 1995; Watts et al., 1997).

By using various sprinkler irrigation techniques, the quantity of water applied can be reduced and uniformity of application improved. For example, Watts et al. (1997) recorded average annual water application rates of 606 and 175 mm for furrow and center-pivot sprinkler irriga-

Table 5. Application rates, concentrations, and losses of NO_3–N in subsurface drainage from an 8.9-ha drainage area in a 38-ha field cropped to corn in 1992 and 1994 and to soybean in 1993 and 1995 (Cambardella et al., 1999).

Year	Drainage	Applied	Total loss	GS† loss	NGS‡ loss	Total concentration	GS concentration	NGS concentration
	mm	kg ha^{-1}				average mg L^{-1}		
1992	240	178	26.3	9.8	16.5	10.6	10.8	10.4
1993	833	0	51.3	35.4	15.9	7.1	6.7	7.5
1994	63	156	4.9	1.1	3.9	8	8.5	7.5
1995	140	0	13.3	4.3	9	9.4	11.8	7.6

† Growing season (June–October).
‡ Nongrowing season (November–May).

Table 6. Irrigation system comparison for water application rate, rainfall, soil nitrogen, fertilizer, and yield data over five years (Watts et al., 1997).

Irrigation system	Irrigation applied	Rainfall†	Rain + irrigation	Residual N‡	Starter N	N fertilizer	Irrigation NO_3–N	Irrigation N	Grain yield
	mm			kg ha^{-1}			mg L^{-1}	kg ha^{-1}	Mg ha^{-1}
Conventional	606	445	1051	100	24	196	30.8	228	11.91
Surge flow	211	445	656	107	24	135	28.9	78	11.49
Center pivot	175	445	620	75	24	122	29.4	57	11.18

† 1 May to 30 September.
‡ Residual nitrate N to a soil depth of 0.9 m.

tion, respectively, in Nebraska. Also, because of the better water control provided by sprinkler irrigation, average fertilizer N rate was also reduced from 196 to 122 kg N ha^{-1} (Table 6). Schepers et al. (1991b) reported that improved N fertilization and irrigation practices reduced ground water nitrate concentrations on about 4000 farms in central Nebraska. Quantity of nitrates leached generally declined as amount of water applied declined (Lowery et al., 1998). Francis and Schepers (1993) showed that water and fertilizer management affected the efficiency with which nitrates in the irrigation water were used by the crop. Interactions of tillage system with irrigation methods often resulted in variable results in regard to nitrate and water movement (Katupitiya et al., 1997; Lamb et al., 1998; Lowery et al., 1998). Albus and Knighton (1998) found that soil nitrate levels and nitrate leaching were both greater for the first few years after initiating irrigation practices on previous dryland. This probably resulted from both leaching of residual nitrate N from dryland subsoils and by enhanced mineralization of soil organic N.

Cropping System and Rotation Effects

The most common comparison of cropping systems made in the MSEA studies was continuous corn versus a corn–soybean rotation. At a few locations, rotations including wheat, alfalfa, or other legumes were studied. The few economic analyses (Table 7) made in MSEA projects generally indicated that at most locations the most profitable cropping system was a corn–soybean rotation (Batte et al., 1998; Prato et al., 1995).

Continuous Corn versus Corn–Soybean Rotation

General results of comparisons of continuous corn to corn–soybean rotations indicated that the amount of nitrate that leached was more closely related to other crop production practices than to cropping system. Depending on what other production practices were used, results could vary widely. Even though usually half or less fertilizer N was applied in a corn–soybean system than with continuous corn, concentration and quantity of nitrates in leachate varied greatly for both systems. However, MSEA results did show that the corn–soybean rotation could under proper circumstances significantly reduce nitrate leaching (Randall et al., 1993a). Also, corn yields were often about 10% greater for the rotation than for continuous corn (Kanwar et al., 1997; Lamb et al., 1998; Omay et al., 1998; Randall et al., 1993a; Varvel et al., 1997a).

Table 7. Estimated average values for soil erosion loss, nitrate N in leachate, and net income for four farming systems (adapted from Batte et al., 1998), according to the EPIC model.

	Continuous corn	Corn–soybean	Corn–soybean–wheat
Soil loss, Mg ha^{-1}	1.6	1.3	0.6
Nitrate in leachate, mg L^{-1}	16	9.7	4
Net income, \$ ha^{-1}	48.87	99.82	53.8

Generally, a package of practices were required to manage nitrate leaching, often including ridge-tillage, N fertilization according to well-calibrated soil tests, split application of N fertilizers (with knife-injection slots covered), and water control, as well as a corn–soybean rotation (Karlen et al., 1998a; Lamb et al., 1998). Kanwar et al. (1997) concluded that with the N fertilizer, tillage, and water control practices (tile drainage) normally used in Iowa, continuous corn was not an environmentally acceptable cropping system in terms of maintaining water quality. Randall et al. (1993a) measured considerably more nitrate removal in tile drainage from continuous corn than from corn and soybean. With furrow irrigation in Nebraska, Katupitiya et al. (1997) observed greater nitrate leaching for continuous corn than for corn–soybean, particularly when disk-plant systems were used (in contrast to ridge-till or slot plant). (Rotation research conducted here showed that a corn–soybean rotation usually controls risk better than continuous corn but both may result in unacceptable nitrate leaching.) Omay et al. (1997) found that inclusion of soybean in rotation with corn improved fertilizer N recovery for two soil types and helped to maintain the mineralizable N pool. Using ^{15}N isotopes on these same soils, Rice et al. (1995b) found little difference in ^{15}N remaining as nitrates after harvest but found larger quantities of ^{15}N immobilized in soil organic matter of soil cropped to the rotation than of soils cropped to continuous corn. Varvel et al. (1995) measured 28% less water percolation and 19% less nitrate leaching for corn and soybean than for continuous corn.

Major factors affecting the impact of cropping system on residual soil nitrates include selection of crop yield goals and N credits given to soybean when included in a cropping system. Schepers et al. (1991a) found that more than 50% of corn producers in the Central Platte Valley of Nebraska set yield goals that exceeded actual yields by 10% or more. As a consequence, fertilizer N rates based on yield goals were greater than required, increasing nitrate leaching potential.

A question that has not been adequately answered relates to nitrogen credits for soybean. In most states,

Table 8. Distribution of ^{15}N from soybean residue after corn harvest (adapted from Omay et al., 1998).

Location	Eudora loam†	Crete silt loam‡
	% of ^{15}N	
Corn crop	3	16
Soil inorganic forms	4	1
Microbial biomass	3	2
Soil organic N	84	66

† Coarse-silty, mixed, superactive, mesic Fluventic Hapludoll.
‡ Fine, smectitic, mesic Pachic Argiustoll.

fertilizer recommendations usually suggest reducing N fertilizer rates by about 40 to 45 kg N ha^{-1} when corn follows soybean. While experience has proven that this is a reasonable average figure for soybean N credits, it appears that there may need to be considerable variation in this value, depending on soil type, growing conditions, and other factors. A good soybean crop may return as much as 100 kg N ha^{-1} in the soybean residues. Much of this N is contained in soybean leaves, which are high in N and fall to the ground in early fall when soil temperatures are still warm. Often, visual evidence of such leaf fall has disappeared before winter arrives. Also, appreciable quantities of N may reside in soybean root nodules, which also shed and decompose rapidly during maturation. Presumably, much of the N in the leaves and root nodules is rapidly mineralized and shows up in the residual nitrate pool after harvest. Thus, while soybean is an excellent scavenger crop and can reduce soil nitrate concentrations to very low levels before harvest, soil nitrate levels after soybean harvest often differ little from those after corn (Varvel et al., 1995). Because these are weather-related processes, it is difficult to predict how much and how rapidly N in soybean residues become available to the following corn crop. In addition, surfaces of most soils after producing a soybean crop are more friable than after corn, possibly affecting rates of soil N mineralization the next season. Omay et al. (1998) showed that most of the ^{15}N in soybean residues was found in the soil organic matter 2 yr after application (Table 8), suggesting that soybean residues increase mineralizable soil N. On two soils they found that an additional 150 kg fertilizer N ha^{-1} would need to be applied to continuous corn to maintain crop yield levels obtained with the corn–soybean rotation. Our knowledge of the processes by which soybean affects soil N availability is inadequate and much more research is needed.

Other Rotations

Several crops other than soybean were grown in rotation with corn in some MSEA studies. Wheat in a cropping system was effective in reducing soil nitrate levels in Ohio and Missouri studies (Kitchen et al., 1995a; Subler et al., 1995b), thereby reducing nitrate leaching potential. However, rotations in the Midwest containing wheat were usually not economically competitive. Blanchard et al. (1995) showed that a sorghum–soybean rotation was effective in maintaining crop yields (economic income) and water quality for the claypan soils of Missouri. Alfalfa effectively removes nitrate accumulations from subsoils and was an excellent scavenger crop (Kanwar et al., 1997; Lamb et al., 1995; Randall et al., 1993b; Watts et al., 1997). In several locations, residual effects of an alfalfa crop lasted for a number of years, increasing soil N availability more than anticipated (Lamb et al., 1998). In other studies, perennial grasses were also found effective in reducing subsoil nitrate concentrations (Randall et al., 1993b; Rickerl et al., 1993).

Weather Impacts

Mention has been made several times of interactions that weather may have with components of farming systems to affect nitrate availability, uptake, and leaching. Weather directly affects ambient water and temperature regimes, thereby affecting plant water relationships, soil oxygen status, plant growth rates, mineralization and nitrification processes, and ultimately crop yield and nitrate leaching. Also, weather conditions can directly affect crop N requirement by direct damage to the crop (wind and hail) or indirectly by stimulating growth and activity of plant pests (weeds, insects, and disease), which reduce crop yield potential and N requirements. At most MSEA sites during the five years of study, approximately two-fold variations in crop yields were measured, largely attributed to direct or indirect effects of weather. (Colvin et al., 1997; Lamb et al., 1997). This is well illustrated by data presented by Lamb et al. (1997). They divided a 1.7-ha field into 60 subplots, harvesting each subplot separately for 5 yr. Results showed that there was little consistency in high and low producing areas from year to year, and that only 4 to 42% of the grain yield variability for a given year could be predicted by knowledge of yield from a previous year. When crop yields for each subplot were arranged in 10% intervals of maximum yield, only 3.3% of subplots were in the same yield interval all five years (Fig. 2). Most of this variation was attributed to the effects of annual weather variation on crop growth and factors affecting N availability.

In 1993, most sites experienced record or near record growing season precipitation, greatly affecting water movement and leaching. For example, Hatfield et al. (1996) showed that quantity of nitrates removed in tile drainage from fields, subbasins, and the entire Walnut Creek watershed in central Iowa in 1993 was more than twice that for any other year. At all scales (tile discharge, basins, and watersheds), discharge between the wettest and driest years varied more than 10-fold. In a related study, Hatfield et al. (1999b) found little spatial variability in rainfall on a monthly or season basis for 22 rain gages located within the 5130-ha watershed.

Rice and Havlin (1994) reviewed the literature on N mineralization indices and found that few if any field studies showed a good relationship between any mineralization index and field measurements of crop yield or N uptake. Likewise, many existing mineralization models failed in this regard. They concluded that model failure usually results from weather variations, especially during the late growing season, because these variations affect N dynamics and plant N needs. There are many published examples of the effects of soil water and temperature regimes on soil N transformations, movement, avail-

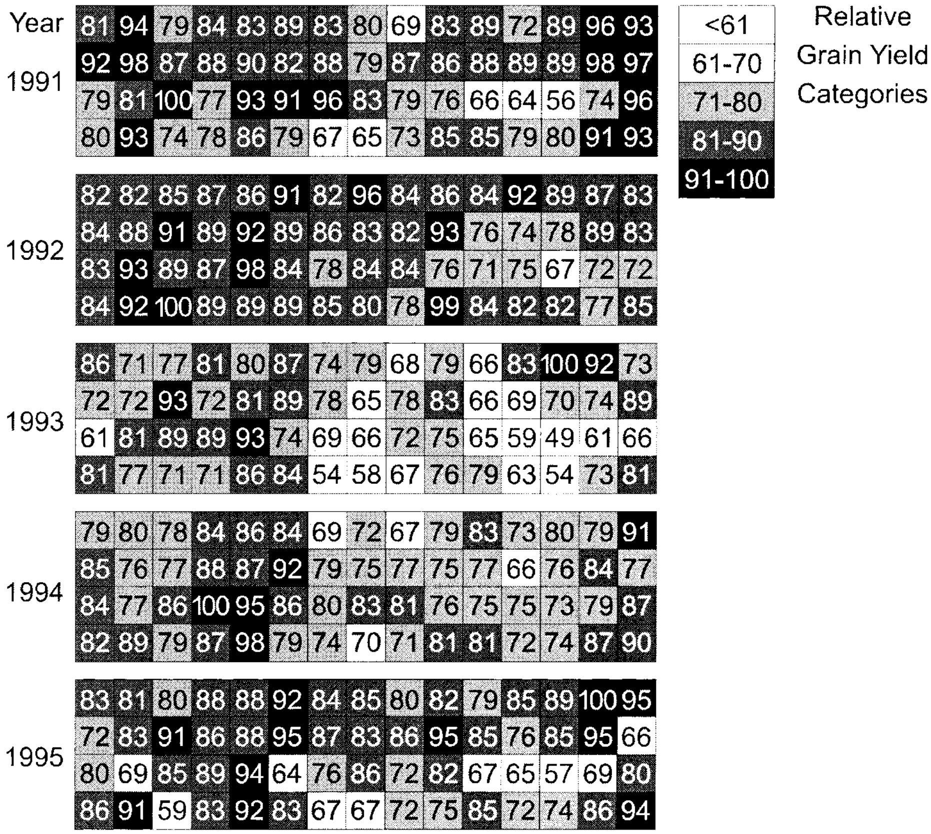

Fig. 2. Relative grain yields for 60 cells in each of five 1.7-ha fields from 1991 to 1995 at Princeton, MN (Lamb et al., 1997).

ability, and losses. Generally, these show that N cycling processes are upset by abnormally high or low soil temperatures and by deficient or excessive water supplies. Linn and Doran (1984) demonstrated that for most soils, most aerobic N transformations were near optimum when approximately 60% of the soil pore space is filled with water (near field capacity for most soils).

Soil Nitrogen Transformation Processes

Several MSEA studies investigated the effects of farming system components on various soil N transformations, especially mineralization, nitrification, and denitrification. As mentioned above, Rice and Havlin (1994) concluded that soil N mineralization indices were poorly related to field data on crop N uptake largely because of variations in weather conditions. Drinkwater et al. (1996) reviewed methods for measuring the mineralizable N pool in soils, and provided detailed procedures for using short-term aerobic and anaerobic incubation techniques. They concluded that the active soil N pool can be represented by the sum of soil inorganic N, microbial biomass N, and mineralizable N. Large inputs of organic matter to soils increase especially the microbial biomass and mineralizable N pools, thereby increasing the N supplying power of soils without greatly increasing inorganic N pools and subsequent N leaching potential. Omay et al. (1997) used long-term incubation techniques (370 d) to study N mineralization and transformations in two soils. They found that the percent of soil organic N that was mineralized during incubation was not affected by crop rotation (continuous corn versus corn–soybean rotation) or N fertilizer in the silt loam soil, but was affected by these factors for a loam soil. The fraction of the mineralizable N found in microbial biomass was reduced by fertilization in the silt loam soil, and by rotation in the loam soil. Omay et al. (1997) concluded that mineralizable N responses to N fertilization and crop rotation were dependent on quantity of crop residues returned to soils and on soil texture.

Subler et al. (1995b) estimated mineralization and nitrification rates by measuring the absorption of nitrates on buried anion-exchange membranes placed in the soil. Addition of wheat straw or hairy vetch leaves to the soil reduced nitrification rates, probably because of increased immobilization of N by microbial biomass. In another study, Subler et al. (1995a,c) compared N mineralization rates as measured by the buried plastic bag technique to those measured by inserting metal or PVC-bound cores into the soil. They found that both methods were reasonably reliable for estimating N mineralization in field soils with no treatment as well as for soils receiving N fertilizers, legumes, or animal manures. Using δ-N^{15} measurements of plants and soils, Clay et al. (1997) calculated soil N mineralization rates and concluded that results varied with elevation and topographic positions. Poor drainage increased denitrification. They also concluded that the δ-N^{15} technique complemented the difference method in calculating N fertilizer use efficiency. Subler et al. (1996) showed that addition of earthworms to the soil increased potentially mineralizable and microbial biomass N, as well as dissolved organic N concentrations. Omay et al. (1998) demonstrated that 49 to 67% of the N^{15} fertilizer applied was immobilized as organic N after the first growing season. Only a few percent of this N was mineralized and utilized by the crop the next year, indicating that by returning crop residues, the mineralizable soil N pool was enhanced. Jacinthe et al. (1999) found that low soil pH slowed or stopped nitrification.

Under certain conditions, considerable amounts of N may be removed from a soil by denitrification. Kessavalou et al. (1996), using N^{15} techniques, found that in a wet year (1993), 13% of the applied N was lost by denitrification during the growing season of irrigated corn, and an additional 41% was lost by leaching. However, in drier years at the same location and using surface gas chamber techniques, Qian et al. (1997) measured denitrification rates of only 1 to 5% of the amount of fertilizer N applied. Only when water-filled pore space in the soil exceeded 70% did they measure denitrification activities of 0.2 to 1.4 kg N ha^{-1} d^{-1}. Clay et al. (1997) showed a net loss of 95 kg N ha^{-1} during a growing season for a poorly drained soil, which they attributed to denitrification. Jacinthe and Dick (1997) also measured short-lived fluxes on N_2O following rainfall events. Seasonal N losses by denitrification were greatest for continuous corn (3.7% of the fertilizer N applied) and least in soybean (0.6%) plots. Over the entire season losses accounted for 0.5 to 3.0% of the fertilizer N added. Jacinthe and Dick (1997) concluded that denitrification rates could also be restricted by lack of soluble organic C in the soil or by presence of a restricted number of denitrifying organisms. In other studies, Jacinthe et al. (1999) demonstrated denitrification techniques that may be used for bioremediation on nitrate-laden ground water near a well head. Jacinthe and Dick (1996) also published some improvements in methods used to measure denitrification.

Soil Variability

Probably one of the more important results of MSEA activity was recognition of the widespread inherent soil variability that exists within most fields and even within a given soil type within a field. Historically, most farmers have treated a field as a unit, applying the same management practices over the entire field. As a consequence, while presumably the treatments used are best for a major part of the field, they may be inappropriate for parts of the field. Such variability often results in adverse effects on crop yields and water quality from those areas of the field not properly managed (Tomer et al., 1997; Blackmer and Schepers, 1996; Blackmer et al., 1996a; Clay et al., 1997; Lamb et al., 1997; J.G. Lyon and others, Ohio State University, unpublished data, 1998; Senay et al., 1998). Thus, most of the nitrate leaching measured occurred from only a fraction of the total area of the field (Kranz and Kanwar, 1995; Wu et al., 1996). To address this problem, many MSEA scientists initiated research on site-specific or precision farming methods that would manage each area of the field according to its needs rather than manage the field as a unit.

The nature of this inherent soil variability existing within fields was investigated in greater detail at several locations. Cambardella et al. (1994) grid-sampled two fields in Boone County, Iowa and measured a number of properties of the upper 15 cm of soil. They assessed spatial variability by calculating the ratio of values for nugget to total semivariance for each property, and classified those with ratios less than 0.25 as strongly spatially dependent, those between 0.25 and 0.75 as moderately spatially dependent, and those greater than 0.75 as weakly spatially dependent. They found that twelve properties in one field (including organic C, total N, pH, and microaggregation) and four properties in the second field (including organic C and total N) were strongly spatially dependent. Six properties in the first field (including biomass C and N, bulk density, and denitrification) and nine properties in the second field (including biomass C and N and bulk density) were moderately dependent. Three properties in the first field (including nitrate N and ergosterol) and one property in the second field (mineral-associated N) were weakly spatially dependent. While results showed that variability within a soil type was often as great as between soil types, they often found a relationship between soil properties and landscape position or elevation.

In studies on claypan soils in Missouri, Sudduth et al. (1995, 1996) and Birrell et al. (1993) found that much of the variability in crop yield and soil properties they encountered could be explained by depth to the claypan layer. Nutrient management plans could be based on depth to claypan because this feature strongly influenced soil water relationships for the crop (Kitchen et al., 1995b; Kitchen et al., 1997). Depth to claypan could be easily and accurately measured using electromagnetic induction techniques (Sudduth et al., 1995). Birrell et al. (1996) compared grid and kriged maps for soil potassium, phosphorus, and pH and found that accuracy and variability of maps derived from these data varied greatly among these analyses.

In Minnesota, Tomer and associates (Tomer and Anderson, 1994; Tomer et al., 1995) studied variability across a sandy plain hillslope. They found that variability in water storage was often associated with clay lenses within the solum, changing infiltration and water retention patterns. They concluded that topographic trends can be used to model soil variability for this landscape. In South Dakota, Clay et al. (1997) also concluded that many soil N transformations and properties were related to topography and soil water relationships. However, on similar soils in Minnesota, Lamb et al. (1997) found that annual corn yield variability was not related to topography. Laboski et al. (1998) showed that wheel track packing of these soils also affected soil water relationships and corn rooting patterns.

In a series of studies, Schepers, Blackmer, and associates demonstrated that soil variability, particularly as it affects soil and crop N status, can be assessed and mapped by remote sensing (Blackmer and Schepers, 1995, 1996; Blackmer et al., 1996a,b). They found that black and white aerial photographs provided reliable information on soil and crop variability. Also, plant reflectivity near 550 and 710 nm provided good measures of relative plant greenness, an attribute that they demonstrated could be used for managing N inputs. This technology is discussed later in this paper. The utility of aerial photographs in assessing soil and crop variability was confirmed in Ohio by Senay and associates (Senay et al., 1998, 2000) and in Minnesota by Tomer et al. (1997).

Nitrogen Management Systems

Many details of results of the N managements systems studied in MSEA were reviewed earlier in this paper and in a previous paper (Power et al., 2000). In this section we summarize present technologies and the potential use of models to characterize these systems, and take a look at new technologies that have or are being developed to manage N in farming systems.

Current Technology

The best pre-MSEA technologies for managing N in farming systems includes monitoring soil nitrate N levels by sampling soils for nitrates shortly before or after planting, then applying sufficient fertilizer N to meet anticipated crop needs based on soil test results and anticipated crop yield, giving due credit for legumes used in the rotation or previous use of animal manures. Most State Agricultural Experiment Stations have developed algorithms for calculating fertilizer N needs, based on field calibrations under the prevailing soil and climatic conditions of that state. In most instances this approach provides a reasonably accurate estimate of fertilizer N needs.

The present approach has several inherent problems that may result in serious over- or underfertilization, in some instances increasing potential for nitrate pollution of water resource or reduction of economic returns. First, when only one soil sample for nitrate analysis is collected for each 3 to 15 ha, because of inherent soil variability, there is a good likelihood that this one sample will not be representative of the entire area. Second, most often the entire field is treated as a unit, receiving uniform management and fertilization practices over the entire field. Inherent soil variability is not taken into account, resulting in some areas of the field receiving more fertilizer N than required. Third, fertilizer N requirements are based on anticipated crop yields. Schepers et al. (1991a) found that over a 4-yr period for several thousand corn producers, yield goals exceeded harvested yield by 10% or more. Yield goals are usually established on the assumption that normal growing conditions (precipitation and temperatures) will prevail during the forthcoming season. Reality shows that excesses or deficiencies in rainfall and/or temperature dictate and often result in reduced crop yields. This of course results in less fertilizer N uptake than anticipated, increasing potential for nitrate leaching. These inherent errors in our present technologies indicate a need for improved N management practices as world population pressures force us to strive for even higher crop yields and, presumably, use of higher fertilizer N rates.

New Technologies

As mentioned earlier in this paper, at most locations it was readily apparent that spatial variability and unpredictable weather were major factors affecting nitrate movement and degradation of water quality. The MSEA results showed that present technologies, if properly used, would be helpful to reduce present levels of nitrate pollution of water resources, but these technologies appear to be incapable of reducing adverse effects of present practices to acceptable levels in all situations. Thus, more new technologies are needed.

The major new technology arising from these MSEA activities is the realization that remote sensing techniques can be used to assess crop N status, allowing us to spoon-feed N to the crop to provide sufficient N for economically acceptable yields without overloading the soil with nitrates. This has been reviewed thoroughly in a previous paper (Power et al., 2000).

CONCLUSIONS

While the MSEA project stood as the largest research and demonstration activity on the effects of agriculture on water quality ever conducted in the United States, it remained too limited to provide comparisons of more than just a few complete farming systems. However, this is not necessarily a deficiency of MSEA because, as explained earlier in this paper, the best farming systems for one farm may not be best for a neighboring farm due to differences in resource inputs. Best farming systems can vary greatly, depending on available inputs. As a consequence, much of the MSEA activity centered on studies of the effects of certain farming system components or packages of components on crop production and water quality. Results indicate that there are indeed some packages of practices that seem almost universally to be better than other packages. For example, at most locations, packages that included a corn–soybean rotation, ridge-till, soil testing, and appropriate N fertilization practices often provided near maximum economic returns and the lowest degradation of water resources. However, any one of these practices by itself was often not particularly effective. Many more limitations exist than stated in this review of MSEA research. Diseases, insects, various nutrient deficiencies, weeds, and limited oxygen in soil can all cause reduced yield. Sensing technologies must be able to differentiate these causes before true gains in N management can be realized.

Other general results from the MSEA project often provided data that validated our present practices and recommendations or provided direction for the development of new technologies. For example:

- For tile drained soils in Iowa, continuous corn was not environmentally acceptable because of the large quantities of nitrates intercepted and discharged into surface waters.
- Reduced and no-till practices, compared with clean tillage, did a better job of synchronizing soil N mineralization activity with N uptake requirements of

corn, thereby reducing nitrate accumulations in the soil and subsequent potential for nitrate leaching.

- In irrigated crop production, sprinkler systems were superior to gravity systems because of more uniform water distribution and their ability to apply limited amounts of water per irrigation.
- A corn–soybean rotation may not reduce nitrate N accumulation and leaching compared with continuous corn if inadequate N credits for soybean are used.
- While present practices usually provide near-maximum economic returns with acceptable levels of nitrate leaching, some of our present farming systems fail under certain conditions. The most likely causes for such failures are abnormal weather during the growing season and inherent soil variability.
- A good soil testing program is a necessity to implement the best rate of fertilizer N for crops.
- New technologies for farming systems are needed that address causes for nitrate degradation of water resources, that is, weather variability and soil variability.
- The interactive play of so many effects on farming systems and the ultimate effect on water quality is clearly a result of the MSEA research effort. Continual research efforts must occur to influence science-based public policy and regulation affecting the crop production component of agriculture.

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