Initiation of Irrigation Effects on Temporal Nitrate Leaching

F. X. M. Casey,* N. Derby, R. E. Knighton, D. D. Steele, and E. C. Stegman

ABSTRACT

Groundwater and surface water are significant resources for rural water supplies, and certain agricultural practices may have substantial effects on these resources. An 11-yr study was started in 1989 near Oakes, ND that continuously monitored NO_3–N concentrations in subsurface water of a field that was converted from dryland to center-pivot irrigation in 1989. The vadose zone was monitored with four disturbed and 16 undisturbed-profile lysimeters, and the groundwater of the surficial aquifer was monitored with 18 sets of nested wells, which sampled shallow, intermediate, and deep depths. The depth to water table of the surficial aquifer was approximately 3 m and the saturated thickness extended to a depth of 7 m. Also, NO_3–N levels from two subsurface drains were monitored. The time series NO_3–N concentration data from each of the monitoring locations exhibited the similar three-phase trend where NO_3–N concentrations first increased, then decreased, and finally reached a steady-state level that was maintained. The first and second phases of this trend were shorter ($\sim$3 yr total) for the lysimeters and increased as the depth of observation increased (5 and 8 yr total for shallow and intermediate wells, respectively). Also, the peak NO_3–N concentration decreased as the observation went deeper into the profile (ranging from 150 mg L^{-1} in lysimeters, to 50 mg L^{-1} in shallow wells, and to 40 mg L^{-1} in intermediate wells). The NO_3–N levels in the deep wells averaged 0.48 mg L^{-1}, had a maximum of 1.59 mg L^{-1}, and exhibited a slight increase through time. The subsurface drainage NO_3–N levels were an average of 77% lower than the groundwater concentrations, which may have been caused by biotic and abiotic reduction. The increase in NO_3–N concentrations in subsurface waters as a result of the initiation of irrigation can be partially explained by the residual N in the soil from dryland agriculture. As soil moisture increased, the availability and mobility of nitrogen increased, which attributed to the flush of NO_3–N through the soil profile.

One hundred thirty-five million people in the USA, including 99% of its rural population, rely on groundwater for their drinking water (USEPA, 2000). It is necessary to understand the impact of irrigation on groundwater quality in rural areas where irrigated agriculture is significant. The initiation of irrigation may result in larger quantities of water moving through the soil profile. Concerns about subsurface water quality grow as more water moves through the soil profile because advective transport of pollutants (e.g., agricultural chemicals) may also increase. Applying excess nutrient fertilizers will directly affect subsurface water quality especially for NO_3–N, which is highly mobile.

The increase in soil moisture that results from the initiation of irrigation dissolves excess NO_3–N present in the soil profile from dryland agriculture and makes it more susceptible to leaching (Albus and Knighton, 1998). Higher moisture contents will also raise microbial activity including mineralization (Skopp et al., 1990). The increase in mineralization rates has been shown to directly affect nutrient leaching (Beck, 1983; Doran, 1980). Albus and Knighton (1998) found that the initiation of irrigation caused a flush of NO_3–N to the shallow groundwater, which raised concentrations to 40 mg L^{-1} from background levels. The background concentrations were near the USEA's drinking water Maximum Contamination Level (MCL) of 10 mg L^{-1}. Albus and Knighton (1998) found that the groundwater NO_3–N concentration remained elevated for nearly 4 yr.

In addition to groundwater degradation by NO_3–N, there is a risk of contamination to surface waters. Nitrate-contaminated water from subsurface drained watersheds is the primary source of NO_3–N loading to surface water within the Midwest (David et al., 1997). The excessive NO_3–N levels are not only costly to treat for human consumption, but they have been implicated in the formation of a hypoxic zone in the Gulf of Mexico (Rabalais et al., 1996).

Although there has been research on corn (*Zea mays* L.) yield and N fertilizer application rates for nearly half a century (Krantz and Chandler, 1954), there has been much less focus on the relationship between management practices and NO_3–N leaching (Angle et al., 1993; Rasse et al., 1999). By managing N fertilizer, Baker and Johnson (1981) found that increasing the rate from 100 to 250 kg ha^{-1} on corn grown in rotation with either soybean [*Glycine max* (L.) Merr.] or oat (*Avena sativa* L.) would double the NO_3–N concentration in subsurface drainage from 20 to 40 mg L^{-1}. Albus and Knighton (1998) showed that through proper N management, high N concentrations in shallow groundwater can be significantly lowered. Steele et al. (2000) also demonstrated that irrigation water management can be used to optimize corn yield, which can decrease the amount of NO_3–N leached by improving N uptake by corn. Nitrogen management practices can be developed to minimize adverse impacts on surface and subsurface water, while maintaining sufficient yields.

The objective of this research was to quantify trends in NO_3–N concentration in the root-zone, groundwater, and subsurface drain flow leaving a field after initiation of sprinkler-irrigated agriculture. Another objective of this research was to observe the effects of best management practices on the subsurface water quality. This research was part of a field-scale study of best management practices associated with N fertilizer management on irrigable soils representative of the Garrison Diversion Unit in southeastern North Dakota, USA. Several

F.X.M. Casey and N.E. Derby, Dep. of Soil Science; D.D. Steele and E.C. Stegman, Dep. of Agriculture and Biosystems Engineering, North Dakota State University, Fargo, ND 58105; R.E. Knighton, USDA-CSREES-Natural Resources and Environment, 1400 Independence Ave., SW- STOP 2210, Washington, DC 20250-2210. Received 25 Feb. 2002. *Corresponding author (francis.casey@ndsu.nodak.edu).

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Abbreviations: BMP, Best Management Practices [site]; MCL, maximum contaminant level; MSEA, Management Systems Evaluation Area; OITA, Oakes Irrigation Test Area.

other aspects of the field-scale study included the development and construction of disturbed and undisturbed-profile lysimeters for the measurement of groundwater quality (Derby et al., 1997; Derby and Knighton, 2001), collection of data sets for crop and environmental modeling (Steele et al., 1997), and irrigation management for corn (Steele et al., 2000).

MATERIALS AND METHODS

Research was done on a 65-ha field in Dickey County of North Dakota (Fig. 1; NW1/4, Sec.29, 46.0483° N, 98.1068° W) on a field site known as the Best Management Practices (BMP) site. This 65 ha site was part of a larger 2000 ha test area of the Garrison Diversion Unit, known as the Oakes Irrigation Test Area (OITA; Fig. 1). Approximately 53 ha of the 65-ha research area were irrigated by center pivot irrigation, which was initiated in 1989. The predominant soils of the BMP site are Hecla loamy fine sand (sandy, mixed, frigid Oxyaquic Hapludoll) and Wyndmere fine sandy loam (coarse-loamy, mixed, superactive, frigid Aeric Calciaquoll), both formed from glacial outwash parent material. Cropping history is provided in Table 1.

Steele et al. (2000) provided a detailed description of the irrigation methods used on the BMP field between 1990 and 1995. Different irrigation scheduling methods were used for each of the four quadrants beneath the center pivot irrigation unit, and each method was rotated among the quadrants for each year of the study. These irrigation methods were based on real-time soil matric potential measurements, two water balance methods, and irrigating based on estimates of plant-extractable soil water. After 1995, the management of irrigation was returned to the farmer-cooperator and we were not privy to the irrigation decision methods but only able to measure the irrigation. Rain gauge measurements were used to calculate the field average irrigation and precipitation (Table 1).

The N fertilizer applications were based on modified North Dakota State University extension recommendations (Dahnke et al., 1992). These N fertilizer recommendations were used from 1990 to 1995. The N fertilizer application for corn was

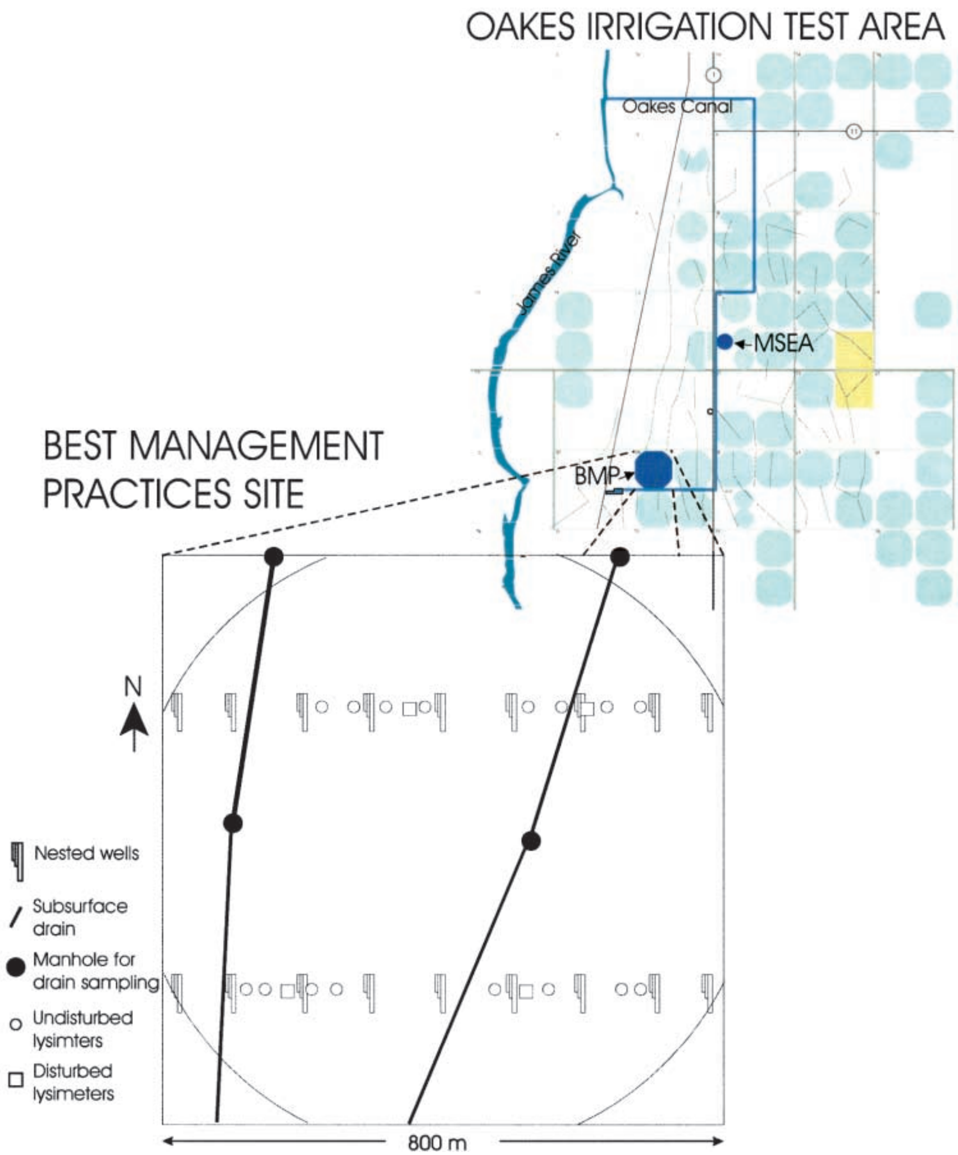

Fig. 1. A map of the 2000-ha Oakes Irrigation Test Area (OITA) with an exploded map of the 65-ha center-pivot irrigation Best Management Practices (BMP) research area. The BMP map indicates the type and location of the different instrumentation used to observe subsurface water quality. Light blue areas in OITA map are other irrigated fields.

Table 1. Cropping history, total N fertilizer amounts, and average irrigation plus rainfall amounts for the 65-ha research area for 11 yr starting with the initiation of irrigation.

Year	Crop	Average irrigation input amounts	Average precipitation from 1 May–30 Sept.	Total growing season water inputs	Applied N fertilizer
		mm			kg N ha^{-1}
1990	Corn	190	267	457	127
1991	Corn	139	335	474	112
1992	Corn	92	315	407	103
1993	Corn	57	443	500	165
1994	Corn	150	340	490	115
1995	Corn	175	314	489	179
1996	Potato	183	405	588	247
1997	Corn	179	273	452	191†
1998	Corn	111	489	600	204
1999	Potato	195	490	685	198
2000	Corn/soybean‡	205/146	337/337	512/483	251/11
Averages		149	364	513	172

† 201 kg N ha^{-1} was applied to the south half of the field in 1997.
‡ Corn on the south half, soybean on the north half of the field in 2000.

determined by supplying approximately 1.12 kg N ha^{-1} for every 62.8 kg ha^{-1} yield goal. The soil test N was subtracted from the amount to be applied. A grain yield goal of 9416 to 10 044 kg ha^{-1} and soil test information taken during preplanting and/or early crop development periods was used. At planting, diammonium phosphate was used as the N fertilizer and its application was minimized to <34 kg ha^{-1}. The primary N fertilizer was applied later at the six- to eight-leaf stage as side-dress anhydrous ammonia. The total yearly N fertilizer amounts are provided in Table 1. Nitrogen fertilizer application was done by hand above the lysimeter locations between 1990 to 1995 to achieve maximum accuracy. After 1995, the local farmer cooperator used his own N fertilizer rates, which were generally higher than the extension recommendations (Table 1). Also, after 1995 the majority of N was applied as side-dress anhydrous ammonia and additional N applied as urea ammonium nitrate through the sprinkler irrigator. The information from the Agricultural Practices Inventory (Esser and Weigel, 1997) was used to compare the BMP and OITA yields and N applications (D. Esser, Agricultural Practices Inventory, Garrison Diversion Conservancy District, Oakes, ND; personal communication for Agriculture Practices Inventory from 1996 to 2001).

In 1996 and 1999, when potato (*Solanum tuberosum* L.) was planted, the bulk of the N fertilizer was applied in the form of urea. At planting, approximately 11 kg ha^{-1} of N fertilizer as diammonium phosphate was broadcast on the soil surface and incorporated. When the crop was about 25% emerged, the primary N fertilizer application was broadcast on the field and incorporated during a hilling operation. A second broadcast of N fertilizer was applied and incorporated during the final hilling.

Soil cores were taken with a hydraulic probe each year, usually in the spring and again in the fall. Normally, two cores were taken adjacent to each lysimeter to a depth of approximately 1.8 m and divided into 0.3-m intervals. Depth increments were composited and a subsample was sealed in a thick-walled plastic bag. From 1990 to 1995, the samples were frozen until extractions with 2 *M* KCl could be done. The resulting extract was sent to the US Bureau of Reclamation lab in Bismarck, ND where NH_4^+–N and NO_3–N were determined colorimetrically (USEPA, 1983) and reported to the nearest 0.01 mg L^{-1}. After 1995, the soil samples were frozen until they could be air dried. The air-dried samples were then extracted and analyzed for NO_3–N to the nearest 1 mg L^{-1} by the North Dakota State Soil Testing lab using an ion specific electrode.

Water Quality Monitoring

The research field was intensively instrumented for water quality monitoring of the vadose zone, saturated zone, and subsurface drainage. Figure 1 shows the locations of the instruments used to monitor NO_3–N levels. The monitoring instrumentation included disturbed- and undisturbed-profile lysimeters; two transects of nested shallow, intermediate, and deep wells; and two subsurface drains. The water quality samples from each instrument were collected and 50 mL was subsampled. The subsamples were preserved with 0.1 mL of sulfuric acid immediately and then frozen until analysis. The samples were analyzed for NO_3–N colorimetrically using automated cadmium reduction (USEPA, 1983).

Derby et al. (1997) provided a detailed description of the construction of the disturbed and undisturbed-profile lysimeters. Four box-shaped, disturbed-profile lysimeters sampled an area of 1.86 m^2, and the bottom of the lysimeters were enclosed to a depth of 1.8 m. Sixteen cylindrical, undisturbed-profile lysimeters had a diameter of 0.6 m, and the bottoms of these lysimeters were located at a depth of 1.8 m. The major difference between the two lysimeter types was soil structure. During construction of the disturbed-profile lysimeters, the soil structure was destroyed when the soil was repacked into the lysimeter tanks. In contrast, during the construction of the undisturbed-profile lysimeters (Derby et al., 2002), the structure of the soil was carefully maintained. Both disturbed- and undisturbed-profile lysimeters were buried to a depth of 0.36 m to accommodate tillage. Free drainage from the disturbed-profile lysimeters and a combination of vacuum extractor and free drainage from the undisturbed-profile lysimeters were usually collected weekly starting in April 1990. Generally, samples were not collected during winter months.

Shallow groundwater was monitored using nested wells installed at shallow (screened between 2.4 and 3.0 m), intermediate (screened between 3.0 and 3.6 m) and deep (screened between 5.4 and 6.0 m) depths. Two transects of nine nested-well sites were installed within the field (Fig.1), for a total of 54 wells (18 shallow, 18 intermediate, and 18 deep). The well casing and screens were constructed of schedule 40 polyvinyl chloride that was 5 cm in diameter. The nest of wells provided samples from the top 0.3 m of the surficial (or unconfined) aquifer, the next 0.3- to 0.9-m interval, and at the top of the confining layer of the saturated thickness, which was a compacted till and ranged in depth between 5.5 to 6.1 m. Water quality samples have been collected from the wells monthly since January 1989 and periodically during winter months.

Drainage from two subsurface drains, located at an average depth of 2.5 m, was sampled monthly since January 1989 except during winter. Drainage was sampled from four manhole access points (Fig. 1). The subsurface drains are headwater drains; therefore, water quality samples collected from the drains were not influenced by upstream effects from nearby fields.

Data Analysis

For each sample date, NO_3–N concentrations were measured from 18 samples taken from each of the following: undisturbed profile lysimeters, shallow wells, intermediate wells, or deep wells. Also, there were two samples taken from the subsurface drainage lines, and four samples taken from the disturbed lysimeters. To simplify the presentation of this sample data, the NO_3–N concentrations were averaged from each sampling date from each instrument type. Additionally, confidence intervals (CI) were determined for the NO_3–N concentration at each sample date from the 18 shallow and 18 intermediate wells. This was done using a student t test and assuming a normal distribution of the eighteen NO_3–N concentration values (Steel and Torrie, 1980).

A function was fit to the NO_3–N time series data to help interpret trends through time. The NO_3–N concentrations increased rapidly and reached high levels following the initiation of irrigation. After the initial increase, the NO_3–N concentrations decreased and reached steady-state levels that were near concentrations prior to irrigation. The following dampened sine function was chosen because it provided a good description of the trends in the measured data for all of the sampling instruments:

$$C(t) = a \exp(-t/c) \sin\left[\frac{\pi(t - t_0)}{b}\right] + d \qquad [1]$$

where C is the concentration of NO_3–N (M L^{-3}); t is time (T); t_0 is time offset (T); and a (M L^{-3}), b (T), c (T), and d (M L^{-3}) are constants that controlled the amplitude, periodicity, damping, and intercept, respectively. There was rough physical significance for the parameters in Eq. [1] to the time series data. Approximately, parameter a corresponded to the magnitude of the peak NO_3–N concentration, b corresponded to the length of time it took for the initial flush of NO_3–N to move through the soil, $1/c$ corresponded to the rate at which the NO_3–N concentrations dropped from the peak concentration, and d corresponded to steady-state NO_3–N concentrations. Also, t_0 was the time that the irrigation was initiated.

The first derivative of Eq. [1], with respect to time, was used to identify local maxima of NO_3–N concentrations where $f'[C(t)] = 0$. Also, the second derivative of Eq. [1] was used to identify inflections in the time series data where NO_3–N concentrations began to reach a steady value, where $f''[C(t)] = 0$. The maxima and inflection points in the time series data were used to separate three distinct phases in the NO_3–N data, where the NO_3–N concentrations (i) increased, (ii) decreased, and then (iii) reached and maintained steady state.

Equation [1] was fit to the observed NO_3–N concentration data by minimizing an objective function, the sum of squared residuals. The optimization routine was a simplification of the nonlinear least-squares curve-fitting program of Meeter (1966). A detailed description of the method is given by Press et al. (1992). Minimization of the objective function was performed iteratively by adjusting the parameter estimates for a, b, c, and d. Constraints were placed on the parameters so that uniqueness of parameter estimates could be attained. Combinations of parameter estimates where restricted by using the following condition:

$$C(t) = 0, \qquad [2]$$

This condition was most limiting in the first cycle of the time series data. In Eq. [1], sin $= -1$ when $t = 1.5b$ in the first cycle of data. Substituting Eq. [2] and $t = 1.5b$ into Eq. [1] yields the following restriction on the parameter estimates:

$$1.5b/c \geq \ln(a/b) \qquad [3]$$

Furthermore, initial parameters values were estimated visually from the time series graphs and all parameters were constrained to be positive.

RESULTS AND DISCUSSION

Agronomic Comparison of Best Management Practices Site with the Oakes Irrigation Test Area

Figure 2 shows the comparison between the BMP site and the mean of all irrigated fields in the OITA with the same crop. This was a comparison of the temporal

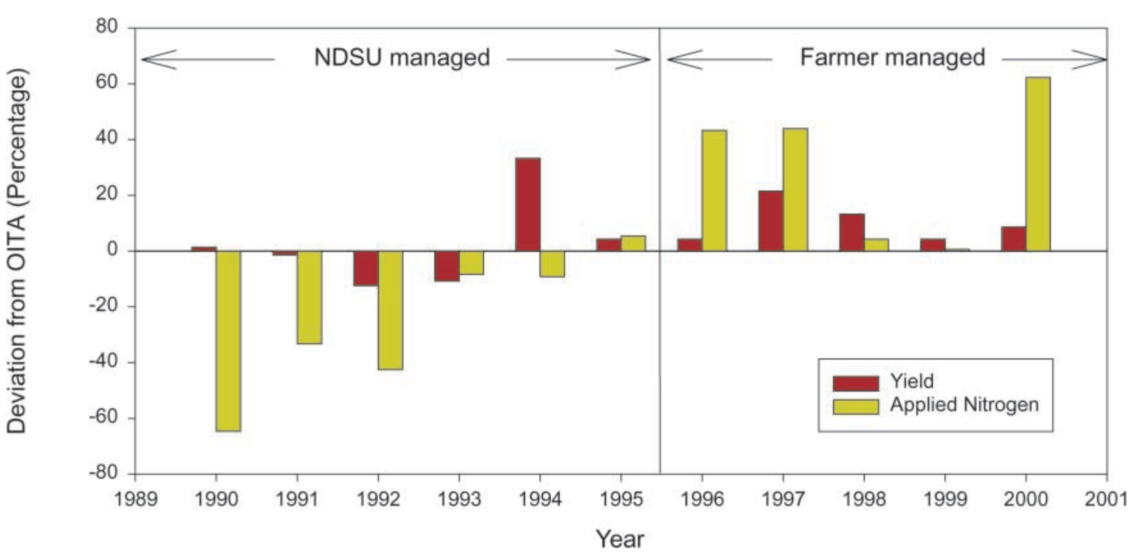

Fig. 2. A comparison between the Best Management Practices research field and the average of the Oakes Irrigation Test Area for yield and N fertilizer application rates.

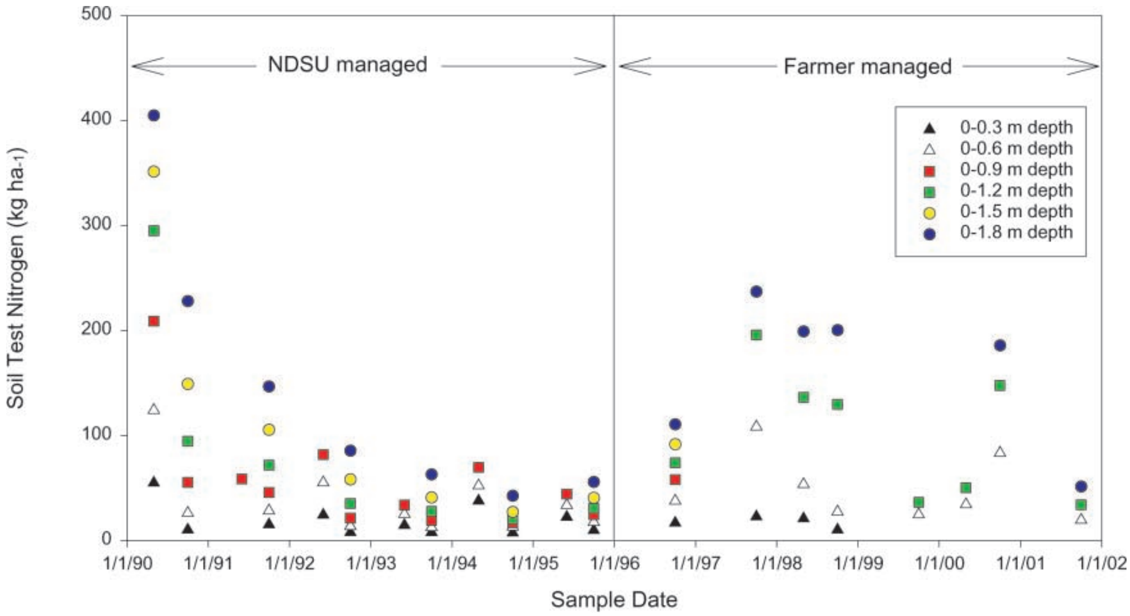

Fig. 3. The average N test values in the soil profile to several depths through time.

average of yield at the BMP site with the temporal and spatial average over all OITA fields for the same period. From 1990 to 1995, where soil tests were used to determine the split application rates of N fertilizer, the temporal average yield of the BMP site was approximately 2.4% greater than the average of the OITA, primarily due to the large positive difference in 1994. Additionally, during the 1990 to 1995 time period, there was 25.5% savings in applied N fertilizers compared with the average of the OITA. After 1995, when the management was returned to the farmer, the temporal average yield of the BMP site was 13.0% above the average OITA yield; however, the N fertilizer inputs exceeded the average of the OITA by 30.5%. The implementation of the best management practices demonstrated that reasonable yields can be achieved and application of N can be minimized.

The soil N was well managed from 1990 to 1995. This was indicated by a continuous decrease in the soil test values (0–1.8 m depth) from approximately 400 to 50 kg N ha^{-1} over the time period of 1990 to 1995, respectively (Fig. 3). After 1995, either soil tests were not used to determine the amount of N fertilizer to be applied or unrealistic yield goals were used, and as a result, the measured soil N values reached peak values of 250 kg ha^{-1} for a depth of 0 to 1.8 m.

Root Zone

Time series data of the mean NO_3–N concentrations from the undisturbed- and disturbed-profile lysimeters increased just after initiation of irrigation (Fig. 4 and 5). The antecedent NO_3–N levels in the upper root zone were near 10 to 20 mg L^{-1}, but after irrigation began the concentrations increased to 100 and 160 mg L^{-1}, for the disturbed- and undisturbed-profile lysimeters,

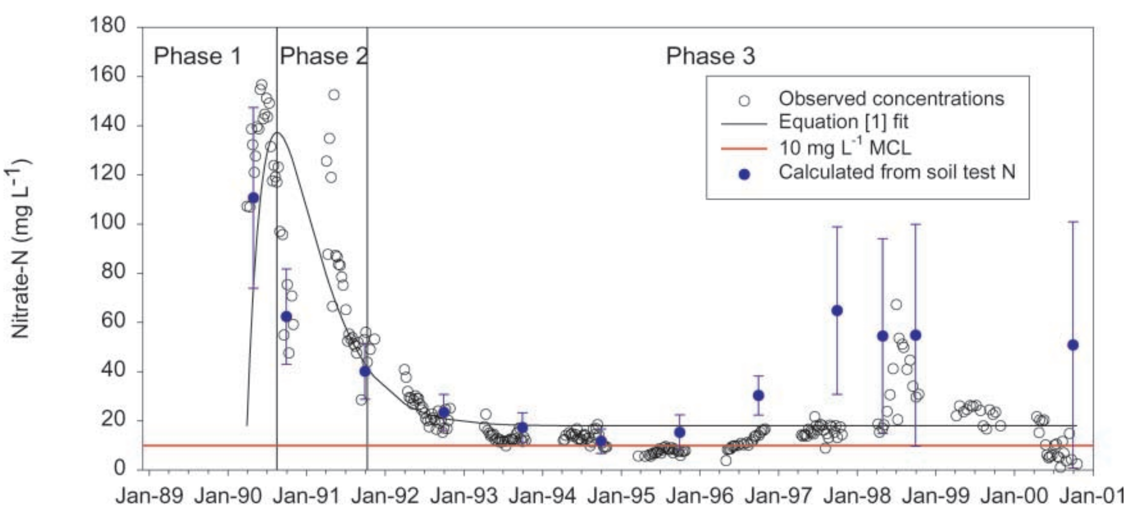

Fig. 4. Time series data of the average NO_3 concentrations observed from the undisturbed-profile lysimeters fitted with Eq. [1]. Nitrate concentrations increase in Phase 1, decrease in Phase 2, and reach and maintain a steady value in Phase 3. Also included are concentrations calculated from the soil N test values from 0 to 1.8 m along with one standard deviation error bars.

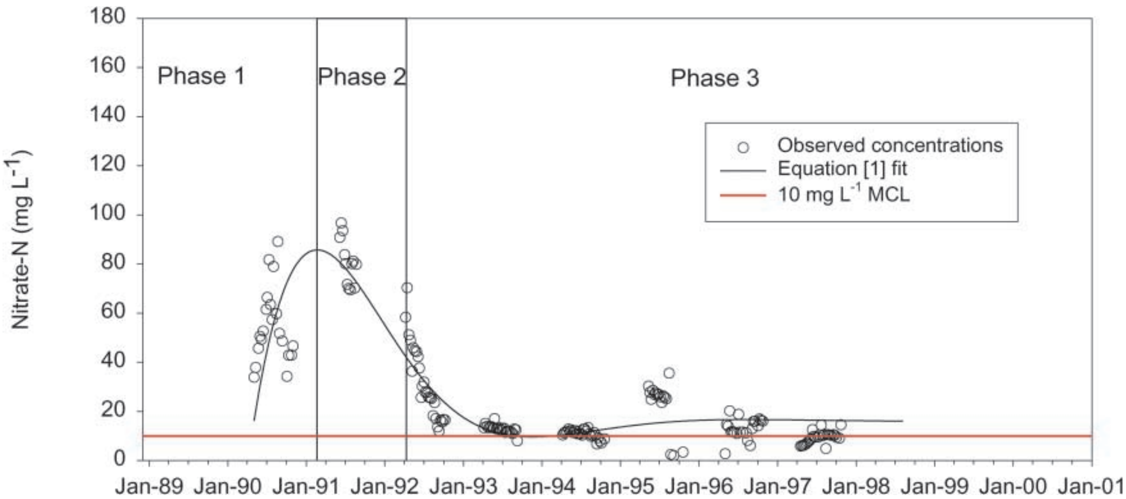

Fig. 5. Time series data of the average NO_3 concentrations observed from the disturbed-profile lysimeters fit with Eq. [1]. Nitrate concentrations increase in Phase 1, decrease in Phase 2, and reach and maintain a steady value in Phase 3.

respectively. This increase in NO_3–N concentrations can be partially explained by the high NO_3–N concentrations in the soil profile from dryland agriculture. When irrigation was initiated, it was likely that residual N in the profile was advectively transported downward when water application increased. Figure 3 shows that the N test values for the soil profiles follow a similar trend as the NO_3–N time series data from the lysimeters. When these soil N values from Fig. 3 were roughly converted to concentrations based on the increase in the water content of the soil profile, the values follow a very similar trend as the undisturbed-profile lysimeter data (Fig. 4). To convert the soil test N mass values (Fig. 3) to a concentration (Fig. 4), they were divided by the estimated volume of water present in a 1.8 m deep by 1 ha volume of soil. The similar trends in soil N test values and lysimeter data suggest that residual soil N will have a considerable effect on the amount of NO_3–N that is leached when irrigation is initiated. Furthermore, it also suggests that water was the yield-limiting factor under dryland agriculture. This result may lead to management practices that reduce N leaching by testing the soil N and/or controlling irrigation amounts.

Increased mineralization rates may have also contributed to the elevated NO_3–N concentrations in the subsurface water. Albus and Knighton (1998) and Beck (1983) observed similar increases in NO_3–N concentration in groundwater when dryland agriculture was converted to irrigation. Both Albus and Knighton (1998) and Beck (1983) attributed this increase in NO_3–N concentrations to higher mineralization rates. Other studies have demonstrated that increased soil water will increase mineralization rates, exposing more NO_3–N to leaching (Doran, 1980; Skopp et al., 1990). A more intensive tillage practice used under irrigated agriculture would also have increased the mineralization rate. Peak concentrations of NO_3–N decreased and eventually returned to levels close to antecedent concentrations. This may have been caused by a depletion in leachable N in the soil profile (Fig. 3), lower total mineralization due to a decrease in organic matter, and/or best management practices (Albus and Knighton, 1998). Unfortunately, a record of soil organic matter content was not kept to verify whether mineralization rates were increasing or decreasing.

Equation [1] described the time series data well (Fig. 4 and 5), which was indicated by the r^2 values of 0.70 and 0.79 for the undisturbed- and disturbed-profile lysimeters, respectively. Between 1990 and 1992, there were seasonal trends in the NO_3–N concentrations that were not picked up by Eq. [1]; nonetheless, the long-term trend dominated overall. The time series was separated into three distinct phases determined by the local maximum and inflection in the function (Eq. [1]). Table 2 provides a summary of the transition dates between each phase, the length of each phase, and the least-square

Table 2. The optimized model parameters that were used to fit Eq. [1] to the NO_3–N time series data. Peak date indicates where NO_3–N concentrations were highest, and the inflection date indicates where NO_3–N concentrations began to reach a steady level. The coefficient of determination indicates the goodness-of-fit of Eq. [1] to the time series data.

	Model parameter						Length of each phase			
Time series data	*a*	*b*	*c*	*d*	Peak date	Inflection date	Phase 1	Phase 2	Phase 3	r^2
	ppm	d	d	ppm			d			
Undisturbed profile lysimeters	478	866	192	19	16 Aug. 1990	10 Oct. 1991	140	419	3296	0.70
Disturbed profile lysimeters	176	1000	420	16	20 Feb. 1991	10 Apr. 1991	266	414	2092	0.79
Shallow wells	44	2568	1907	13	31 July 1991	21 May 1994	953	1025	2334	0.86
Intermediate wells	71	4034	1868	4	16 May 1992	15 Nov. 1996	1244	1693	1425	0.90
Drainage lines	6	4292	1571	3	16 Nov. 1992	11 Dec. 1997	1111	1851	1042	0.41
Albus and Knighton wells†	70	3242	782	13	27 Feb. 1993	20 Mar. 1997	636	1481	1287	0.78

† Albus and Knighton (1998).

parameter estimates of Eq. [1]. The first phase was indicated by an increase in NO_3–N concentrations, NO_3–N concentrations then decreased in the second phase, and in the third phase NO_3–N concentrations reached a steady state. In the third phase, NO_3–N concentrations decreased until 1995 for both undisturbed and disturbed profile lysimeters. After 1995, when best management practices were abandoned, the NO_3–N concentrations fluctuated in this third phase, which was probably a result of excessive N fertilization.

The NO_3–N concentration time series of the undisturbed- and disturbed-profile lysimeters were different even though the data were collected from the same depth in the soil profile. The peak NO_3–N concentrations from the undisturbed-profile lysimeter were greater, which was indicated by the higher amplitude of Eq. [1] (Table 2). Also, Phase 1 of the undisturbed-profile lysimeter was shorter than that of the disturbed-profile lysimeter (Table 2), which indicated that NO_3–N transport times were shorter. Advection in the large, inter-aggregate pore regions present in the undisturbed-profile lysimeters would have caused faster NO_3–N transport times as a result of higher saturated and near-saturated conductivities. In miscible-displacement experiments, the peak solute concentration in the column effluent of structured soils is greater than in unstructured soils (Nielsen and Biggar, 1962). The faster transport times in the structured soils would have also decreased the mean residence time within the soil profile, thereby leading to higher peak concentrations because there was less time for NO_3–N reduction and plant uptake.

Groundwater

There was a slight increase in NO_3–N in the groundwater (depth of 5.4 to 6.0 m) monitored by the deep wells (data not shown), but concentrations remained well below the 10 mg L^{-1} MCl. The deep well NO_3–N concentrations remained in the range from 0.006 to 1.59 mg L^{-1} for the duration of the study, with an average of 0.48 mg L^{-1}. The time series data of NO_3–N from the shallow and intermediate wells (between 2.4 and 3.6 m deep) indicated a significant impact on groundwater quality for long periods of time (Fig. 6 and 7). After irrigation began, there were long periods of time (indicated by the red area in Fig. 6 and 7) when there was a 50% probability that all 36 shallow and intermediate wells on the research field would contain NO_3–N levels above the 10 mg L^{-1} MCl. This occurred when the lower limit of the 50% confidence interval for NO_3–N concentrations was above 10 mg L^{-1} (approximately between June 1990 and April 1994 for the shallow wells and October 1990 to March 1995 for the intermediate wells). Although the peak NO_3–N concentrations found in the intermediate wells were not as high as the concentrations found in the shallow wells, they remained above 10 mg L^{-1} for a longer period of time.

The same three-phase trend that was observed in the lysimeter NO_3–N time series data was also observed in the shallow (Fig. 6) and intermediate (Fig. 7) wells; however, the peak NO_3–N concentrations were lower and more spread-out through time. Equation [1] also provided a good description of the well data (r^2 values of 0.86 and 0.90 for shallow and intermediate wells, respectively). The intermediate well data had lower peak concentrations and longer first and second phases than the shallow well data (Table. 2). The lower peak concentrations and longer Phase 1 and 2 durations can be partially explained by physical transport phenomena. Much of the spreading and damping of the pulse of NO_3–N could have been caused by hydrodynamic dispersion and would be proportional to the travel distance in the soil profile. The intermediate wells were deeper than the shallow wells, resulting in a longer travel distance. Thus, the NO_3–N peak would be more spread and damped because of hydrodynamic-dispersion. The longer travel distance also resulted in longer residence times within the soil profile. The longer residence time could also result in more NO_3–N reduction, which may

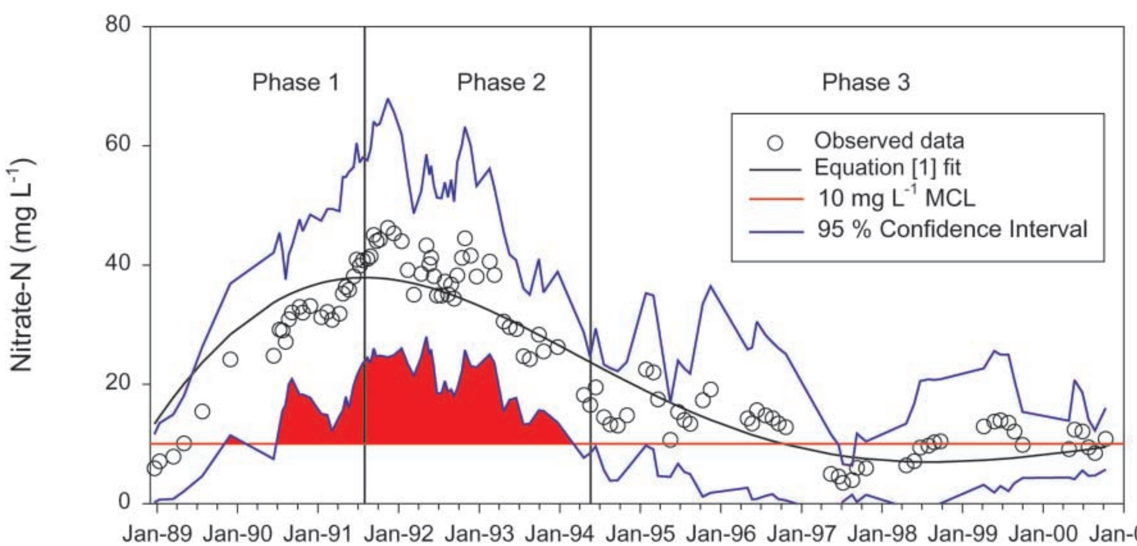

Fig. 6. Time series data of the average NO_3 concentrations observed from the shallow wells fit with Eq. [1]. Nitrate concentrations increase in Phase 1, decrease in Phase 2, and reach and maintain a steady value in Phase 3. The jagged lines are the 50% confidence limit and the red area indicates a period of time where 50% of the well samples tested above the 10 mg L^{-1} level.

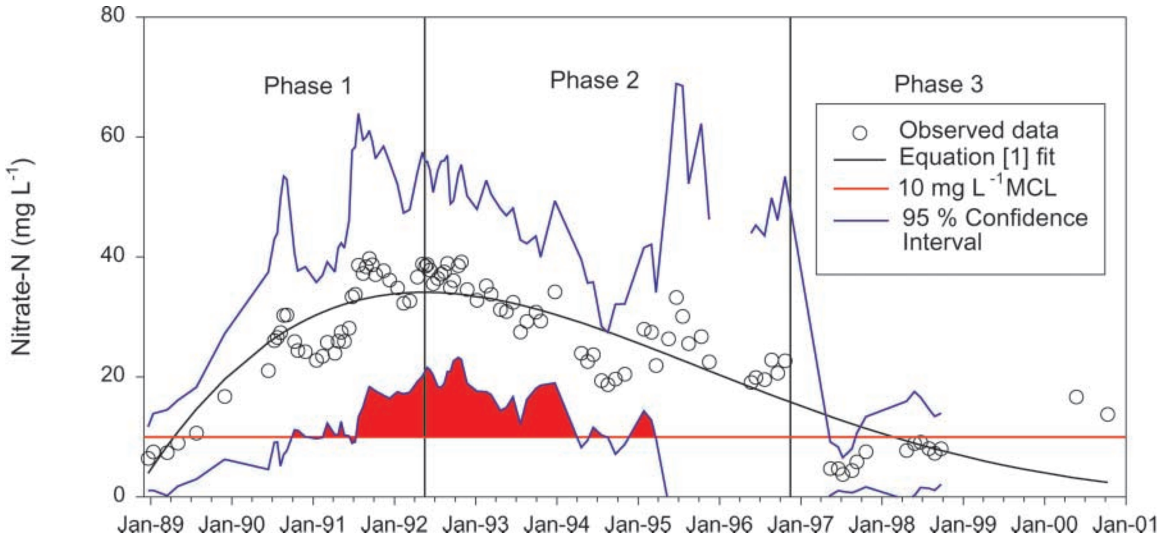

Fig. 7. Time series data of the average nitrate concentrations observed from the intermediate wells fit with Eq. [1]. Nitrate concentrations increase in Phase 1, decrease in Phase 2, and reach and maintain a steady value in Phase 3. The jagged lines are the 50% confidence limit and the red area indicates a period of time where 50% of the well samples tested above the 10 mg L^{-1} level.

explain some decrease in NO_3–N concentrations. This stratification of NO_3–N concentration in the surficial aquifer under irrigation was also reported by Olson (1992) and by Knighton (1997). It should also be noted that the irrigation water source for the BMP field was not from a groundwater well located at the BMP field, but from a pipeline (Fig. 1) supplied via canal from off site.

Similar groundwater time series data for NO_3–N were presented in an independent study by Albus and Knighton (1998), where irrigation was initiated on dryland agriculture in a field referred to as the Management Systems Evaluation Area (MSEA, Fig. 1). The MSEA site was located 1.61 km from the BMP site, and the groundwater level observations (average water table depth = 4.6 m) were similar to the shallow and intermediate well observations from the BMP site. The irrigation water was supplied via pipe from the canal system (Fig.1) near the MSEA field, and there was no subsurface drainage located at this site. When the NO_3–N time series data from MSEA were fit with Eq. [1] (Fig. 8; Table 2), the same three-phase trend was observed. The presence of the three-phase trend in NO_3–N data at two independent areas and times suggests a general trend of NO_3–N increase and decrease in groundwater following initiation of irrigation in dryland agriculture.

Nitrogen fertilizer best management practices used in Phase 1 and 2 did not decrease NO_3–N levels in the shallow groundwater in the Albus and Knighton (1998) study nor the current study. However, the best management practices did help decrease the soil NO_3–N values, which may have decreased the length of phase 1 and 2. Sound N fertilization management practices may also be important in maintaining steady-state NO_3–N concentrations during Phase 3. Nitrate concentrations in the undisturbed and disturbed profile lysimeter increased

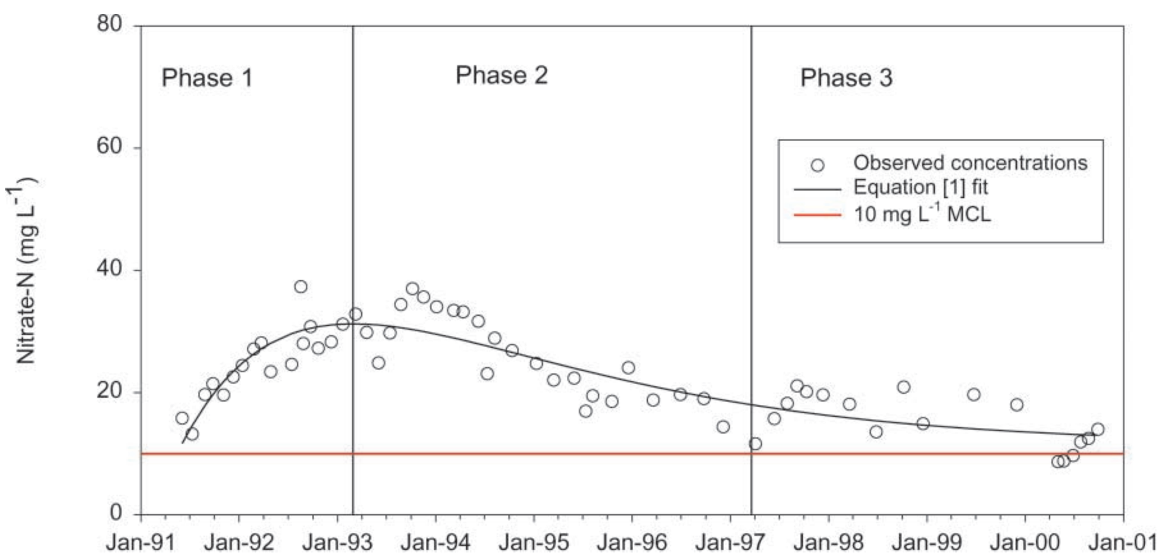

Fig. 8. Time series data of the average NO_3 concentrations observed from shallow groundwater wells from the Albus and Knighton (1998) study and fit with Eq. [1]. Nitrate concentrations increase in Phase 1, decrease in Phase 2, and reach and maintain a steady value in Phase 3.

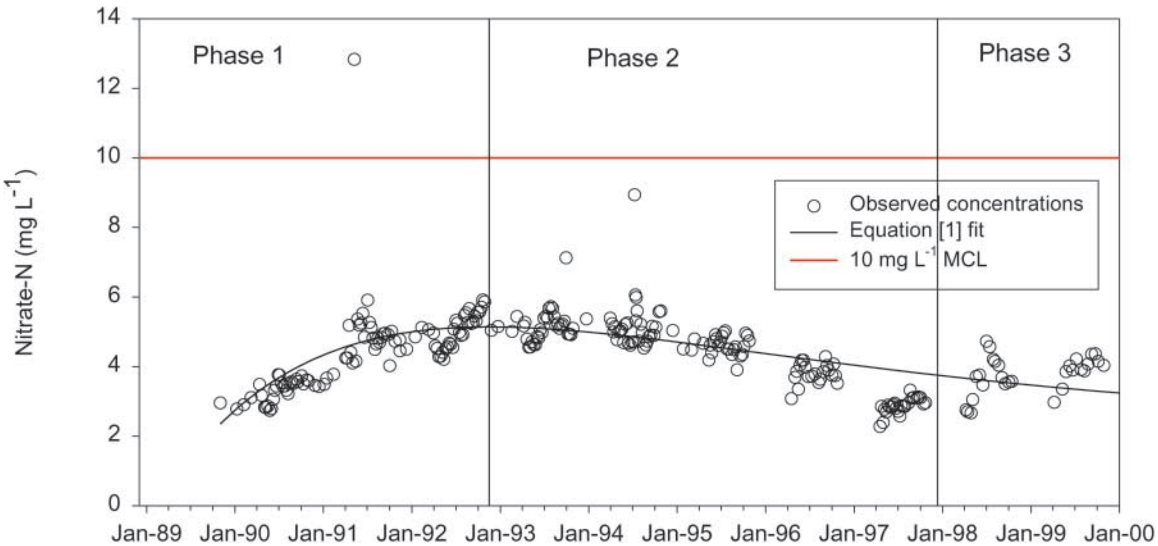

Fig. 9. Time series data of the average NO_3 concentrations observed from the tile drains fit with Eq. [1]. Nitrate concentrations increase in Phase 1, decrease in Phase 2, and reach and maintain a steady value in Phase 3.

after 1995. Also, there is approximately a 2-yr lag in the water quality response to surface management practices by the shallow wells. The return of management to the farmer in 1996 was indicated by a corresponding increase in NO_3–N concentrations observed in the shallow wells 2 yr later in 1998. These elevated NO_3–N concentrations were preceded by heavy N fertilizer applications in 1996 through 2000 (Table 1) and greater than average total water input in 1998. The departure from the use of best management practices during this time period may explain the elevated NO_3–N during Phase 3.

Subsurface Drainage

The subsurface drainage NO_3–N time series data also followed the same three-phase trend observed in the lysimeter and well data; however, the concentrations were surprisingly low (Fig. 9). The NO_3–N concentrations of the subsurface drainage ranged between 4.34 to 12.83 mg L^{-1} and averaged 2.28 mg L^{-1}. Only one data point on 8 May 1991 tested above the 10 mg L^{-1} MCL and corresponded to a heavy rainfall of 89 mm after fertilizer application. Another spike in the subsurface drainage data on 8 July 1994 also corresponded to a heavy rainfall of 126 mm. It was likely that the heavy rainfall increased drainage and preferential flow and caused higher NO_3–N levels in the subsurface drainage. Derby and Knighton (2001) showed that depressional-focused recharge can result in preferential movement of solutes in this field. Nonetheless, it is uncommon to find NO_3–N concentrations much below 10 mg L^{-1} leaving subsurface drainage on intensively managed agricultural areas. Jaynes et al. (1999) measured between 4 and 66 kg ha^{-1} of NO_3–N lost in the surface waters of a 5200-ha intensively farmed agricultural watershed. They attributed most of this loss to subsurface drains that outlet to a surface water stream that often had NO_3–N concentrations above the 10 mg L^{-1} MCL.

The depths of the subsurface drains were similar to depths of the shallow groundwater wells; however, the average relative difference in NO_3–N concentration between the shallow wells and subsurface drainage was 77% for each sampling date. The decrease in NO_3–N concentrations between the groundwater and drainage lines may have been caused by a combination of biotic and abiotic processes. Knighton (1997) reported that a biofilm containing *Brevibacter* spp. and *Arthrobacter* spp., which are both capable of NO_3 reduction, was present in the subsurface drains and in the gravel envelope surrounding the drains. Also present in the biofilm was a C source that appeared to be roots and could be used by the *Brevibacter* spp. and *Arthrobacter* spp. to reduce NO_3–N. Another source of NO_3–N reduction may have been Fe that was present in the water at high levels. Ferrous Fe in the water would have been capable of reducing NO_3–N to other forms, such as N_2O or N_2 (Buresh and Moraghan, 1976). The reduction of NO_3–N in the subsurface drains decreased concentrations to a point where the initiation of irrigation had little impact on the quality of the water leaving the field to surface water. Furthermore, even the high N fertilizer application between 1996 and 2000 had little impact on NO_3–N levels in the drains.

CONCLUSION

Changes in NO_3–N concentrations in water of the vadose zone, groundwater, and subsurface drainage were observed as a result of the initiation of irrigation. At all depths of observation, the NO_3–N concentrations increased as a result of an increase in applied water; however, the NO_3–N concentrations decreased and eventually returned to antecedent levels through time. Much of the increase in NO_3–N concentrations can be explained by residual N in the soil profile from the previous dryland agriculture. This suggests that water was the limiting factor for the availability of N during dryland agriculture. When soil water content increased, so did the mobility and availability of NO_3–N. It may be possible to manage the amount of NO_3–N that is leached due to the initiation of irrigation if either the soil N is lowered or the soil water content is controlled by properly scheduled irrigation. A practical management practice that may improve more efficient utilization of the initial levels of soil test N would be to phase in irrigation amounts more gradually.

There was a similar three-phase trend in the NO_3–N time series data for all depths of observation, where concentrations increased, then decreased, and then reached a quasi-steady-state level. A similar trend in the NO_3–N time series data was independently observed by another study where irrigation was initiated on a previously dryland agricultural field (Albus and Knighton, 1998). Management of N application at the surface had little impact on controlling the increase in NO_3–N concentrations during Phases 1 and 2; however, the best management practices reduced N fertilizer inputs and perhaps decreased the length of the first two phases. Also, the abandonment of best management practices that were in place between 1990 and 1995 appear to have increased NO_3–N concentrations in the vadose zone and shallow groundwater during Phase 3. This study illustrates the need to monitor subsurface water quality for longer periods of time to adequately assess the effects of management practices on water quality trends under irrigated agriculture.

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