Nitrate Nitrogen in Surface Waters as Influenced by Climatic Conditions and Agricultural Practices

Gyles W. Randall* and David J. Mulla

ABSTRACT

Subsurface tile drainage from row-crop agricultural production systems has been identified as a major source of nitrate entering surface waters in the Mississippi River basin. Noncontrollable factors such as precipitation and mineralization of soil organic matter have a tremendous effect on drainage losses, nitrate concentrations, and nitrate loadings in subsurface drainage water. Cropping system and nutrient management inputs are controllable factors that have a varying influence on nitrate losses. Row crops leak substantially greater amounts of nitrate compared with perennial crops; however, satisfactory economic return with many perennials is an obstacle at present. Improving N management by applying the correct rate of N at the optimum time and giving proper credits to previous legume crops and animal manure applications will also lead to reduced nitrate losses. Nitrate losses have been shown to be minimally affected by tillage systems compared with N management practices. Scientists and policymakers must understand these factors as they develop educational materials and environmental guidelines for reducing nitrate losses to surface waters.

NITROGEN (N) is a naturally occurring element that is essential to plant growth and crop production. However, nitrate N can cause eutrophication of surface waters primarily by stimulating algae production. In a soil system, nitrate N is continually supplied through the natural processes of mineralization and nitrification of soil organic matter. Other sources of N include fertilizers, animal manures, municipal sewage wastes, agricultural and industrial wastes, atmospheric deposition, and dinitrogen fixation, all of which can be converted to nitrate N through mineralization and nitrification. Nitrate N is mobile and, therefore, can be lost from the soil profile by leaching. Subsequent transport of nitrate N to surface waters occurs through subsurface drainage (tile lines) or base flow. Very little nitrate N is lost from the landscape via surface runoff (Jackson et al., 1973). Increasing concentrations of nitrate in the Mississippi River have been linked to the hypoxic conditions in the Gulf of Mexico (Rabalais et al., 1996; Turner and Rabalais, 1991).

ROLE OF AGRICULTURE

Agriculture has been identified as a potential major contributor of nitrate N to surface water. Omernik (1977) reported that total N concentrations were nearly nine times greater downstream from agricultural lands than downstream from forested areas, with the highest concentrations being found in the Corn Belt states. Streamwater collected from 1984 through 1993 for a portion of the Upper Mississippi River basin was analyzed for nitrate N (Kroening, 1996). Nitrate N concentrations were significantly greater (2 to 6 mg/L) from those rivers that drain a large percentage of agricultural land compared with those that drain a larger percentage of forested land (0.1 to 0.5 mg/L). In the Mississippi River, mean concentrations were significantly greater (1.8 to 2.5 mg/L) downstream of the confluence with the Minnesota River (an agricultural watershed) than upstream (0.2 to 0.9 mg/L). Keeney and DeLuca (1993) examined nitrate N concentrations in the Des Moines River in 1945, 1955, 1976, and annually from 1980 through 1990 and found the average nitrate N concentration had changed little in the last 45 years (5.0 mg/L in 1945 to 5.6 mg/L in 1980–1990). They concluded that intensive agricultural practices that enhance mineralization of soil N coupled with subsurface tile drainage are the major contributors of nitrate N rather than solely fertilizer N.

Somewhat similar conclusions were drawn by David et al. (1997), who surmised that high soil mineralization rates and N fertilization combined with tile drainage contributed significantly to nitrate export in the Embarras River in Illinois. In their 6-yr study, an average of 49% (range from 25 to 85%) of the pool of residual nitrate N remaining after harvest was leached through drain tiles and exported into the river. Precipitation exerted a tremendous influence on drainage losses with a few days of high-flow events leading to most of the annual loss in some years.

The use of N and P fertilizer has been identified as a possible cause of the zone of hypoxia in the Gulf of Mexico (Rabalais et al., 1996). However, Smith et al. (1993) noted that increasing trends in nitrate concentration in rivers draining agricultural areas in the U.S. were far fewer than they had been in the late 1970s, and attributed this to the leveling off of N fertilizer use in the United States during the 1980s. This was in contrast to the increased nitrate concentrations found during the 1974–1981 period, which were attributed to increasing trends in fertilizer N use (Smith et al., 1987).

Fertilizer N use data, based on tons of fertilizer sold within each state, have been compiled annually since 1945 (Terry et al., 1995). The amount of fertilizer used and the rate of application per crop acre for nine midwestern states draining into the Mississippi River are shown in Fig. 1. Although significant year-to-year variation exists, it is apparent that total fertilizer N use has increased little in this nine-state area (Illinois, Indiana, Iowa, Kansas, Minnesota, Missouri, Nebraska, South

G.W. Randall, Southern Experiment Station, Univ. of Minnesota, Waseca, MN 56093. D.J. Mulla, Dep. of Soil, Water, and Climate, Univ. of Minnesota, St. Paul, MN 55108. Received 14 July 2000. *Corresponding author (grandall@soils.umn.edu).

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Abbreviations: RSN, residual soil nitrate nitrogen.

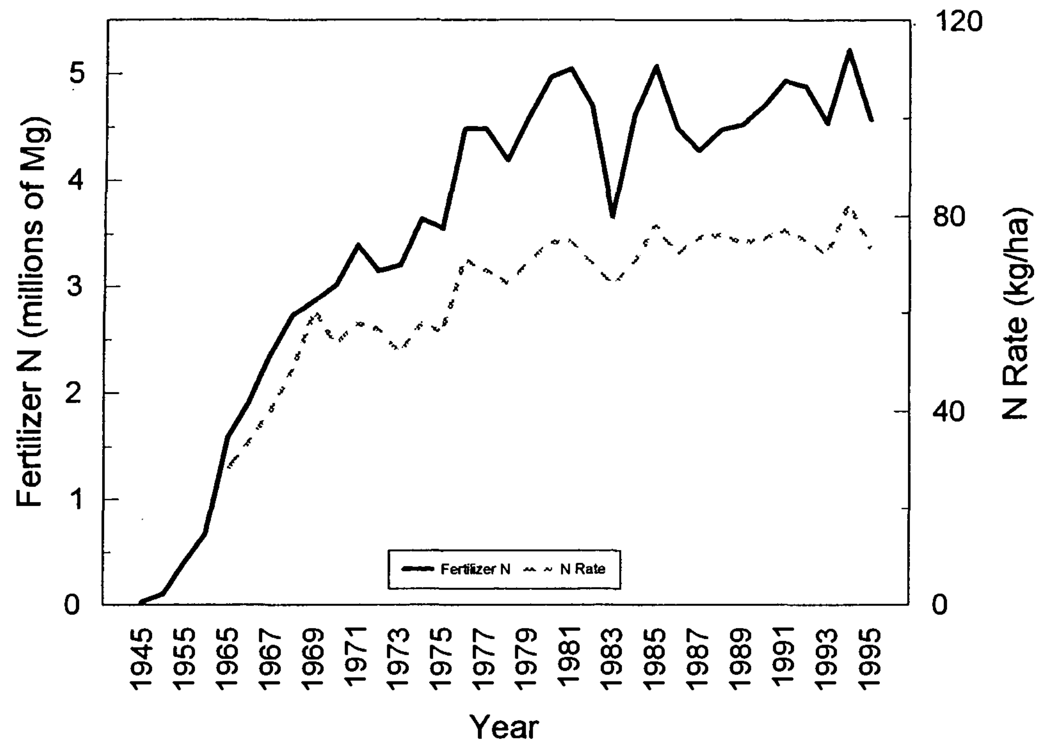


Fig. 1. Fertilizer N sold and rate of application for the nine-state midwestern area draining into the Mississippi River basin.

Dakota, and Wisconsin) since the early 1980s. Various mathematical models were applied to the data to determine when fertilizer use peaked or plateaued. The quadratic model indicated that fertilizer N use (sales) peaked in 1989. The linear response plateau (LRP) and quadratic response plateau (QRP) models indicated fertilizer N plateaued beginning in 1980 and 1987, respectively. These data refute the frequent statement of increasing N fertilizer use in the Midwest.

INFLUENCE OF PRECIPITATION

Loading of nitrate N into surface water is a function of transport volume (amount of water) and nitrate N concentration in the transported water. The amount of drainage water leaving the landscape is largely a function of climate and soil properties (i.e., precipitation, texture, infiltration rate, etc.). Drainage is further influenced by the temporal distribution of precipitation within a particular year (i.e., the amount of total annual vs. growing season precipitation that occurs). For instance, an 80-mm rainfall in the spring, when evapotranspiration (ET) losses are low and soil moisture in the profile is probably near field capacity, will have a much greater effect on drainage volume than the same rainfall during the middle of the summer, when daily ET losses are high and soil moisture is far short of field capacity. In the former scenario, storage capacity is minimal and drainage water carrying nitrates is plentiful. A significant storage reservoir can exist in the soil in the latter scenario and drainage may or may not occur.

Goolsby et al. (1997) noted that the concentration and flux of nitrate tend to be highest in the spring when stream flow is highest. This direct relationship between nitrate concentration and flow may result from leaching of nitrate from the soil during periods of high rainfall. Increased flows and elevated concentrations in agricultural tile drains were also speculated as contributing to this relationship.

The effect of climate on subsurface drainage is abundantly clear in the following tile drainage studies. Annual tile drainage in a Minnesota study conducted from 1986 to 1992 on a Webster clay loam (fine-loamy, mixed, superactive, mesic Typic Endoaquoll) with continuous corn (*Zea mays* L.) ranged from 26 to 618 mm/yr with an average of 297 mm (Randall and Iragavarapu, 1995). Drainage was least in 1989 when growing season precipitation was 35% below normal and greatest in 1991 when growing season precipitation was 51% above normal (Table 1). In addition, drainage in the 3-yr dry period (1987–1989) averaged only 38 mm compared with the following 3-yr wet period (1990–1992) when drainage averaged 507 mm. Similar findings were reported by Weed and Kanwar (1996) who measured tile drainage under both continuous corn and a corn–soybean [*Gly-*

Table 1. Influence of precipitation on subsurface tile drainage and annual nitrate N concentration and losses.

Year	April–October rainfall†	Drainage	Nitrate N Conc.‡	Nitrate N Lost
	mm	mm	mg/L	kg/ha
1986	796	402	14	55
1987	586	42	9	4
1988	426	46	15	6
1989	414	26	12	2
1990	789	486	24	112
1991	961	618	24	139
1992	726	417	14	55

† 1961–1990 normal = 639 mm.
‡ Annual flow-weighted concentration.

Table 2. Annual water loss via subsurface tile drainage for two cropping systems in Iowa (Weed and Kanwar, 1996).

Crop system	Year 1990	1991	1992	Avg.
	mm			
Continuous corn	185	280	122	195
Rotation corn	143	167	72	127
Rotation soybean	160	288	113	187

cine max (L.) Merr.] rotation on Kenyon (fine-loamy, mixed, mesic Typic Hapludoll)–Clyde (fine-loamy, mixed, superactive, mesic Typic Endoaquoll)–Floyd (fine-loamy, mixed, superactive, mesic Aquic Hapludoll) soils in Iowa. Averaged across four tillage systems, drainage in 1991 totaled 244 mm or 44% above the 1990–1992 average (Table 2). A 6-yr study conducted on a Normania clay loam (fine-loamy, mixed, superactive, mesic Aquic Hapludoll) at Lamberton, MN showed no tile drainage in 1988 and 1989 when annual precipitation was 69 and 76% of normal, respectively (Randall et al., 1997). Drainage under continuous corn and a corn–soybean rotation averaged 22 mm in 1990, 223 mm in 1991, 143 mm in 1992, and 469 mm in 1993 (Table 3). Annual precipitation in those four years was 95, 125, 117, and 160% of normal, respectively. Data from these three studies clearly indicate the strong relationship between precipitation and volume of subsurface tile drainage.

Nitrate N concentrations and losses are also greatly affected by dry and wet climatic cycles (Randall, 1998). Thirty-two tile drainage plots were planted to a corn (16 plots)–soybean (16 plots) rotation from 1987 through 1993 at Waseca, MN. Late each fall after soybean harvest, anhydrous ammonia was applied to four plots at a rate of 150 kg N/ha for corn the following year. Average annual flow-weighted nitrate N concentrations and losses from the corn plots are shown in Fig. 2. In 1987 and 1988, when April through October rainfall was 8 and 33% below normal, respectively, subsurface drainage was <50 mm/yr and nitrate N concentrations ranged between 7 and 18 mg/L. Less than 2 mm of drainage occurred in 1989 when April–October rainfall was 35% below normal, and no samples were collected for nitrate N analyses. Under these dry conditions during the 3-yr period, corn yields and N uptake were low. However, residual soil nitrate nitrogen (RSN) continued to increase in the soil profile to levels as high as 259 kg/ha in the top 1.5-m profile. April–October precipitation in

Table 3. Effect of crop system on amount of subsurface drainage water.

Crop system	Year 1990	1991	1992	1993
	mm			
Continuous corn	20	178	132	442
Corn–soybean	18	274	122	488
Soybean–corn	28	218	175	478
Alfalfa	0	41	56	320
CRP†	0	43	86	510
Percent of normal annual precipitation	95	125	117	160

† Conservation Reserve Program.

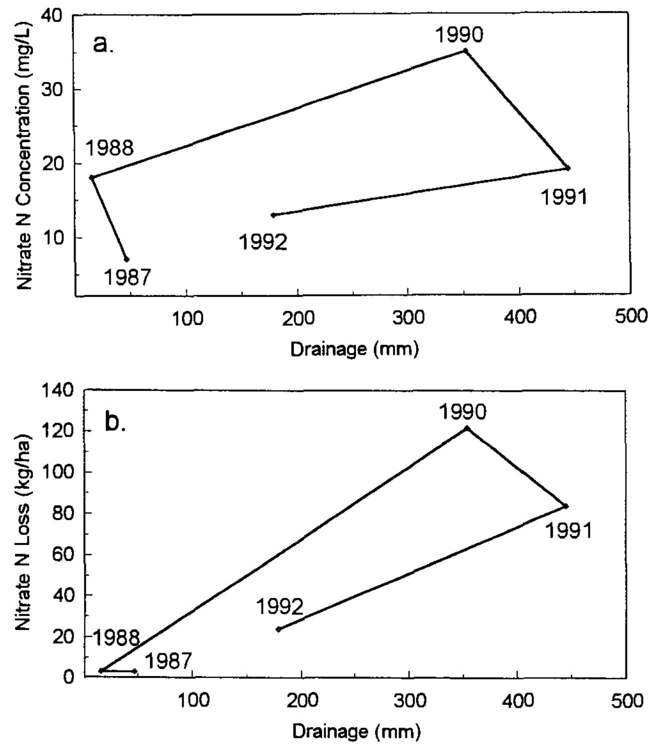


Fig. 2. Relationship between subsurface tile drainage and (*a*) annual flow-weighted nitrate N concentration and (*b*) annual nitrate N loss in tile drainage water from a corn–soybean rotation that received 150 kg N/ha as anhydrous ammonia in late October each year following soybean at Waseca, MN.

1990 was 23% above normal, causing drainage volume to total >350 mm. Moreover, the annual flow-weighted nitrate N concentration averaged 35 mg/L, twice as high as during the dry years (Fig. 2a). Nitrate N concentrations in the soil and drainage water returned to background levels in 1991 and 1992 when rainfall was 50 and 14% above normal, respectively. Nitrate N losses shown in Fig. 2b show the combined effect of drainage and nitrate N concentration. These data strongly suggest that RSN can accumulate in the soil profile during dry climatic cycles because of soil mineralization and every-other-year N fertilization, even in a corn–soybean rotation. These elevated RSN levels are then poised for transport from the soil profile via subsurface tile drainage and delivery to streams when growing season precipitation returns to above-normal amounts.

In another set of four drainage plots at Waseca, continuous corn was grown from 1985 through 1992. Fertilizer N was applied at a rate of 200 kg/ha each spring. Annual flow-weighted nitrate N concentrations in 1985 and 1986 averaged 13 and 14 mg/L, respectively, although drainage ranged from 143 mm in 1985 to 402 mm in 1986 (Fig. 3). Dry conditions during 1987–1989, when April–October rainfall was 25% below normal, resulted in <50 mm drainage/yr and annual average nitrate N concentrations ranging from 9 to 15 mg/L. Residual soil nitrate nitrogen totaled 225 kg/ha in the 0- to 1.5-m profile. In 1990 and 1991, April–October rainfall averaged 36% above normal and generated annual drainage >480 mm/yr (Fig. 3a). In addition, nitrate N concentrations in the drainage water doubled from

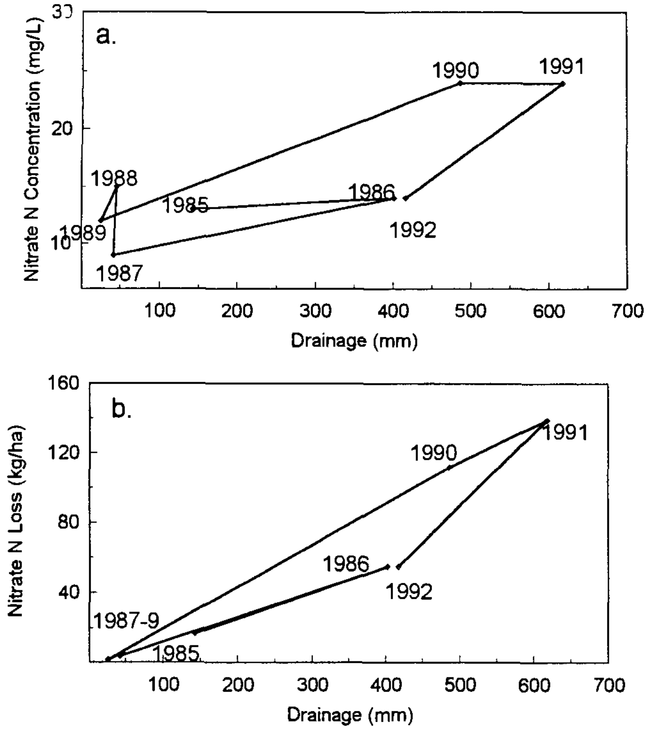


Fig. 3. Relationship between subsurface tile drainage and (*a*) annual flow-weighted nitrate N concentration and (*b*) annual nitrate N loss in tile drainage water from continuous corn that received 200 kg N/ha each spring at Waseca, MN.

the previous three dry years to 24 mg/L in these two wet years. Residual soil nitrate nitrogen at the end of 1991 was 50% lower than at the end of the dry years. In the third consecutive wet year (1992), more than 400 mm/ha of water drained from the plots, nitrate N concentrations in the drainage water returned to 14 mg/L, and RSN totaled only 50 kg/ha. Nitrate N loading in the subsurface drainage water each year was greatly affected by both nitrate N concentration and drainage volume (Fig. 3b). These data clearly indicate a buildup of RSN in the soil profile during dry years when drainage was limited. Much of the RSN buildup could probably be attributed to mineralization of soil organic matter, annual additions of fertilizer N, and limited uptake of N by the poor-yielding corn. In the subsequent wet years, substantial losses of nitrate N occurred in subsurface drainage due to high concentrations of nitrate N and high drainage volumes.

The general effects of precipitation on nitrate N losses can also be illustrated using basin-wide water quality monitoring data collected in the Minnesota River basin, a 4.0 million-hectare agricultural basin draining to the Upper Mississippi River basin. Mean annual precipitation in the Minnesota River basin varies from 560 mm (22 in) on the western side of the basin to 790 mm (31 in) on the eastern side (Fig. 4). The basin is dominated by intensive row-crop agriculture, has soils that generally have organic matter levels greater than 3%, and has subsurface tile-drainage on more than one-half of the farmed acreage.

The Minnesota River basin is subdivided into 12 major watersheds (Fig. 5). From 1977–1994, at the mouth

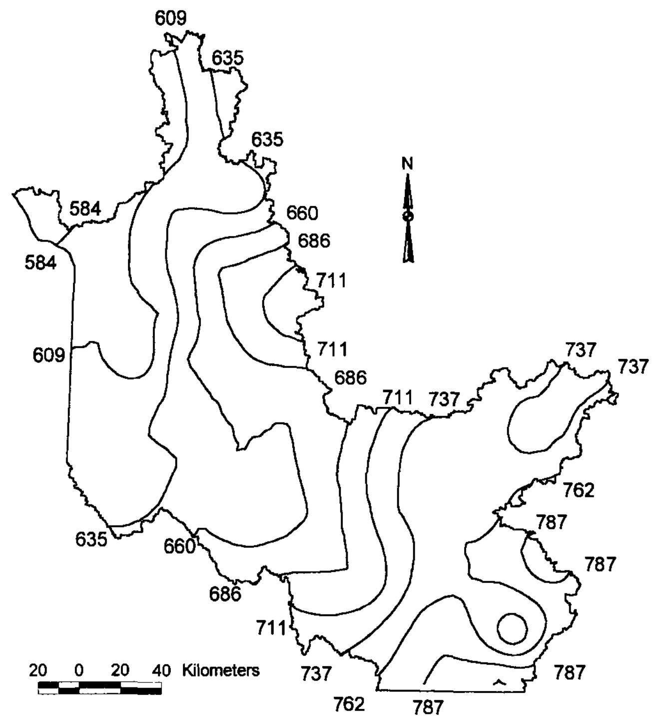
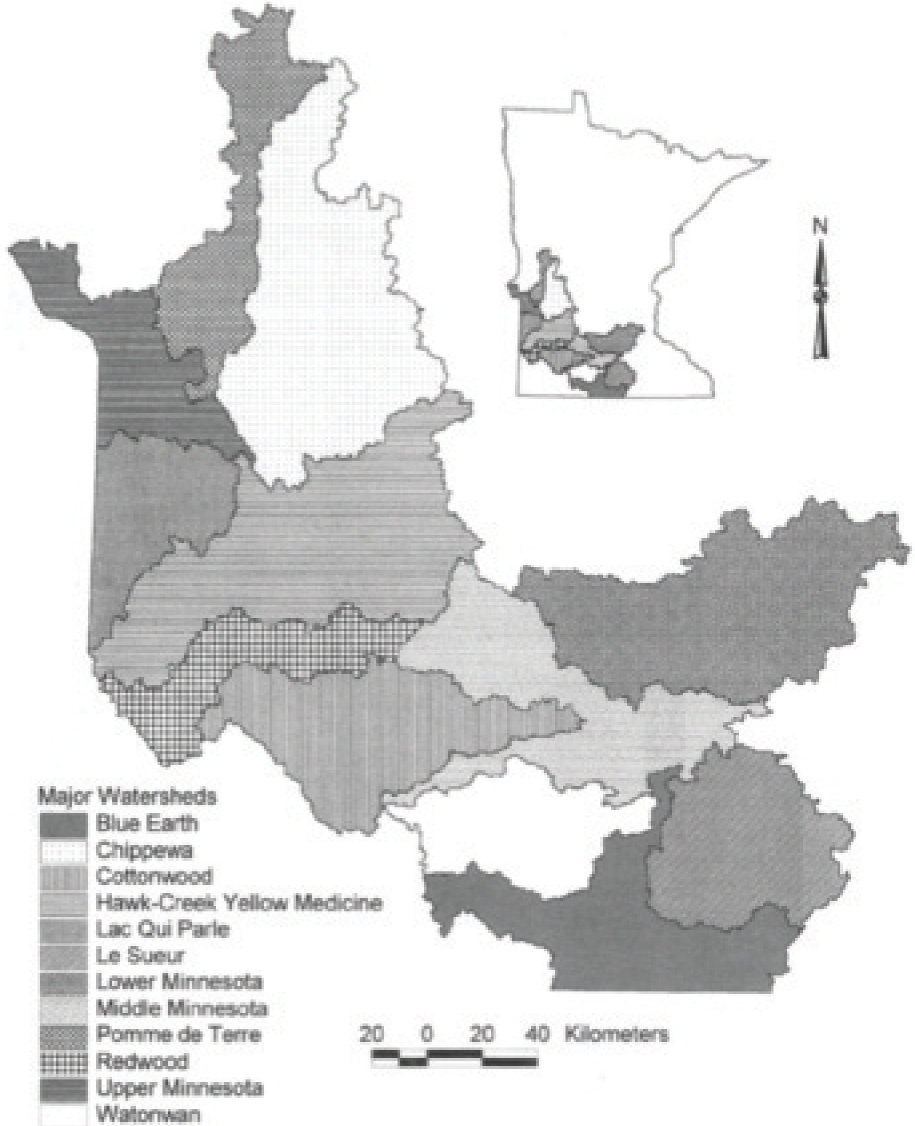


Fig. 4. Long-term precipitation patterns (inches) within the Minnesota River Basin.

of nearly every watershed, water quality monitoring data for nitrate N were collected by the Minnesota Pollution Control Agency (MPCA). These water quality monitoring data show that nitrate N concentrations range from 0.36 mg/L in the headwaters to 4.6 mg/L at the mouth of the river where it enters the Mississippi River. Mean annual precipitation increases by about 254 mm (10 inches) across this distance, which produces a corresponding and dramatic increase in the discharge from subsurface tile drains. This discharge enters ditches and streams that eventually flow into the Minnesota River. Along the Minnesota River from the uppermost reaches to the middle reaches, less than 1% of the water quality samples collected since 1977 have a nitrate N concentration that exceeds 10 mg/L, the maximum contaminant level for drinking water. From the middle reaches (downstream of the Blue Earth watershed) to the confluence between the Minnesota and Mississippi Rivers, about 10% of the water quality samples collected since 1977 exceed 10 mg/L.

Differences in nitrate N contributions across the basin in response to a gradient in precipitation are even larger when nitrate N loads are compared rather than nitrate N concentrations. Loads of nitrate N in the twelve tributaries to the Minnesota River basin were estimated for the period from 1977–1994 using the U.S. Army Corps of Engineers FLUX model, water flow data collected by the U.S. Geological Survey, and nitrate N concentration data collected by the Minnesota Pollution Control Agency. The proportion of the total nitrate N load leaving the Minnesota River basin from each of the 12 major watersheds is shown in Fig. 6. Four watersheds located in the wetter, eastern portion of the basin (the Lower Minnesota watershed, and the Greater Blue Earth wa-



tershed consisting of the Blue Earth, Le Sueur, and Watonwan watersheds) account for 75% of the total nitrate N load (2565 Mg/mo) in the entire basin, yet they drain only 31% of the total basin area. Six watersheds on the drier western side of the basin collectively generate only 7% of the nitrate N load.

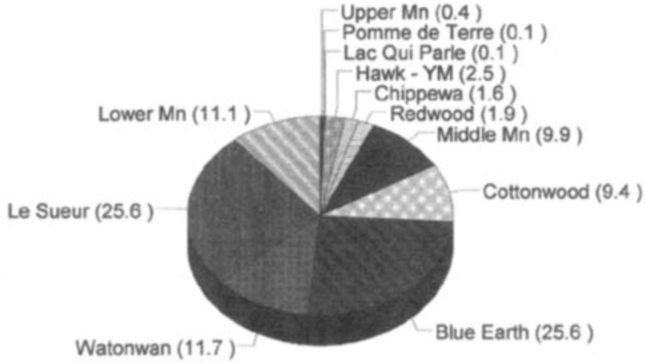


Fig. 6. Fraction of the total nitrate N loading from major watersheds within the Minnesota River basin.

Median values for nitrate N yields (load per unit area) for watersheds in the Minnesota River basin are shown in Fig. 7. Yields vary from about 0.5 to more than 6 kg/km^2/d (1.8 to more than 21.9 kg/ha/yr), with the larger yields occurring in the watersheds on the wetter eastern

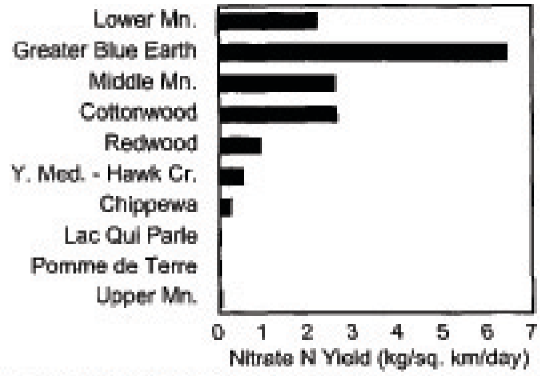


Fig. 7. Average daily nitrate N yields (365 d/yr) from major watersheds within the Minnesota River basin.

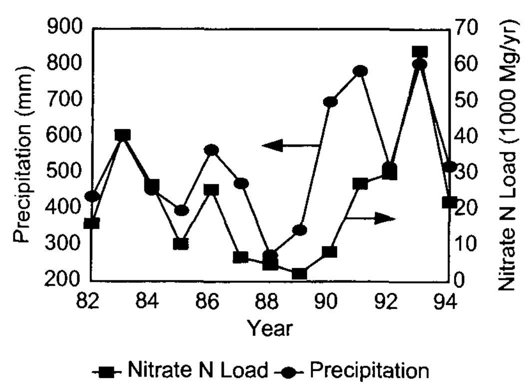


Fig. 8. Temporal changes in nitrate N loading from the Greater Blue Earth River watershed in relationship to annual precipitation recorded at Waseca from 1982–1994.

side of the basin. The mean value for annual nitrate N yield in the Minnesota River basin is 2.1 kg/km^2/d (7.7 kg/ha/yr). By way of comparison with other watersheds in the Mississippi River basin, the median nitrate N yields from 1973–1993 are 1.7, 4.0, 4.4, 0.1, 0.2, and 0.5 kg/km^2/d for the Ohio, Iowa, Illinois, Platte, Missouri, and Yazoo Rivers, respectively.

Temporal changes in nitrate N loads for the Blue Earth watershed of the Minnesota River basin illustrate the relationship between nitrate N loadings and growing season precipitation amounts at Waseca from 1982–1994 (Fig. 8). With the exception of the three years following the drought of the late 1980s, there is a good relationship between precipitation and nitrate N loads in the river. The exceptions are probably due to rainfall amounts at Waseca in 1992 that are not representative of the amounts in the entire basin, as well as the long-term effects of drought on mineralization of soil nitrogen.

Lastly, long-term precipitation trends need to be considered when characterizing the nitrate contamination of surface waters. In the 1930s, when very dry conditions prevailed across much of the U.S., drainage volumes and subsequent loading of nitrates to surface waters were minimal. Recent analysis of climatic data indicate that annual precipitation amounts have increased steadily in portions of the Upper Midwest since the early 1940s. Consequently, loading of nitrates to surface waters probably has increased during this time of wetter weather and greater drainage amounts.

INFLUENCE OF SOIL MINERALIZATION

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. Tile drainage from continuous corn plots that received only 20 kg N/ha/yr at Lamberton, MN contained annual flow-weighted nitrate N concentrations of 13, 19, and 19 mg/L in 1973, 1974, and 1975, respectively (Gast et al., 1978). No drainage occurred in 1976, an extremely dry year. In 1977, with slightly above-normal rainfall, nitrate N concentrations averaged 28 mg/L from these plots. In a study at Waseca, MN, four plots were fallowed (no crop grown and no N applied) from 1987 through 1993. Nitrate N concentration in the tile drainage water averaged 57 mg/L in 1990 following three dry years. Concentrations dropped to 38, 25, and 23 mg/L in 1991, 1992, and 1993, respectively (Randall, unpublished data, 1993). Based on data from these studies, high concentrations of nitrate N can easily be lost to tile drainage from high organic matter soils even if no N or very small amounts of N are applied, especially in wet years following dry years when crop production is limited. Hatfield (1996) found that nitrate N concentrations in the Walnut Creek (Iowa) watershed ranged from 15 to 20 mg/L throughout most of the year and stated that this loss is due primarily to the high organic matter content of the soils and their ability to mineralize N. Under these conditions, elevated levels of nitrate N will be lost to drainage water regardless of soil or nutrient management practices.

Table 4. Effect of crop system on flow-weighted annual nitrate N concentrations.

Crop system	1990	1991	1992	1993
	Year			
	mg NO_3–N/L			
Continuous corn	30	39	40	20
Corn–soybean	22	29	26	14
Soybean–corn	26	38	27	13
Alfalfa	–	4	4	1
CRP†	–	4	1	0.3

† Conservation Reserve Program.

INFLUENCE OF CROPPING SYSTEMS

Nitrate N concentrations in subsurface drainage water are related to crop rotation plus rate and timing of fertilizer N application (Baker and Melvin, 1994). Tile drainage water from row crop systems (continuous corn and a corn–soybean rotation) that were fertilized with N based on a soil nitrate test averaged between 14 and 40 mg nitrate N/L from 1990 to 1993 at Lamberton, MN (Table 4). In comparison, perennial crops (alfalfa [*Medicago sativa* L.] and a Conservation Reserve Program [CRP] grass–alfalfa mix) gave nitrate N concentrations ranging from 0.3 to 4 mg/L. Due to higher flow volumes from the plots planted to row crops, nitrate N losses from the row crops ranged from 30 to 50 times higher than from the perennial crops (Table 5) (Randall et al., 1997). Nitrate N concentrations under alfalfa were also shown to be much lower compared with corn or soybean in Iowa (Baker and Melvin, 1994). These findings are similar to those reported by Logan et al. (1980) who found highest NO_3–N losses with corn, intermediate with soybean or systems where other crops were in rotation, and lowest with alfalfa. Weed and Kanwar (1996) found higher nitrate N losses from plots planted

Table 5. Effect of crop system on nitrate N losses in subsurface drainage.

Crop system	Nitrate N lost, 4-yr total
	kg/ha
Continuous corn	217
Corn–soybean	204
Soybean–corn	202
Alfalfa	7
CRP†	4

† Conservation Reserve Program.

Table 6. Average NO_3-N concentration and annual NO_3-N loss in subsurface tile drainage water in Iowa (Weed and Kanwar, 1996).

Crop rotation	Tillage†	NO_3-N concentration			NO_3-N loss		
		1990	1991	1992	1990	1991	1992
		mg/L			kg/ha		
Continuous corn	MP	64	34	12	58	63	13
	CP	55	28	10	100	76	13
	RT	44	21	–	83	68	–
	NT	39	19	8	107	62	12
Corn–soybean	MP	39	24	8	41	36	6
	CP	33	21	7	51	36	5
	RT	24	19	3	34	30	3
	NT	19	17	8	32	31	4

† MP, moldboard plow; CP, chisel plow; RT, ridge tillage; NT, no tillage.

to continuous corn compared with a corn–soybean rotation in Iowa (Table 6). In summary, these studies show substantially higher nitrate N concentrations in row crops, especially continuous corn, compared with perennial crops that have an extended period of greater root activity (water and nutrient uptake) and where cycling of N is optimized.

INFLUENCE OF TILLAGE

Studies conducted in Iowa showed that tillage methods have less effect on nitrate N loss to drainage water than do crop rotations (Weed and Kanwar, 1996). Moldboard plowing gave the lowest flow volumes while ridge tillage and no tillage had the lowest nitrate N concentrations (Table 6). An 11-yr study with continuous corn at Waseca, MN showed similar results (Randall and Iragavarapu, 1995). Although slightly more water drained from the no-till plots, nitrate N concentrations were slightly lower compared with moldboard plow plots (Table 7). Thus, nitrate N flux in subsurface drainage was not influenced by tillage system.

INFLUENCE OF RATE AND TIME OF NITROGEN APPLICATION

Nitrogen was applied as ^{15}N depleted ammonium sulfate in the fall and spring for continuous corn during a 6-yr period at Waseca, MN. Corn yields from the late fall application (early November) of 134 and 202 kg N/ha averaged 8% lower than with spring (late April) application (Table 8). In addition, annual losses of nitrate N in the tile drainage water averaged 36% higher with fall application compared with spring application. Averaged across time of application, yields and nitrate N losses in the drainage water were 17 and 30% higher for the 202-kg rate compared with the 134-kg rate. At the end of the study, 65% of the N being lost in the drainage from the 202-kg fall treatment was derived from the fertilizer, whereas only 15% of the N in the drainage water lost from the 134-kg spring treatment was derived from the fertilizer (Buzicky et al., 1983).

Table 7. Effect of tillage on nitrate N losses in subsurface tile drainage.

Parameter	Tillage system†	
	Moldboard plow	No till
Drainage (mm)	279	315
Nitrate N concentration (mg/L)	15	13
Nitrate N lost (kg/ha)	43	41
N lost as a percent of applied N	21	20

† 11-yr (1982–1992) average.

Anhydrous ammonia was applied at a rate of 150 kg N/ha in four treatments (late fall, late fall + nitrapyrin, spring preplant, and split [40% preplant + 60% sidedress]) to drainage plots at Waseca, MN from 1987 through 1993. Flow-weighted nitrate N concentrations across the four-yr flow period (1990–1993) averaged 20, 17, 16, and 16 mg/L for the four treatments, respectively (Table 9). Nitrate N concentrations in 1990, following three dry years, were three times higher than in 1993—the fourth consecutive wet year. Corn yields were highest for the split treatment and lowest for fall application without nitrapyrin. Yields were increased significantly in the very wet years by the addition of nitrapyrin to the fall application (Randall and Vetsch, 1995).

Split application of N does not always result in increased N efficiency and reduced nitrate losses. Baker and Melvin (1994) reported losses of nitrate N to be higher for split application compared with a preplant application for continuous corn. Losses with split application for the corn–soybean rotation were lower in the year of application but tended to be higher in the following year when soybean followed corn. Based on data from these studies, fertilizer N management, particu-

Table 8. Effect of N rate and time of application on nitrate N losses and corn yield.

N†		Annual loss of nitrate N in drainage	5-yr yield avg.
Rate	Time		
kg/ha		kg/ha/yr	Mg/ha
0	–	8	4.1
134	fall	30	8.2
134	spring	21	9.4
202	fall	38	10.0
202	spring	29	10.5

† Ammonium sulfate applied about 1 Nov. or 1 May.

Table 9. Effect of time of N application and nitrapyrin on nitrate N losses and corn yield.

N treatment†	4-yr avg. annual NO_3-N concentration	4-yr total nitrate N lost	4-yr yield avg.
	mg/L	kg/ha	Mg/ha
Fall	20	264	8.0
Fall + nitrapyrin	17	208	8.6
Spring	16	177	8.6
Split	16	190	9.0
Fallow	36	365	–

† Anhydrous ammonia applied 25 Oct. or 1 May.

larly rate and time of application, plays a dominant role in the loss of nitrate N to surface waters.

STEPS TOWARD MINIMIZING NITRATE NITROGEN LOSS TO SURFACE WATERS

(i) The most obvious but least economical way to reduce nitrate N losses to surface water would be to abandon subsurface tile-drainage systems. The reality of this measure is not likely, however, as crop production on millions of acres of poorly drained soils in the Corn Belt would be reduced markedly.

(ii) An alternative to present tile discharge systems would be to construct wetland restoration areas or denitrifying ponds where drainage water could be routed and "treated" to remove excess concentrations of nitrate before discharge into drainage ditches or rivers. This may be a cost-effective practice in strategic portions of drainage watersheds.

(iii) Fertilizer N management could be fine-tuned. Applying the correct rate of N at the optimum time has been shown to have a substantial effect on nitrate N losses. Also, giving N credits to previous legume crops and animal manure applications is necessary to avoid overapplication of fertilizer N.

(iv) Development of improved soil N testing methods to determine the availability of mineralizable N and carryover N from the previous crop would be helpful, especially following dry years, legumes, or past manure applications.

(v) Alternative cropping systems that contain perennial crops would also probably reduce nitrate N losses. However, obtaining a market and a satisfactory economic return are obstacles facing farmers at the present time.

(vi) Improved management of animal manure would help lower nitrate N losses in livestock producing areas. Knowing the nutrient content and application rate of the manure, spreading it uniformly, and incorporating it in a timely manner would all lead to better management and confidence in manure N as a nutrient source.

SUMMARY

Noncontrollable factors such as climate and soil organic matter have a profound influence on nitrate N concentrations and loadings in subsurface drainage water. The dynamics of N behavior in drained agricultural soils during these periodic climatic events and the management of both crops and nutrient inputs (controllable factors) must be considered carefully by agriculturalists as they manage the land. Furthermore, these factors must be understood by scientists and policymakers as they educate the public and develop environmental guidelines regarding nitrate loading to surface waters.

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