Nitrate Loss in Subsurface Drainage as Affected by Nitrogen Fertilizer Rate

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ABSTRACT

The relationships between N fertilizer rate, yield, and NO_3 leaching need to be quantified to develop soil and crop management practices that are economically and environmentally sustainable. From 1996 through 1999, we measured yield and NO_3 loss from a subsurface drained field in central Iowa at three N fertilizer rates: a low (L) rate of 67 kg ha^{-1} in 1996 and 57 kg ha^{-1} in 1998, a medium (M) rate of 135 kg ha^{-1} in 1996 and 114 kg ha^{-1} in 1998, and a high (H) rate of 202 kg ha^{-1} in 1996 and 172 kg ha^{-1} in 1998. Corn (*Zea mays* L.) and soybean [*Glycine max* (L.) Merr.] were grown in rotation with N fertilizer applied in the spring to corn only. For the L treatment, NO_3 concentrations in the drainage water exceeded the 10 mg N L^{-1} maximum contaminant level (MCL) established by the USEPA for drinking water only during the years that corn was grown. For the M and H treatments, NO_3 concentrations exceeded the MCL in all years, regardless of crop grown. For all years, the NO_3 mass loss in tile drainage water from the H treatment (48 kg N ha^{-1}) was significantly greater than the mass losses from the M (35 kg N ha^{-1}) and L (29 kg N ha^{-1}) treatments, which were not significantly different. The economically optimum N fertilizer rate for corn was between 67 and 135 kg ha^{-1} in 1996 and 114 and 172 kg ha^{-1} in 1998, but the net N mass balance indicated that N was being mined from the soil at these N fertilizer levels and that the system would not be sustainable.

Excessive NO_3 contamination of surface waters can require costly treatment of water for human consumption and is implicated in the formation of a hypoxic zone in the Gulf of Mexico (Rabalais et al., 1996). Nitrate-contaminated drainage water from artificial subsurface drainage systems (*tiles*) is a primary source of NO_3 loadings to surface waters within the Midwest U.S. Corn Belt (David et al., 1997). For example, Jaynes et al. (1999) measured between 4 and 66 kg N ha^{-1} of NO_3 lost in the surface waters of a 5200-ha intensively farmed agricultural watershed. They attributed most of this loss to tile drains that outlet into the stream. While much of this loss of N is commonly attributed to overuse of commercial N fertilizer, one conclusion that can be drawn from studies by Keeney and DeLuca (1993) and Willrich (1969) is that considerable N loss was occurring before the widespread use of inorganic fertilizers and that N leaching loss is more a result of farming per se (drainage, tillage, row-cropping, etc.) than of irresponsible fertilizer use.

The relationship between corn yield and N fertilizer application rate has been the focus of intense research for at least 50 yr (Krantz and Chandler, 1954). Studies on N leaching losses and the relationship between N fertilizer application rate and NO_3 leaching, particularly in artificially drained soils, have been conducted over a much briefer time (Angle et al., 1993; Rasse et al., 1999). In one of the first controlled studies, Baker et al. (1975) found that the concentration of NO_3 in tile drainage water averaged 21 mg N L^{-1} and the losses averaged approximately 30 kg N ha^{-1} yr^{-1} for a N fertilizer application of 112 kg N ha^{-1} on corn grown in rotation with unfertilized oat (*Avena sativa* L.) or soybean. In continuous corn production, Randall and Iragavarapu (1995) found flow-weighted NO_3 concentrations during an 11-yr period of applying 200 kg N ha^{-1} to average 13.4 and 12.0 mg N L^{-1} for conventional-tillage and no-tillage systems, respectively. In comparing the effect of N fertilizer rate, Baker and Johnson (1981) found that increasing the fertilizer rate from 100 to 250 kg N ha^{-1} on corn, grown in rotation with either soybean or oat, doubled the NO_3 concentration in tile drainage from 20 to 40 mg N L^{-1}. Similar results have been reported by Gast et al. (1978) for N fertilizer applied to continuous corn.

These studies have provided valuable information on the effect of N fertilizer rates on NO_3 leaching. However, studies have been conducted on only a limited number of locations and typically on small, intensively managed agricultural plots (~200 to 300 m^2) rather than production fields. Our objective was to quantify the effects of N fertilizer rate on corn and soybean yields and NO_3 losses in subsurface drainage water within a production field on highly productive soils not previously studied.

MATERIALS AND METHODS

The research was conducted on a 22-ha privately owned field in central Iowa, chosen for its uniformity of soils and terrain (Brevik et al., 2001) and the presence of an existing pattern-tiled drainage system. Soils within the field are in the Kossuth (fine-loamy, mixed, mesic Typic Endoaquoll)–Ottosen (fine-loamy, mixed, superactive, mesic Aquic Hapludoll) association. Harps (fine-loamy, mixed, superactive, mesic Typic Calciaquoll) and a small area of Okoboji (fine, smectitic, mesic Cumulic Vertic Endoaquoll) soils are also included (Fig. 1). These soils were formed on nearly level, alluvial or lacustrine sediments, range from very poorly to somewhat poorly drained, and are high in soil organic carbon content compared with most Midwestern soils (Table 1). Large-scale row crop agriculture on these soils was possible only after installation of subsurface drainage systems (Hewes and Frandson, 1952).

In 1992, the field had new subsurface drainage lines installed at a 1.45-m depth. Plastic drain pipe (10.2-cm diam.) was installed in parallel lines, 500 m long and either 27.4 or 36.5 m apart (Fig. 1). For this study, 9 of the 12 tile lines were intercepted before intersecting the collection lateral on the

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Abbreviations: H, high nitrogen fertilizer treatment; L, low nitrogen fertilizer treatment; M, medium nitrogen fertilizer treatment; MCL, maximum contaminant level.

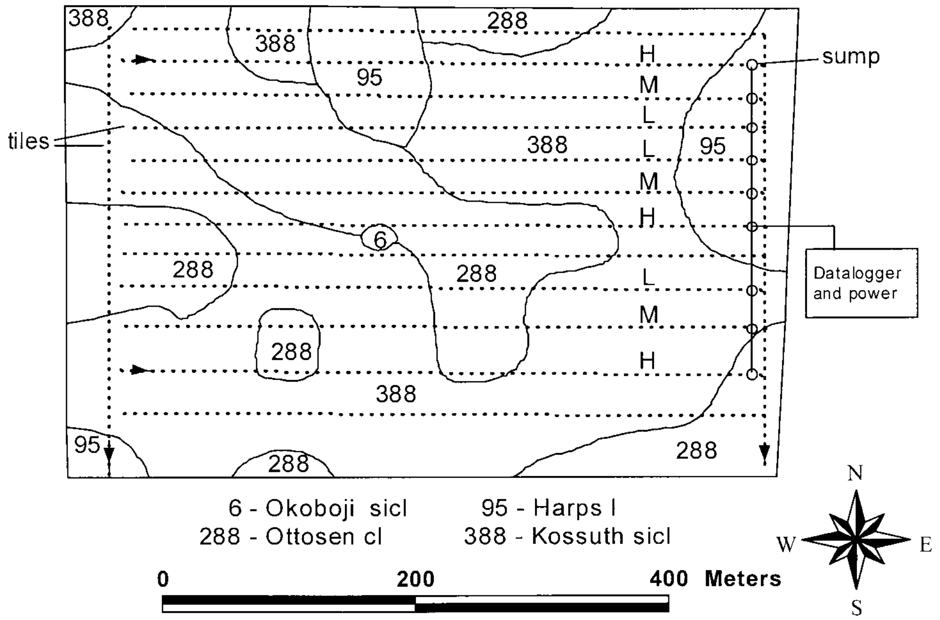

Fig. 1. Schematic of the field showing layout of tile drains, sumps, locations of high (H), medium (M), and low (L) nitrogen fertilizer treatment plots, and distribution of Okoboji silty clay loam, Harps loam, Ottosen clay loam, and Kossuth silty clay loam soils as determined by Brevik et al. (2001).

east side of the field. A 0.6-m-diameter corrugated plastic culvert was installed vertically at each intersection as a sump. Drainage was pumped from each sump into the collection lateral with a submersible sewage ejector pump equipped with a high/low level shutoff switch. Flow volume vs. time was measured with an FP-5300 paddle wheel flow meter (Omega, Stamford, CT[1]) and recorded with a CR10X datalogger (Campbell Scientific, Logan, UT). Cumulative drainage for each tile was calculated by summing the discharge volumes over time from the start of each year and dividing by the area of each plot. The plot areas were assumed equal to the length of the tile lines multiplied by the distance separating midpoints between the parallel tiles. Rainfall since 1996 was measured with a tipping bucket rain gauge and recorded every hour at a location less than 0.5 km from the field.

Flow-weighted composite water samples were collected in glass jars connected by a siphon tube to the sump pump. Water samples were returned to the laboratory on a weekly or shorter basis, depending on tile flow rate, and refrigerated until analysis. Water samples were analyzed for NO_3 using a Lachat Autoanalyzer (Zellweger Analytics, Lachat Instrument Division, Milwaukee, WI). Nitrate was quantitatively reduced to nitrite and the nitrite concentration determined colorimetrically. The method quantitation limit was 1.0 mg N L^{-1} as NO_3. Nitrate mass loss in tiles was calculated by multiplying the NO_3 concentration for the composite sample times the volume of water discharged during the time the composite sample was collected.

The field was planted to corn in 1995, 1996, and 1998 and soybean in 1997 and 1999 (Table 2). Prior to this time, the field had been in a typical 2-yr corn–soybean rotation. Primary tillage consisted of either moldboard or chisel plowing. A field cultivator was used to prepare the soil for planting in the spring and the field was cultivated with a row crop cultivator several times during the early growing season for weed control.

[1] Names are necessary to report factually on available data; however, the USDA neither guarantees nor warrants the standard of the product, and the use of the name by USDA implies no approval of the product to the exclusion of others that may also be suitable.

Plant counts were 66 000 and 75 000 ha^{-1} for corn in 1996 and 1998, respectively and 370 000 ha^{-1} for soybean in 1997 and 1999. All operations other than nitrogen fertilization and harvesting were performed by the farmer–owner as part of his normal production practices.

The nine tiles served as the center lines for nine plots, which we grouped into three blocks and randomly assigned three N fertilizer rates within each block. In 1996, anhydrous ammonia was injected 1 wk before corn planting at rates of 202, 135, and 67 kg ha^{-1} for the high (H), medium (M), and low (L) treatments (Fig. 1), respectively, where the H rate was equivalent to the farmer's normal practice. Prior to the start of this experiment, anhydrous ammonia was injected 1 wk before corn planting in 1995 at the same rates as in 1996 to seven of the nine plots, with the H and L treatments being switched on the south end of the field in the other two plots (Fig. 1). Three weeks after planting in 1998, we attempted to apply the same N rates to the same plots as in 1996 using 32% liquid urea ammonium nitrate (UAN) for better uniformity of application (Weber et al., 1995). However, slippage of the metering wheel caused underapplication of fertilizer, and resulting N rates were 172 (H), 114 (M), and 57 (L) kg ha^{-1}. No N fertilizer was applied to soybean in either 1997 or 1999.

Grain yield was measured on a transect adjacent to the drain line within each of the nine subsurface drainage plots with either a modified Gleaner K combine (Allis–Chalmers, Milwaukee, WI) or a modified John Deere (Moline, IL) 4420 combine (Colvin, 1990). The transect was offset from the drain line by about 3 m to avoid the soil disturbed by tile installation, but the location was the same each year. Along a transect, a

Table 1. Average soil properties with depth.

Depth	Bulk density	Sand	Silt	Clay	Organic carbon
m	Mg m^{-3}	g kg^{-1}			
0–0.15	1.16	220	330	450	29
0.15–0.3	1.22	210	330	460	27
0.3–0.6	1.27	220	320	460	13
0.6–0.9	1.48	470	290	240	4
0.9–1.2	1.56	350	400	250	1

Table 2. Field management practices for 1996–1999.

	1996	1997	1998	1999
Crop	corn†	soybean	corn	soybean
Spring tillage	field cultivator	field cultivator	field cultivator	field cultivator
Planting	late April	early May	late April	early May
N fertilizer application	18 April	none	14 May	none
Harvest date	2 November	1 October	21 September	21 September
Prior fall tillage	moldboard	moldboard	chisel	chisel

† Third-year corn.

20-m length was harvested, the combine stopped to allow grain to finish cycling through the combine, and the grain was weighed and moisture content measured. A three-row-wide strip for corn and a five-row-wide strip for soybean was harvested along each transect. A total of 225 yield values were collected by making 25 contiguous yield measurements on each transect. All grain weights were adjusted to a moisture content of 155 g kg^{-1} for corn and 130 g kg^{-1} for soybean. In 1998 and 1999, grain samples from each plot were collected and grain protein determined using near-infrared spectroscopy at the Iowa State University Grain Quality Laboratory.

Soil cores were taken randomly after harvest from each N treatment plot on 13 Nov. 1996, 1 Oct. 1997, 25 Oct. 1998, and 3 Nov. 1999. The soil cores were taken to a depth of 1.2 m by pushing a 38.1-mm-diameter steel soil probe, fitted with a removable acetate liner, into the soil with a hydraulic ram. The soil core and liner were removed from the steel probe, capped on each end, and stored at −10°C until NO_3 extraction. The frozen soil cores were cut into 150-mm-long sections, removed from the liners, thawed, and mixed by hand. Three 20-g subsamples were taken for determination of soil water and NO_3 content. Water content was determined by change in weight from drying a soil sample at 104°C for 48 h. Nitrate concentrations were measured colorimetrically (Keeney and Nelson, 1982) using flow injection analysis technology and Manifold no. 12-107-04-01-B (Lachat Instruments, Milwaukee, WI).

Exploratory analysis was conducted on all data sets using SAS (SAS Institute, Cary, NC). Nonparametric methods were used where appropriate when distributions failed the Kolmogorov–Smirnov test of normality (Mood et al., 1974). For tile NO_3 concentration data, conventional analysis of variance was explored initially, but diagnostics revealed several problems. The time series records constructed from the treatment means were not stationary and each detrended series was autocorrelated. Hence, an autoregressive time response curve methodology was employed. Following Meek et al. (2000), models were developed for the differences H-L, H-M, and M-L in the paired NO_3 sequences. For each contrast, the difference was modeled with splined polynomial segments. Time was cast as days past the first recorded tile flow (9 Apr. 1996). Polynomial terms included up to quartic powers of time. A single knot was fixed at 600 d past the onset of treatments. Up to two autoregression terms were selected from the set of terms that included up to the first eight lags. The 95% confidence interval was estimated over the entire period of comparison and used to determine the periods of significant difference by using inclusion of the zero line (no difference) within the confidence band as the indicator.

RESULTS

Hydrology

Rainfall for the four years varied from 703 mm in 1997 to 1019 mm in 1996 (Fig. 2), which compares with the 30-yr average of 818 mm for this area (Hatfield et al., 1999). Precipitation occurred in all months, but the April through July period was the wettest period in every year. Evidence of runoff from the field was observed in response to only a few isolated storms during the 4-yr period, notably after a 161-mm rainfall on 16 June 1996, but runoff was not measured.

Tile flow rates varied from 0 (not flowing) to more than 200 m^3 d^{-1} in response to rainfall and water table rise. Figure 3 shows the flow rate and cumulative drainage for the 1996 through 1999 period for one of the M treatment tiles. Annual discharge from the nine tiles ranged from 201 to 333 mm in 1996, 103 to 215 mm in 1997, 262 to 406 mm in 1998, and 264 to 408 mm in 1999. No significant difference ($P = 0.05$) was found in tile drainage volumes due to N treatments (data not shown). Drainage as a percentage of rain varied greatly in response to total rainfall and its distribution pattern throughout the year. On average, drainage represented 28% of the total precipitation in 1996, 18% in 1997, 35% in 1998, and 40% in 1999. The low drainage percentage in 1996 was the result of rain running off rather than infiltrating the field for the 161-mm June rainfall event. The low drainage percentage in 1997 was the result of overall low total precipitation providing little excess water after losses to evapotranspiration (Hatfield et al., 1998). Although the annual discharges were very similar in 1998 and 1999, drainage as a percentage of precipitation was greater in 1999 due to the rain falling early in the season before the crop started growing. Most tile drainage occurred before July in 1997, 1998, and 1999. In 1996, drainage was about equally divided between the May–June period and November–December. Drainage ceased from every tile in mid-summer of each year in response to decreasing precipitation and increasing transpiration from the rapidly growing crops.

Nitrate in Tile Water

Nitrate concentrations in the tile drainage water samples ranged from 5 to 45 mg N L^{-1} during the 1996 to 1999 period. When averaged by month and N fertilizer treatment, NO_3 concentrations ranged from 6 to 28 mg N L^{-1} (Fig. 4). Breaks in the data shown in Fig. 4 represent times when no drainage flow occurred. Except for the L fertilizer treatment in 1997 and 1999 when soybean was grown and no nitrogen fertilizer applied, NO_3 concentrations in the tile drainage consistently exceeded the 10 mg N L^{-1} maximum contaminant limit (MCL) set by the USEPA for drinking water. In 1996, NO_3 concentrations peaked in mid-May following the application of fertilizer and coincident with the maximum drainage discharge period. Concentrations decreased

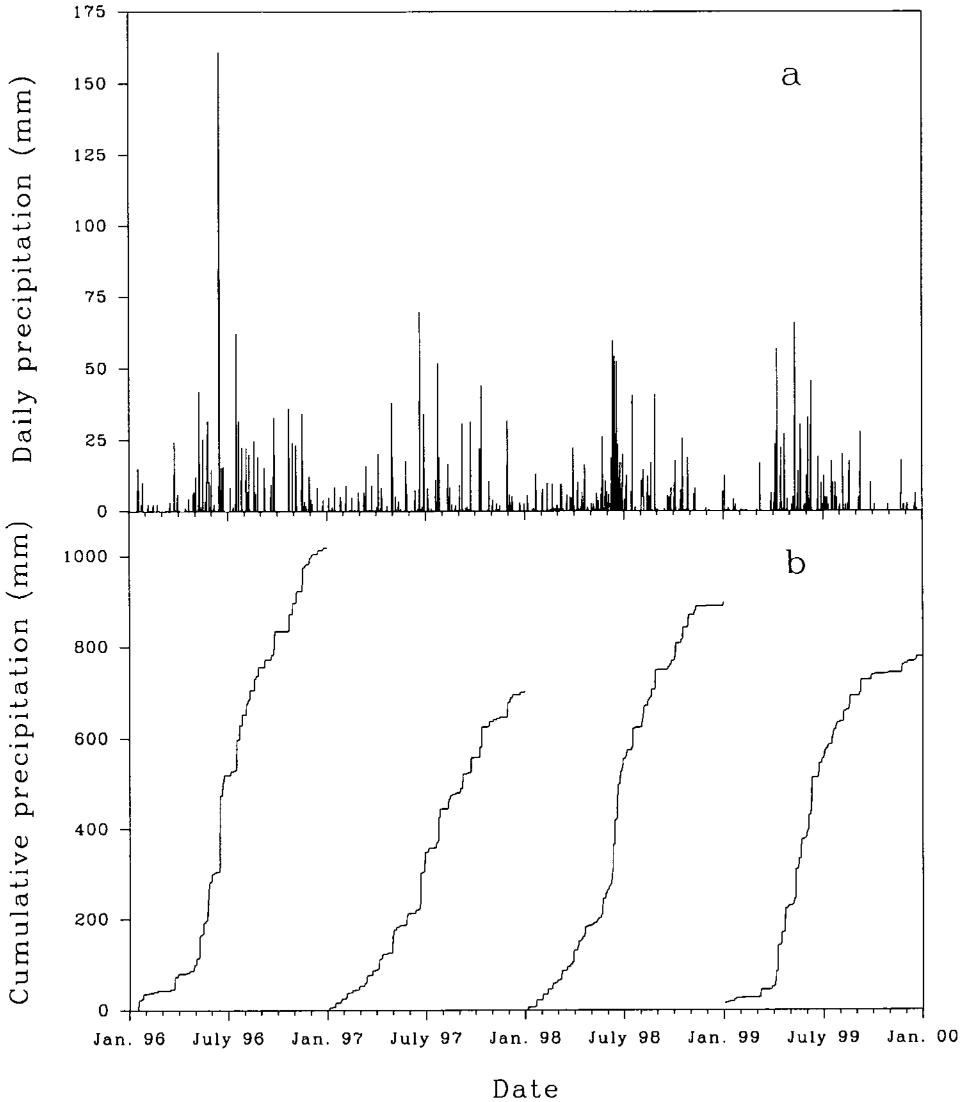

Fig. 2. Daily and annual cumulative precipitation for years 1996 through 1999.

during the summer, then increased again in the fall following harvest in response to greater fall rainfall and increased drainage that year. The NO_3 concentrations then decreased slowly throughout 1997 during soybean production when no N fertilizer was applied. Nitrate concentrations increased again in the summer of 1998 following N fertilization of the corn crop, but did not show a marked peak as in 1996. Concentrations reached their maximum in July 1998 then slowly decreased throughout the remainder of the year, continuing to decrease in 1999 when soybean was grown.

Conventional analysis of variance of the tile drainage NO_3 concentration data indicated that the residuals were autocorrelated and that there was a considerable time by treatment interaction. Thus, autoregressive time response curve methodology was employed to model the differences in NO_3 concentration in tile drainage from each treatment through the 4 yr. For the H–L and M–L differences in NO_3 concentrations, only the first lag term was significant and retained in the model, while for H–M differences, both the first and fourth lags were retained. Figure 5 shows the resulting model and its $P = 0.95$ confidence limits for the differences in NO_3 concentration for the mean H–M, H–L, and M–L treatment comparisons. Where the confidence bands do not overlap the 0.0 value (Fig. 5), the treatments can be considered significantly different (i.e., H_o, difference in NO_3 concentration = 0 is false). From about mid-1996 through the end of 1999, there was a significant difference in NO_3 concentrations between the H and M treatment of about 5 mg N L^{-1} (Fig. 5a). Near the end of 1999 when soybean was grown, the difference in NO_3 concentrations decreased slightly and was no longer significant.

Differences between the H and L treatment NO_3 concentrations were similar to the H–M concentration differences with a short initial phase in 1996 and a period before tile flow ceased in 1999 when they were not significantly different (Fig. 5b). In between these times, the H treatment NO_3 concentrations were significantly higher than the L treatment concentrations, averaging about a 10 mg N L^{-1} difference in the second half of

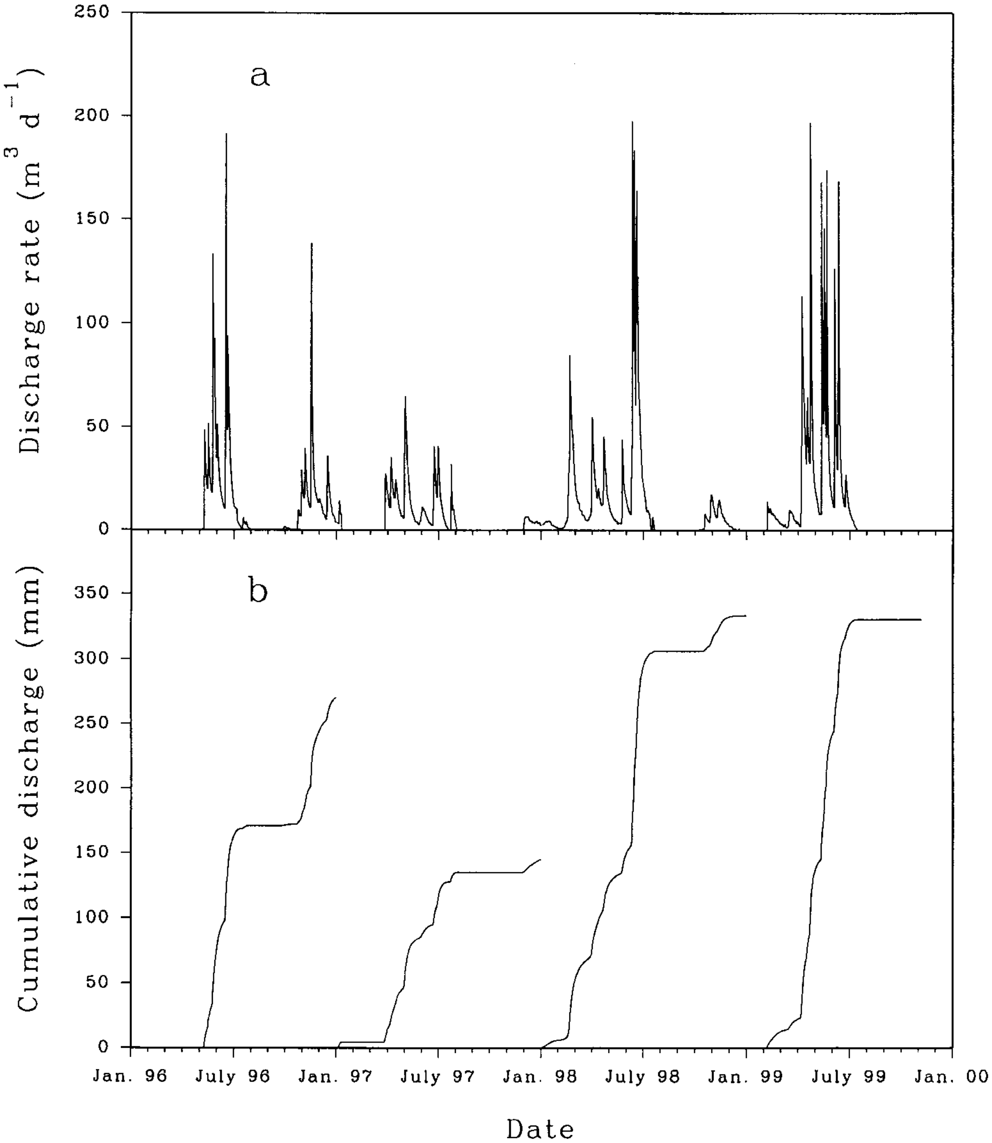

Fig. 3. Example of water discharge rate and cumulative discharge depth from a tile drain for one of the medium (M) nitrogen fertilizer treatment plots for 1996 to 1999.

1996 and decreasing to about a 5 mg N L^{-1} difference in the spring of 1999. Nitrate concentration differences between the M and L treatments also showed a similar startup phase when they were not different, followed by a year-long period lasting until mid-1997 when the M treatment concentrations were significantly higher than the L treatment concentrations (Fig. 5c). After a cessation in tile flow, the tiles again started flowing in the fall of 1997 after soybean harvest, but the NO_3 concentrations for the M and L treatments were no longer different. This period continued through mid-1998 when the planting of corn and the addition of N fertilizer again caused the M–L treatment differences to be significant. Nitrate concentrations remained different until mid-1999 when the difference decreased during the soybean year until no longer significant. Thus, the NO_3 concentrations in the tile drainage showed significant differences between treatments, but these differences varied with season, apparently in response to crop and N fertilizer use and precipitation patterns. Differences in NO_3 concentrations tended to disappear, especially between the M and L treatments, in the years that soybean was grown and no N fertilizer applied.

Yearly mass of NO_3 lost in the tile water was computed by multiplying the flow volume during composite sample collection by the average NO_3 concentration for that time period and summing over the year. A simple analysis of variance (ANOVA) comparison of the mass loss by treatment and block for all years showed no significant difference ($P = 0.05$) between blocks and a significant difference between the mass loss from the H treatment (48 kg N ha^{-1}) versus the M (35 kg N ha^{-1}) and L (29 kg N ha^{-1}) treatments, but no difference between the M and L treatments. Average yearly mass losses of NO_3 by treatment (Table 3) varied from 13 kg ha^{-1} for the L treatment in 1997 to 61 kg ha^{-1} for the H treatment in 1996. The greatest mass losses were in 1996 and 1998, the years that corn was grown and N

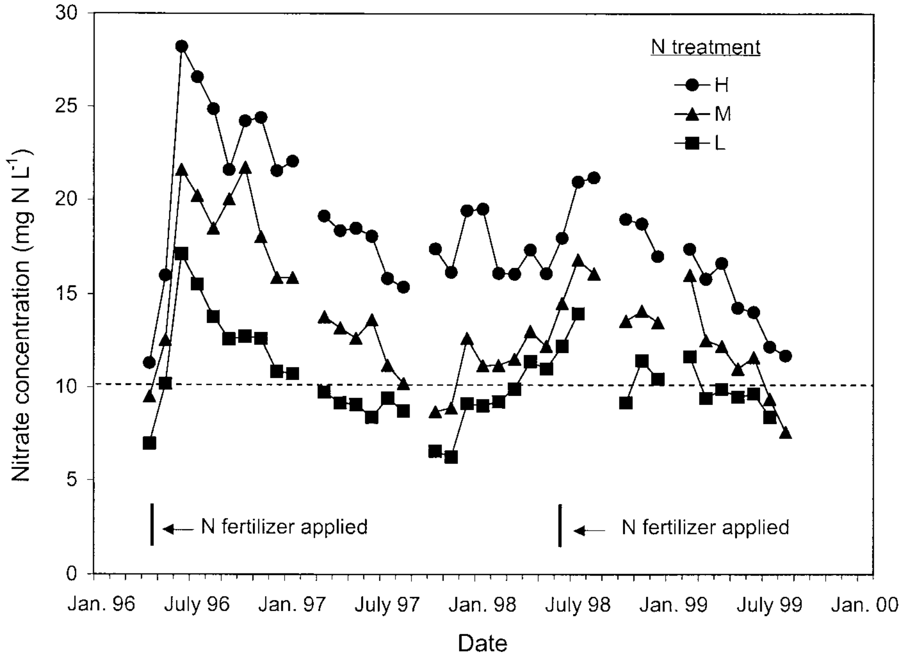

Fig. 4. Nitrate concentrations in tile drains averaged by month and N fertilizer treatment versus time from 1996 through 1999.

fertilizer applied. However, these were the same years having the greatest tile discharge volumes (Fig. 3), which accounts for some of the differences in mass loss among years.

End-of-Season Residual Soil Nitrate

Soil cores were taken after harvest each year and analyzed for residual NO_3. From 1997 to 1999, more than half of the residual NO_3 was recovered in the top 30 cm of the soil (Table 4). This was probably a result of mineralization of soil organic matter that is concentrated in the surface layers. In 1996, the surface horizons were depleted in NO_3 in comparison with the other years, but the soil layers deeper than 90 cm, especially for the H treatment, had a relatively greater mass of NO_3. Apparently, NO_3 was leached to deeper depths in 1996 as a consequence of the considerably greater precipitation and tile drainage that occurred in October and November in 1996 compared with the other years (Fig. 2 and 3).

Nitrate recovery from the 1.2-m soil profile was highly variable within each treatment, resulting in only the H treatment in 1996 having significantly higher residual soil NO_3 than the other treatments within a year. Among years, residual soil NO_3 was variable with significantly more NO_3 in the soil profile following corn in 1998 than after corn in 1996 or soybean in 1997 and 1999. Soybean production had no consistent effect on residual soil NO_3 at the end of the season compared with corn. Thus, fall NO_3 recovery in soil was not consistently related to fertilizer use or crop grown, but was apparently influenced by late-season rainfall and leaching. This is agreement with the findings of Randall et al. (1997), who found no consistent crop effect in a corn–soybean rotation on end-of-season residual NO_3 in the surface 1.5-m soil layer, and marked variability in residual NO_3 levels in response to variations in yearly precipitation amounts. The residual soil NO_3 levels found for this field were 2 to 10 times lower than residual NO_3 levels found in the corn–soybean rotation in Minnesota (Randall et al., 1997) for N fertilizer, and corn yields were similar to those presented here. Residual soil NO_3 levels were lower for this field even when compared with continuous corn plots receiving no N fertilizer (Randall and Iragavarapu, 1995). Reasons for the lower residual soil NO_3 levels in this field are not known, but may be due to more N being removed by the higher soybean yields in this study than in the Minnesota study (~500 kg ha^{-1}).

Grain Yield

Grain yield distribution for each year was negatively skewed and failed normality tests, thus nonparametric tests were used to compare the treatments. Nitrogen fertilizer treatment had a significant affect on yield for both years that corn was grown (Fig. 6). In 1996, the corn yield for the L treatment was significantly lower than the M and H treatments, while the M and H treatment yields were not significantly different. In 1998, corn yields for each treatment were significantly different.

In contrast to corn yields, soybean yields were not significantly different in either year, with soybean yields in 1999 ~4% higher than yields in 1997. Thus, there was no carryover effect of N fertilizer treatment on soybean yields, which is in agreement with other N rate studies (Stone et al., 1985; Bundy et al., 1993).

Following Cerrato and Blackmer (1990), we can assume a quadratic plus plateau model for the response of corn yield to N fertilizer treatment and calculate a fertilizer to corn price ratio of 4.5 using prices of \$0.40 kg^{-1} for fertilizer N (\$0.18 lb^{-1}) and \$88.40 Mg^{-1} for corn grain (\$2.25 bu^{-1}). For the price ratio of 4.5, the economic optimum N fertilizer rate in 1996 was between

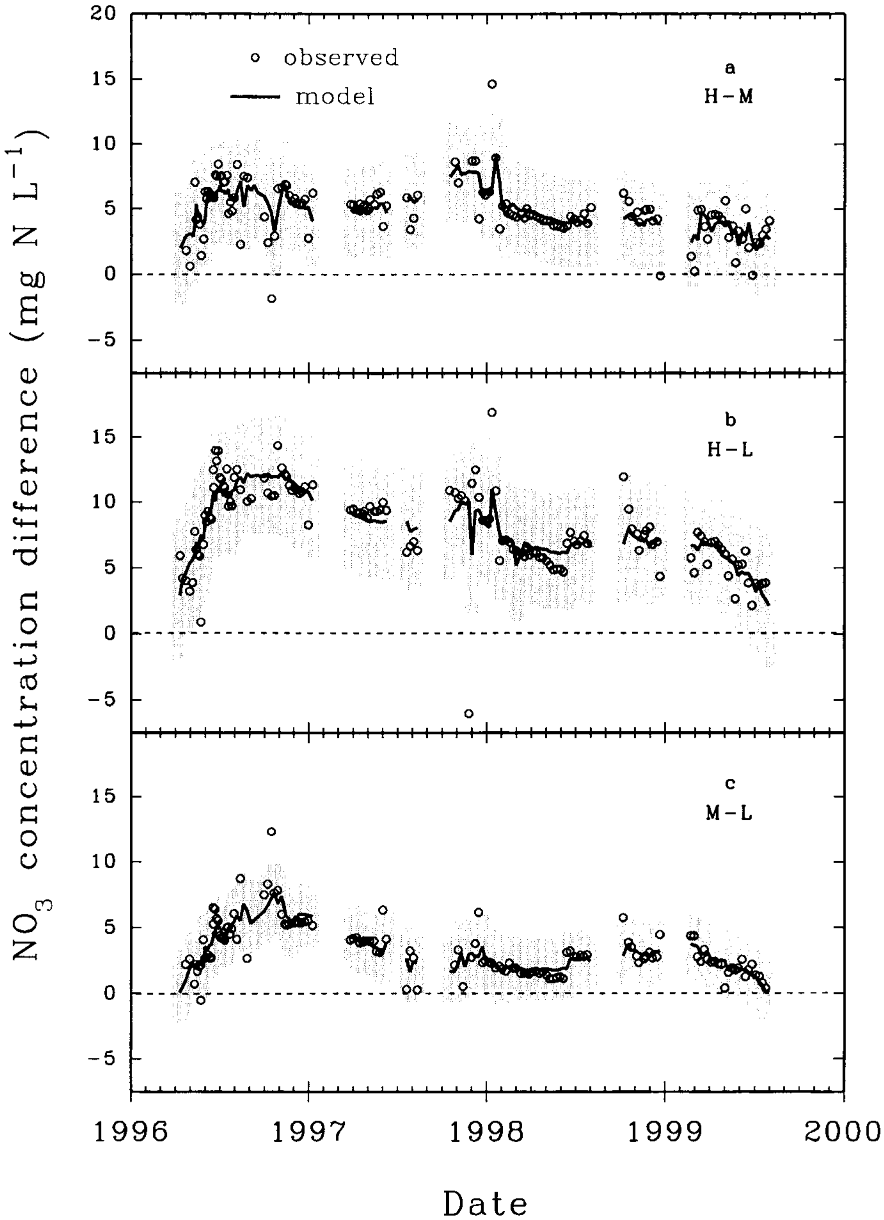

Fig. 5. Observed and modeled differences between mean NO_3 concentrations in tile drainage from plots receiving high (H), medium, (M), and low (L) nitrogen fertilizer treatments starting in spring 1996. Shaded area represents the 95% confidence limits of the modeled NO_3 differences.

67 (L) and 135 (M) kg ha^{-1} and between 114 (M) and 172 (H) kg ha^{-1} in 1998. However, even at the lowest N fertilizer rate, the NO_3 concentration in the tile water still exceeded the 10 mg N L^{-1} MCL when corn was grown (Fig. 6). In the years that soybean was grown, NO_3 concentrations in the tile water exceeded the MCL for all but the L treatment. The average NO_3 concentration in the drainage increased with increasing N fertilizer rate, and while the addition of N fertilizer produced a curve of diminishing returns for corn yield, it produced a curve of increasingly greater NO_3 concentrations in drainage.

Nitrogen Mass Balance

We can calculate a partial mass balance of N within the field for each year (Karlen et al., 1998) by assuming the conservation of mass:

$$\sum \text{inputs} - \sum \text{outputs} - \text{change in soil residual } NO_3 = \text{residual}$$

A residual > 0 would indicate that inputs of N exceed losses from the field and N is available for other processes such as increasing soil organic matter. A residual < 0 would indicate that inputs do not balance outputs and that additional N must be coming from sources not included in the calculation to account for the observed losses. A residual = 0 would indicate that the field is in balance between N inputs and outputs and thus the production system is sustainable from a N perspective.

Inputs of N included in the computation were the application of fertilizer, N contained in rain, and N fixed by soybean (Table 3). Fertilizer inputs were known.

Table 3. Partial mass budget for N covering four years of a corn–soybean rotation in a 22-ha production field.

Year and treatment	Fertilizer applied	Fixed†	Added in precipitation	Change in residual soil NO_3	Crop removal	Removed in tile drainage	N balance residual
	kg N ha^{-1}						
1996—Corn							
L	67	0	15	–‡	53	37	−8
M	135	0	15	–	93	47	10
H	202	0	15	–	93	61	63
1997—Soybean							
L	0	187	11	2	196	13	−13
M	0	195	11	9	198	16	−17
H	0	191	11	8	197	26	−29
1998—Corn							
L	57	0	13	12	69	38	−49
M	114	0	13	0	96	43	−12
H	172	0	13	4	108	59	14
1999—Soybean							
L	0	208	12	−8	204	31	−7
M	0	202	12	4	203	36	−29
H	0	203	12	−19	208	49	−23

† Calculated from $N_{fixed} = 81.1 \times$ yield (Mg ha^{-1}) $- 98.5$.
‡ Soil NO_3 in previous year not measured.

Nitrogen additions with precipitation were calculated using measured precipitation data and the average NO_3 concentration in rain of 1.5 mg N L^{-1} measured by Hatfield et al. (1996) for central Iowa. While these calculations ignore contributions from other forms of N, the resulting estimates are greater than atmospheric deposition rates measured for the Midwest by the National Atmospheric Deposition Program for NO_3 and NH_4 combined (Burkart and James, 1999). Nitrogen fixation was calculated using the relationship between soybean grain yield (Mg ha^{-1}) and N fixed (N_{fixed}) given by Barry et al. (1993):

$$N_{fixed} = 81.1 \times \text{yield} - 98.5 \quad [1]$$

where yield is at 140 g kg^{-1} moisture content. Calculated N inputs from N fixation by soybean in 1997 and 1999 were of the same magnitude as N fertilizer inputs for the H treatment in 1996 and greater than the N fertilizer input in the H treatment in 1998 (Table 3).

Outputs of N included N removed with the grain harvest and NO_3 in tile drainage. Protein content of grain was measured in 1998 for corn to be 64, 70, and 74 g kg^{-1} for the L, M, and H treatments, respectively. We used these values and an assumed protein to N ratio of 6.25:1 (David et al., 1997) to compute N removed in corn. Measured protein content of soybean for 1999 was 395 g kg^{-1} for all treatments and was used to compute N removal with soybean for 1997 and 1999 assuming a protein to N ratio of 6.25:1. For each treatment and year, N loss in tile drainage represented about one-third of the total N loss and grain removal about two-thirds of the total loss, which are about the same proportions found for a 48 173-ha watershed of the Embarras River in Illinois (David et al., 1997).

Not accounted for in the partial N balance were N inputs from wet and dry deposition of N forms other than NO_3, weathering of the soil mineral fraction, or decomposition of soil organic matter or crop residues. Nitrogen outputs not considered were N in deep percolation below the tiles, N loss in surface runoff, and N volatilized from the soil or plant (Francis et al., 1993, 1997) or through denitrification (Parkin and Meisinger, 1989).

Annual changes in NO_3 stored in the soil were calculated for 1997 through 1999 by subtracting the residual soil NO_3 mass measured the previous fall from the mass measured in the fall of the year in question. Change in residual soil NO_3 was not calculated for 1996 because residual soil NO_3 was not measured after harvest in 1995. Losses of NH_4, NO_2, and other forms of N in tile drainage were considered negligible as most N in tile drainage has been found to be in the NO_3 form (Willrich, 1969). Changes in N due to changes in crop residues were considered to be negligible because crop residues

Table 4. Postharvest soil NO_3 content by sampling date and N fertilizer treatment.

	13 Nov. 1996				1 Oct. 1997				25 Oct. 1998				3 Nov. 1999			
Depth	H	M	L	Avg.	H	M	L	Avg.	H	M	L	Avg.	H	M	L	Avg.
cm	kg N ha^{-1}															
0–15	6	4	5		15	10	8		18	11	14		12	18	13	
15–30	7	4	5		12	7	7		13	6	10		6	5	6	
30–45	4	2	3		5	2	2		5	3	5		2	2	2	
45–60	2	2	2		2	2	1		2	2	3		1	1	2	
60–75	2	1	1		1	1	1		1	1	2		1	1	1	
75–90	2	1	1		2	1	1		1	1	1		1	1	1	
90–105	5	1	1		1	2	1		1	1	1		1	1	1	
105–120	7	3	3		4	2	1		4	3	1		2	2	1	
0–120	34†	17	22	24a‡	42	26	24	31ab	46	26	36	39b	26	30	28	26a

† Only the 1996 H treatment was significantly different ($P <$ 0.05) than other treatments within a year.
‡ Average values followed by same letter are not significantly different ($P < 0.05$).

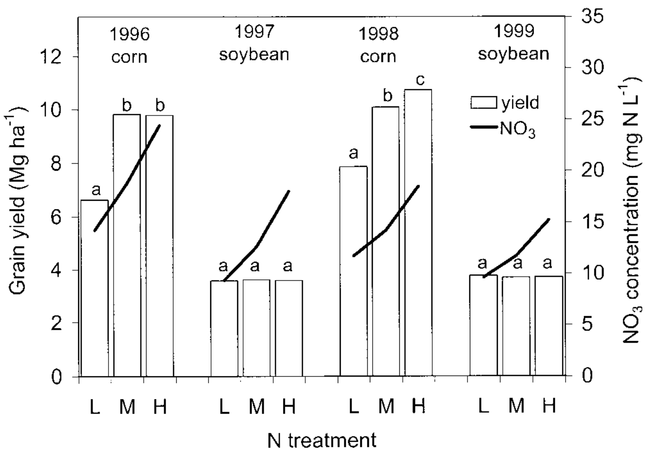

Fig. 6. Average grain yield and flow-weighted concentration of NO_3 in tile drains for 1996 through 1999. Yield bars followed by same letter within a year are not significantly different (P = 0.05).

should be stable over the rotation if biomass production is consistent over time. Changes in N due to changes in soil organic matter were also not considered and probably represent the largest source of error in the mass balance given the large reserve of N represented by the soil organic fraction in the field (>17 000 kg N ha^{-1}) and the potential effect small changes in this reserve would have on the annual N budget.

The mass balance residual for N during corn years indicates a net loss of N from the field for the L treatment in 1996 and 1998 (Table 3). That is, the L treatments must be consuming N from an unaccounted source such as soil organic matter to produce the grain yields and leaching losses observed. As such, the L fertilizer rate would not be sustainable because of long-term net losses of soil organic matter and ultimately yield.

For the M treatment in 1996, the N mass balance residual was positive, indicating greater N inputs than losses. In 1998, the M treatment resulted in a negative N mass balance residual, indicating that more N was lost than added. The H treatment resulted in a positive N mass balance for both years, indicating a surplus of N available to the soil. Thus, a N fertilizer rate between 114 to 135 kg ha^{-1} is required to balance the inputs and outputs of N with the current yields and drainage losses observed for this system.

The mass balance residual for N during the soybean years was always negative, indicating a greater loss of N than known inputs. Although computed N fixed by soybean was nearly 200 kg ha^{-1} in 1997 and 1999, this input of N was more than offset by N removal with soybean grain and in tile drainage. These mass balance calculations for soybean are in agreement with observations in other studies (Harper, 1974; Peoples and Craswell, 1992; Barry, et al., 1993; Vanotti and Bundy, 1995; David et al., 1997). Thus, although it is common to speak of an equivalent N fertilizer credit of ~40 kg ha^{-1} after growing soybean for grain, the mechanism for this credit appears to be something other than an overall increase in soil N (Vanotti and Bundy, 1995; Green and Blackmer, 1995).

For the 2-yr corn–soybean rotation used here (1996–1997 and 1998–1999), the net mass balance residual for N is negative for all treatments except H in 1996–1997. Thus, for the rotation, all treatments (except possibly H) are not sustainable from a N balance perspective and are probably resulting in oxidation of soil organic matter to supply the missing N.

SUMMARY AND CONCLUSIONS

Drainage water quality and quantity and crop yield in a 22-ha field having three N fertilizer treatments (L, M, H) were measured for 4 yr in a corn–soybean rotation. The results showed that even for the L nitrogen fertilizer treatment, NO_3 concentrations in the drainage water exceeded the MCL for drinking water in years corn was grown and in the M and H nitrogen fertilizer treatments in all years. Nitrate concentrations in tile drainage and mass losses were significantly different for the three treatments, but differences varied over time in response to precipitation, cropping, and N fertilizer input patterns.

Economic optimum N fertilizer rates for corn grain production were between the L and M treatments (67 to 135 kg ha^{-1}) in 1996 and between the M and H treatments (114 and 172 kg ha^{-1}) in 1998. Soybean grain production was unaffected by N fertilizer treatments during the corn year of the rotation. To balance the inputs and outputs of N to the system for the 2-yr rotation, N fertilizer rates needed to be at least at the H rate (202 kg N ha^{-1} in 1996 and 172 kg N ha^{-1} in 1998) given the current tile drainage losses experienced by this system. However, NO_3 concentrations in tile drainage consistently exceeded the MCL for drinking water in the years corn was grown for all N fertilizer treatments and also during the years soybean was grown on the M and H treatments.

Thus, it appears that economic corn production cannot be sustained within this field under the current rotation and management scheme without producing tile drainage water that exceeds the MCL for NO_3. The problem is not simply one of N fertilizer use, but of a corn–soybean production system created by artificial soil drainage and intensive tillage. This result is similar to findings found by others studying corn production on different soils in the Midwest U.S. Corn Belt, both on smaller plots (Baker and Johnson, 1981; Drury et al., 1993; Randall and Iragavarapu, 1995; Kanwar et al., 1997) and on watersheds (David et al., 1997; Jaynes et al., 1999). Thus, NO_3 concentrations exceeding the MCL appear endemic to artificially drained soils cropped to corn within the Midwest.

Sustainable agriculture is defined as an integrated system of plant and animal production practices that will, over the long term, "enhance environmental quality and the natural resource base upon which the agricultural economy depends" (Section 1603 of the National Food, Agriculture, Conservation, and Trade Act of 1990). By including water quality as well as productivity as endpoints in the concept of sustainable agricultural production systems, dramatic changes will be required in management practices on artificially drained soils before corn production can be considered sustainable.

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