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Executive Summary

Agrichemicals are widespread groundwater pollutants in many agricultural regions of the US, including the Wisconsin central sand plain. In the Agricultural Statistics District that includes much of the central sands, 22% of exploitable groundwater exceeds the federal maximum contaminant level (MCL) for NO_3-N (10 mg L^{-1}) and > 23% contains at least one herbicide (LeMasters and Baldock, 1995). Present strategies to alleviate agricultural impacts on groundwater in the central sands, primarily implementation of agricultural best-management practices (BMPs), have not substantially improved groundwater quality. New strategies are needed, but their development requires more information about the quality of groundwater that results from the prevailing agricultural systems, and about the impacts, transport, and fate of agrichemicals in groundwater basins.

This study was undertaken to assess the impacts of irrigated vegetable agriculture on groundwater in Wisconsin's central sands. The objectives were to measure groundwater quality and solute loading under an irrigated vegetable field, and investigate the transport of agrichemicals beyond vegetable fields. This study is a partial continuation of an earlier study (Kraft et al., 1995) in the Port Edwards Groundwater Priority Watershed. The current study added two more years of data (3 March 1994 to 30 March 1996) from one irrigated vegetable field and from monitoring wells in the northern portion of the Priority Watershed.

The study area consists of the northern 650 ha of the Priority Watershed (Figure 1). Land uses there include five irrigated vegetable fields covering 210 ha; other major land uses are forest and old field. The geology consists of about 20 m of Pleistocene sand with one or more interbedded silty units overlying finer-textured unconsolidated materials and bedrock. Only the Pleistocene unit is significant from a groundwater standpoint. One silty unit, the New Rome Member of the Big Flats Formation, is nearly continuous; others appear localized. The New Rome has a thickness of 0.3 to 1.5 m, and occurs 6-10 m below the surface. Groundwater flows to the southeast, with the water table 1.5 to 12 m below the surface.

Groundwater under an irrigated vegetable field

Groundwater quality under a vegetable field was determined through groundwater monitoring. Nitrate loading to groundwater was estimated from a nitrogen budget and calculated from monitoring data. Previously, Kraft et al. (1995) instrumented four irrigated vegetable fields in the north of the study area with multilevel piezometers (MLPs) capable of sampling groundwater at 15-40 cm intervals in the upper 3 to 4 m of the saturated zone. Six MLPs were installed in each field, and six upgradient of the fields. The current study continued monitoring of one field (Field 2). The MLPs were sampled 23 times in the earlier study (January 1992-April 1994) and 16 times during the current study (June 1994-March 1996). The current study monitored NO_3, Cl, pH, and specific conductance on every sampling event, and pesticides once.

Inorganic analyses

Analyte values varied with time, depth, and location in the field. Nitrate-N concentrations in individual samples ranged from < 0.2 to 50.5 mg L^{-1}, and Cl from < 1 to 119 mg L^{-1}. The pH range was 3.72 to 7.27, and specific conductance ranged from 3 to 75 mS m^{-1}. Solute concentrations were much greater under the field than in upgradient MLPs.

Upgradient of fields, the mean analyte values in the monitored zone (approximately the uppermost 3 m of groundwater) during the period of record were NO_3-N, 0.7 mg L^{-1}; Cl, 1.4 mg L^{-1}; and specific conductance, 6 mS m^{-1}. Beneath the field, means were NO_3-N, 20.5 mg L^{-1}; Cl, 20.2 mg L^{-1}; and specific conductance, 33.2 mS m^{-1}. Virtually all of the solutes in groundwater below the field originated from farming practices.

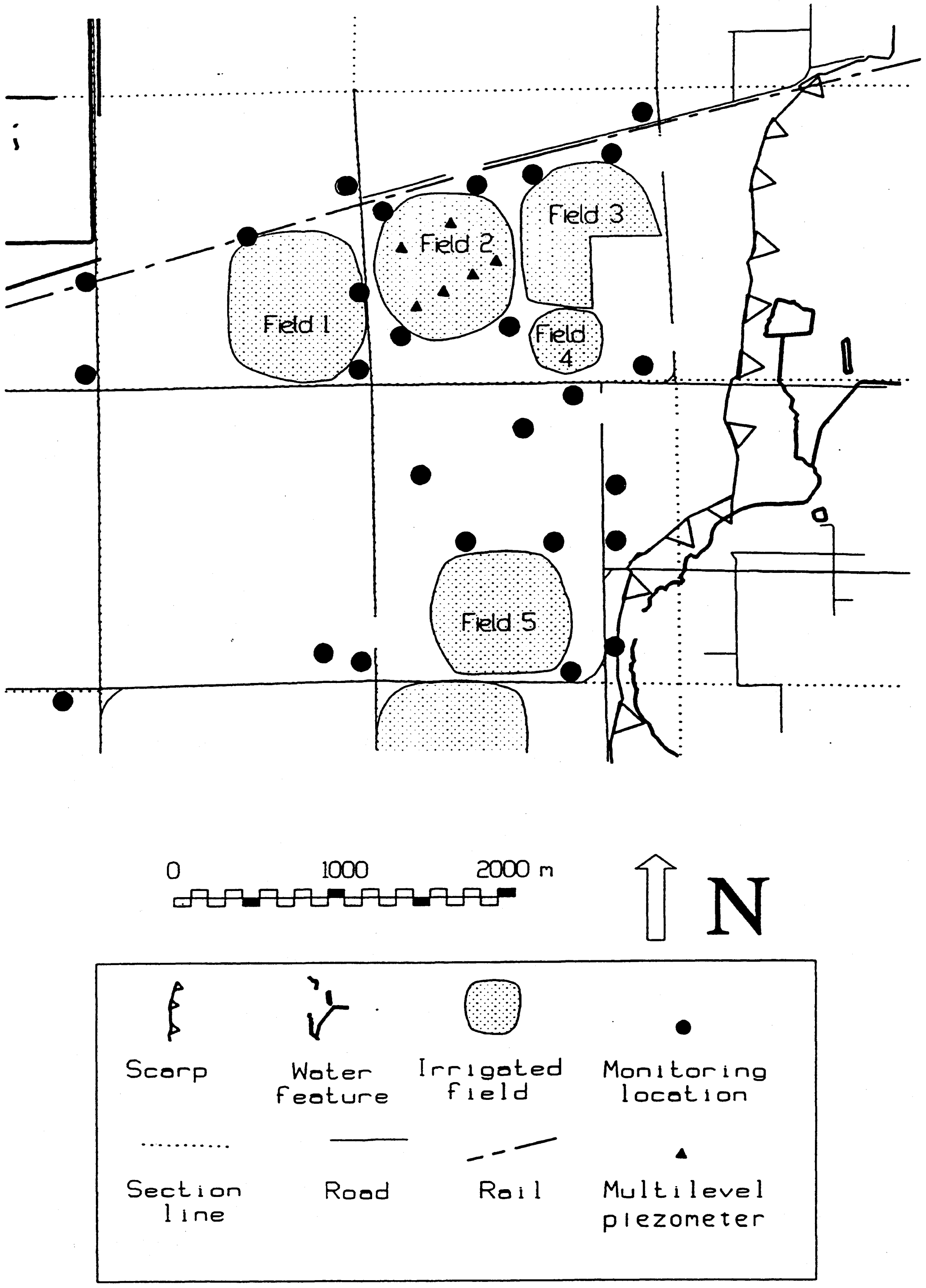


Figure 1. Main features of the study area.

Table 1. Inorganic parameter concentrations in upgradient vs in-field MLPs. Values are averages in the monitored zone for the period of record.

Parameter	Upgradient	In-field
NO_3-N (mg L^{-1})	0.7	20.5
Cl (mg L^{-1})	1.4	20.2
Spec. conductance (mS m^{-1})	5.7	33.2
pH	5.07	4.90

Whole-field average NO_3-N and Cl concentrations varied with time from their long-term averages of 20.5 and 20.2 mg L^{-1}. Averages ranged from 16.2 to 28.1 mg L^{-1} NO_3-N, and 10.3 to 36.4 mg L^{-1} Cl during the period of record.

Sequential concentration-vs-depth profiles exhibited an annual pattern of pulses, where elevated solute concentration appeared at the water table and then moved downward over time. Chloride pulses marked the top and bottom of a zone occupied by a particular year's recharge.

Pesticide analyses

In the current study, 19 ports in four MLPs were sampled for pesticide residues. Residues were detected in all but one sampled port. Residues detected were atrazine, desethylatrazine, desisopropylatrazine, carbofuran, metolachlor, and metribuzin. Other pesticide residues such as alachlor ESA may have been present, but were not detectable by our analytical procedures.

Atrazine and its breakdown products were found in 17 of 19 sampled MLP ports. Metribuzin was found in 17 ports, metolachlor in 11, and carbofuran in two. The mean concentrations of detections were 0.6 µg L^{-1} total triazines, 0.2 µg L^{-1} metolachlor, 0.9 µg L^{-1} metribuzin, and 0.4 µg L^{-1} carbofuran. The Wisconsin preventive action limit (PAL) for atrazine residues was exceeded in 10 ports. No other PAL exceedances were found. Pesticide concentrations were generally rather low; summed pesticide residues in individual ports ranged from undetectable to 4.5 $\mu g\ L^{-1}$ and the mean of summed detections was 1.3 $\mu g\ L^{-1}$.

Nitrate loading to groundwater

We estimated NO_3 loading to groundwater using a nitrogen budget method and a method based on groundwater monitoring results (water-year method). Both methods were also used by Kraft et al. (1995), giving a record of NO_3 loading for 1992-1995.

Nitrogen budget method

The N budget method estimated NO_3-N loading as the difference between N inputs and outputs. The inputs used were fertilizer, atmospheric, and soil-mineralized N, and the outputs were harvested crops and gaseous N losses. Sweet corn crops loaded 129 kg ha^{-1} of NO_3-N to groundwater in 1995 and 154 kg ha^{-1} in 1994. Kraft et al. (1995) found loading rates of 231 and 291 kg ha^{-1} (on field halves receiving different N fertilizer rates) for potato in 1993, and 182 kg ha^{-1} for sweet corn in 1992. An average of 76% of applied fertilizer N or 63% of total N inputs leached to groundwater.

Nitrate loading increased with N fertilizer inputs. The smallest NO_3-N loading was in 1995, when the N fertilizer application was slightly less than the BMP recommendation. However, the loading rate was still 129 kg ha^{-1}.

Water year method

Calculating NO_3 loading from monitoring data utilized the Cl from fertilizer applied in spring as an annual tracer. The sequence of concentration-depth profiles over time at each MLP revealed a Cl pulse arriving at the water table each late spring or summer, then passing through successively greater depths in the saturated zone. The concentration minimum preceding each Cl pulse marked the boundary between water years. When the upper and lower boundaries of a water year were present in a single concentration profile, the annual loading (mass per area) of NO_3-N was calculated as

$$\text{Annual loading} = \int_{\text{lower boundary}}^{\text{upper boundary}} \text{porosity} \times \text{concentration} \; d(\text{depth}).$$

In practice, a summation over MLP ports substituted for the integral.

Comparison of Cl input to Cl loading supported the validity of the water-year method. Fertilizer was by far the largest Cl input (> 99% of the total), and Cl loading estimates were 93% of the fertilizer application in 1994 and 102% in 1995. We assume some small portion of Cl input is exported in the crop, but data to estimate the amount exported are not available.

The water-year NO_3-N loading (average among six MLPs) for sweet corn was 132 ± 36 kg ha^{-1} in 1995 and 171 ± 62 kg ha^{-1} in 1994. Kraft et al. (1995) measured a loading of 178 kg ha^{-1} for the 1992 sweet corn crop, and for the 1993 potato crop, 195 or 271 kg ha^{-1} (north and south halves of the field received different fertilizer rates; the average loading was 233 kg ha^{-1}). Nitrate loading increased with fertilizer N input. On average, 62% of the total N input or 74% of fertilizer N appeared as NO_3 in groundwater.

NO_3 loading comparison and summary

Although NO_3 loading varied considerably among MLPs in the water-year method, agreement was excellent between the four annual average NO_3 loading rates and the N-budget estimates (Table 2).

The potato crop took up more N and a larger fraction of N inputs than sweet corn did. But the NO_3 loading rate under potato was greater because the crop received more N fertilizer. The four years of NO_3-N loading data can be used to estimate the annual loading rate in a potato - sweet corn - sweet corn rotation by calculating a weighted average. This loading rate was 190 kg ha^{-1} y^{-1} using N-budget loading rates. The water-year loading rate was virtually the same. Since our potato data came from an atypical year (record rainfall which the

Table 2. Comparison of N budget and water-year estimates of NO_3-N loading (both kg ha^{-1}). The N budget was chosen as the basis for calculating discrepancies.

	1992	1993	1994	1995	Mean
Crop	Sweet corn	Potato	Sweet corn	Sweet corn	
N budget	182	261	154	129	182
Water year	178	233	171	132	178
Discrepancy	-2%	-11%	11%	2%	-2%[1]

[1] Cumulative four-year discrepancy

grower responded to with extra N fertilizer), this estimate may be high. A possibly more typical potato N budget using the UW-Extension BMP fertilizer recommendation implies a three-year average annual loading to groundwater of 173 kg ha^{-1}.

Groundwater quality outside fields

To determine impacts of irrigated vegetable agriculture on downgradient groundwater quality, we monitored a network of 77 wells at 21 locations in the study area. Thirty-five wells at 13 locations were installed previously (Kraft et al., 1995); 42 additional wells and eight new locations were established for the current study. The new monitoring wells were positioned to intercept groundwater advecting from fields. Wells were sampled three times during the current study and up to four times during the previous study. All samples were analyzed for NO_3-N, Cl, pH, and specific conductance (a relative few samples were not analyzed for pH or specific conductance because of equipment failure). A subset of the monitoring wells was sampled for pesticide residues once during the current study and four times in the earlier study. Groundwater flow and particle-tracking models were used to simulate advective transport from fields, define flow paths, and outline plumes of groundwater affected by agricultural contaminants.

Analytical results

In the current study, 191 samples for inorganic analysis were taken from 72 wells. Nitrate-N concentrations in individual samples ranged from <0.2 to 30.9 mg L^{-1}, and <1 to 84 mg L^{-1} for Cl. The pH range in individual samples was 4.62 to 7.73, and specific conductance was 2 to 67 mS m^{-1}. The record including the earlier study comprises 277 samples from 77 wells. The averages among monitoring wells for the period of record were 8.4 mg L^{-1} NO_3-N, 20 mg L^{-1} Cl, pH 6.27, and specific conductance 23 mS m^{-1}.

In the current study, 30 wells at 12 selected locations were sampled for pesticide analysis in March 1996. Wells were selected for sampling based on their greater vulnerability to pesticide pollution. Nineteen of the wells at 10 locations contained at least one detectable pesticide residue. Atrazine, desethylatrazine, metolachlor, and metribuzin were detected.

Combining these results with those of Kraft et al (1995) produces a record of 38 wells at 12 locations, sampled for pesticides between August 1992 and March 1996. Twenty of these well contained detectable pesticide residues. In addition to the compounds listed above, carbofuran was detected once. Pesticide residues that were not detectable by our analytical methods may have been present; notably alachlor ESA. Concentrations tended to be low; 14 of the 20 wells with detections never exceeded 1 µg L^{-1} of summed pesticide residues. Of the six wells with summed residues > 1 µg L^{-1}, five were less than 250 m downgradient of a vegetable field. The maximum sum of detectable pesticide residues was 22 µg L^{-1}. This was mostly metolachlor: 20.6 µg L^{-1}, a concentration that exceeded the Wisconsin enforcement standard. The metolachlor may have been associated with a strong metolachlor pulse identified in the nearby upgradient field, possibly from some unusual event such as a spill. Additional study would be needed to determine how frequent and how serious such concentration spikes might be.

Effect of position in the flow system

Groundwater upgradient of irrigated vegetable fields contained little NO_3-N and Cl, averaging 0.5 and 2.1 mg L^{-1}. Immediately downgradient of fields in the north part of the study area, nearly all the wells had 10-40 fold greater NO_3 and Cl concentrations and exceeded 10 mg L^{-1} NO_3-N. Pesticides were detected with summed residues from 0.3 to 22.0 μg L^{-1} immediately downgradient from fields. At three of four such locations, all wells sampled contained pesticide residues; at the fourth location, no residues were detected in the three wells sampled. The New Rome silt did not prevent movement of contaminated water into the deepest parts of the aquifer.

About 400 m downgradient of the northern tier of fields (Figure 1), the flowpaths had traversed areas of mainly forest and old fields. Nitrate and Cl concentrations in shallow groundwater were similar to those at locations upgradient of fields. Deeper wells frequently contained as much or more NO_3 and Cl as the wells just downgradient from the fields. Pesticide detections were sporadic, with generally low concentrations. Denitrification was strongly indicated in deep groundwater at one monitoring location.

Some flow paths crossed another field before reaching the last monitoring location near the discharge zone. Groundwater there contained elevated Cl through the entire saturated thickness, and 9-21 mg L^{-1} NO_3-N through the upper 80%. Pesticides were detected only in the uppermost well.

We estimated the portion of the aquifer that met selected concentration criteria for Cl and NO_3 in three cross sections. Concentrations typical of agriculturally impacted water occupied the greatest aquifer fraction near the downgradient margin of four vegetable fields, viz., 80% of the aquifer exceeded 15 mg L^{-1} Cl. The impacted fraction decreased farther downgradient, due to intervening non-agricultural land uses and due to localized denitrification. In the farthest-downgradient cross section, the impacted fraction increased again due to the effects of a vegetable field near the discharge zone.

Comparisons between agriculturally impacted and nonimpacted groundwater

Wells in areas underlain by plumes originating in irrigated vegetable fields had significantly ($P < 0.05$) greater solute concentrations than wells outside such areas. The respective medians were: NO_3-N, 9.0 and 0.2 mg L^{-1}; Cl, 23 and 2 mg L^{-1}; and specific conductance, 23 and 10 mS m^{-1}.

Some of the 58 wells in plume-containing areas were above or below the plumes. We identified them by their low Cl concentrations. Grouping eight such wells with the 19 plume-absent wells slightly increased the differences in NO_3-N and specific conductance, and the statistical significance also increased.

Impacts of other land uses

The other major land uses were old field and forest. Monitoring data indicated these land uses loaded very little NO_3 and Cl to groundwater. Highway salt produced groundwater Cl concentrations up to 84 mg L^{-1} adjacent to roads. A small residential / hobby farm area along a highway may have produced 10 mg L^{-1} NO_3-N and 17 mg L^{-1} Cl at a downgradient monitoring location, but these effects could also have come from a field farther upgradient.

Implications for the Wisconsin central sand plain

Shallow (upper 3 m) groundwater under the study field averaged greater than 20 mg L^{-1} NO_3-N and almost always contained pesticide residues. Nitrate-N loading rates during the period of record varied from 130 to 180 kg ha^{-1} y^{-1} for sweet corn, and were up to 270 kg ha^{-1} for potato. For the period of record, 64% of total N inputs, or 74% of fertilizer N inputs, leached to groundwater as NO_3-N. The residues of four pesticides were identified in shallow groundwater. These were present at concentrations generally below regulatory standards, with the exception of atrazine residues which frequently exceeded the Wisconsin PAL.

We expect that the NO_3 loading rates determined for sweet corn and potato are roughly typical of much of the Wisconsin central sand plain, since management practices, field conditions, and climate during the period of study were also roughly typical. An exception was the unusually wet year of 1993, but N management probably was typical for a wet year. Nitrate loading rates measured for potato in this study are congruent with rates measured at the plot scale.

Nitrate concentrations under the study field were probably lower than typical sand plain values. Because land uses upgradient of the study field had low NO_3 loading rates, upgradient groundwater is virtually NO_3-free. Nitrate from recharge in the field efficiently disperses into groundwater recharged upgradient, effectively reducing concentrations in shallow groundwater under the field. Pesticide results under the study field are difficult to generalize. Pesticide persistence is highly dependent on soil and groundwater chemistry, which varies substantially across the sand plain.

Vegetable fields had a dominant influence on downgradient groundwater quality in the study area, elevating NO_3 and Cl concentrations through much of the saturated thickness. Nitrate concentrations in agricultural plumes frequently exceeded the MCL. Pesticide detections were also common, usually at low concentrations ($< 1\ \mu g\ L^{-1}$), though a metolachlor detection at $22\ \mu g\ L^{-1}$ exceeded the Wisconsin enforcement standard. Denitrification in the aquifer was limited to a small area, and was apparently uncommon.

The downgradient impacts of vegetable fields determined in this study may differ from what is typical for much of the sand plain. We expect that NO_3 pollution would be common downgradient of fields in the sand plain, as in this study, but the patterns and severity would be dissimilar. The northern tier of fields in this study was close to the upgradient extreme of the flow system, which led to a large fraction of the saturated thickness downgradient being affected by NO_3 and pesticides. Relatively few fields in the sand plain are likely to be in this position, and so we expect that a typical plume would be thinner. The limited denitrification observed in this study indicates that denitrification is not likely to be a significant NO_3 sink, except in places associated with wetlands. Agricultural impacts will be greater through much of the sand plain because of a higher density of this agricultural land. For instance, 40% of the recharge area for the Stevens Point, Whiting, and Plover municipal wells is in vegetable agriculture, compared with 23% in the study area.

Chapter 1
Introduction

The leaching of agrichemicals, principally nitrate and pesticide residues, is a major source of groundwater pollution (Hallberg, 1986) and an issue of "great public concern" (USOTA, 1990). Nationally, 2.4% of US domestic wells exceed the maximum contaminant level (MCL) for NO_3-N (10 mg L^{-1}) and 4.2% contain at least one pesticide (USEPA, 1990). These rates are often higher in agricultural areas. In Wisconsin, NO_3 MCL exceedance rates in "exploitable groundwater" are 10% statewide, but 17-26% in predominantly agricultural districts (LeMasters and Baldock, 1995). Herbicide detection rates in exploitable groundwater are 14% statewide, and also generally higher in agricultural districts. In some districts, atrazine residues are present in 13-32% of exploitable groundwater, and alachlor residues in 11-24%.

Agrichemicals are a long-standing problem for Wisconsin Central Sand Plain ("Central Sands") groundwater. In the Agricultural Statistics District that includes much of the Central Sands, the NO_3 MCL is exceeded in 22% of domestic wells, atrazine is detectable in 23% of wells, and alachlor is detectable in 15% (LeMasters and Baldock, 1995). Other pesticides identified in Central Sands groundwater include alachlor, aldicarb, carbofuran, metribuzin, metolachlor, and ethylene dibromide (WDNR, 1993).

Present strategies to alleviate agricultural impacts on groundwater in the Central Sands rely on agricultural best-management practices (BMPs), especially reductions in agrichemical inputs. However, the reductions prescribed by BMPs have not resulted in acceptable groundwater quality. If current BMPs do not adequately protect long-term groundwater quality while allowing profitable farming, alternative strategies must be developed. New strategies will require more information about the quality of groundwater that results from the prevailing agricultural systems, and about the transport and fate of agrichemicals in groundwater basins.

Goal and Objectives

The goal of this study was to assess the impacts of irrigated vegetable agriculture on Central Sands groundwater.

The objectives were to:

(1) measure the loading of agrichemicals to groundwater under an irrigated vegetable field during a two-year period,

(2) investigate the transport of agrichemicals downgradient of fields, using groundwater monitoring and computer modeling.

This work continues efforts in the study area (Kraft et al., 1995) to measure groundwater quality and NO_3 loading under irrigated vegetable fields and to determine groundwater impacts downgradient from fields. Kraft et al. studied groundwater quality under four fields, and groundwater conditions under the entire former Port Edwards Groundwater Priority Watershed. They installed an extensive groundwater monitoring network, investigated geologic and hydrogeologic conditions, and determined organic and inorganic chemical characteristics of the groundwater. The present study covered a subarea of the Port Edwards Groundwater Priority Watershed. We continued monitoring multilevel piezometers (MLPs) in one of the previous study fields and downgradient wells. Important tasks for this study included installing an additional 42 wells at five existing and eight new locations, updating the previous geology investigation with new information, monitoring groundwater quality under fields about monthly (less in winter), monitoring groundwater quality along flow paths downgradient of four study fields, and creating groundwater-flow and particle-track models to help understand transport processes.

Study area

The study area (Figures 1-1, 1-2) comprised the four northern sections of the former Port Edwards Groundwater Priority Watershed, and some of the surroundings. The four sections cover about 925 ha (2280 acres). The study area includes five irrigated fields with a combined area of 210 ha, or 23% of the total. Most of the remainder is forest, grassland, and brush. Residential land occupies about 2% of the area.

The study area is generally flat except at a terrace escarpment along the eastern edge (Figure 1-2). The principal soils are excessively-drained Plainfield sand and loamy sand, moderately well-drained Friendship loamy sand, somewhat poorly-drained Meehan loamy sand, and poorly-drained Newson loamy sand (Bartelme, 1977). Plainfield soil dominates; the wetter three soils were found mostly in the western part of Sections 6 and 7.

Geology consists of Pleistocene deposits overlying Hillslope sediment, Cambrian sandstone, or Precambrian crystalline rock. Groundwater flow is primarily in the Pleistocene units. More details of the geology and hydrogeology are given in Chapter 2.

A general discussion on area climate is provided by Bartelme (1977). Monthly precipitation from 1988 through April 1996 is shown in Table 1-1. Figure 1-3 shows precipitation from February 1992 through April 1996. The cumulative precipitation for 1994 and 1995 was compared with records from Wisconsin Rapids. Daily differences were sometimes substantial, but long-term agreement was excellent. The long-term annual average precipitation in Wisconsin Rapids is 795 mm; 1994 and 1995 were about average in total precipitation with 798 mm and 810 mm respectively; August 1995 was wet.

Lengths of growing seasons for 1988-1995 are given in Table 1-2. The growing seasons (last freeze in spring to first freeze in fall) were 116 days in 1994 and 125 days in 1995. Growing-season heat units above 18° C (65°F) were 217 Celsius degree-days. To summarize the years of the study, precipitation was close to average and the growing

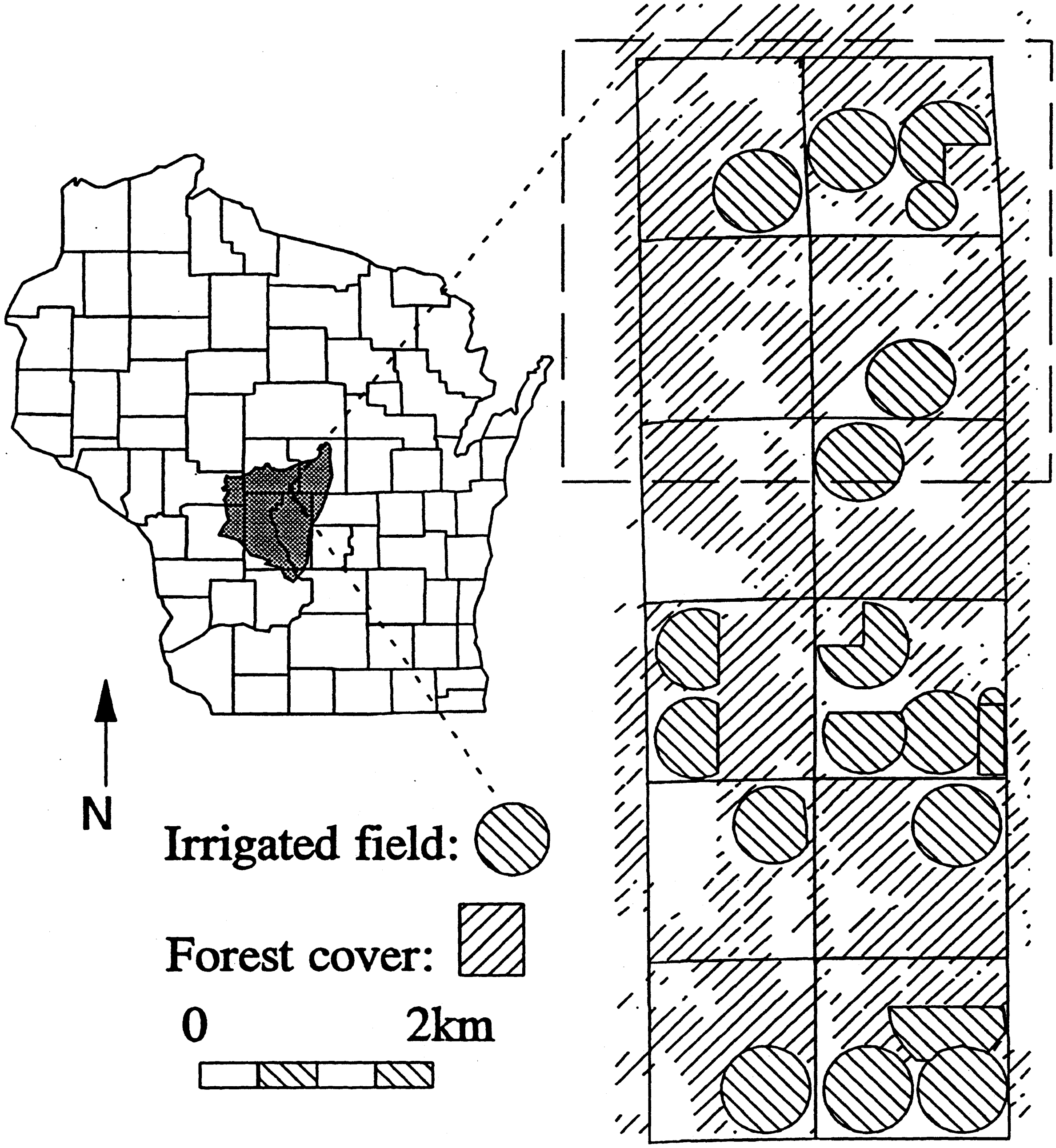


Figure 1-1. Location map of the former Port Edwards Groundwater Priority Watershed.

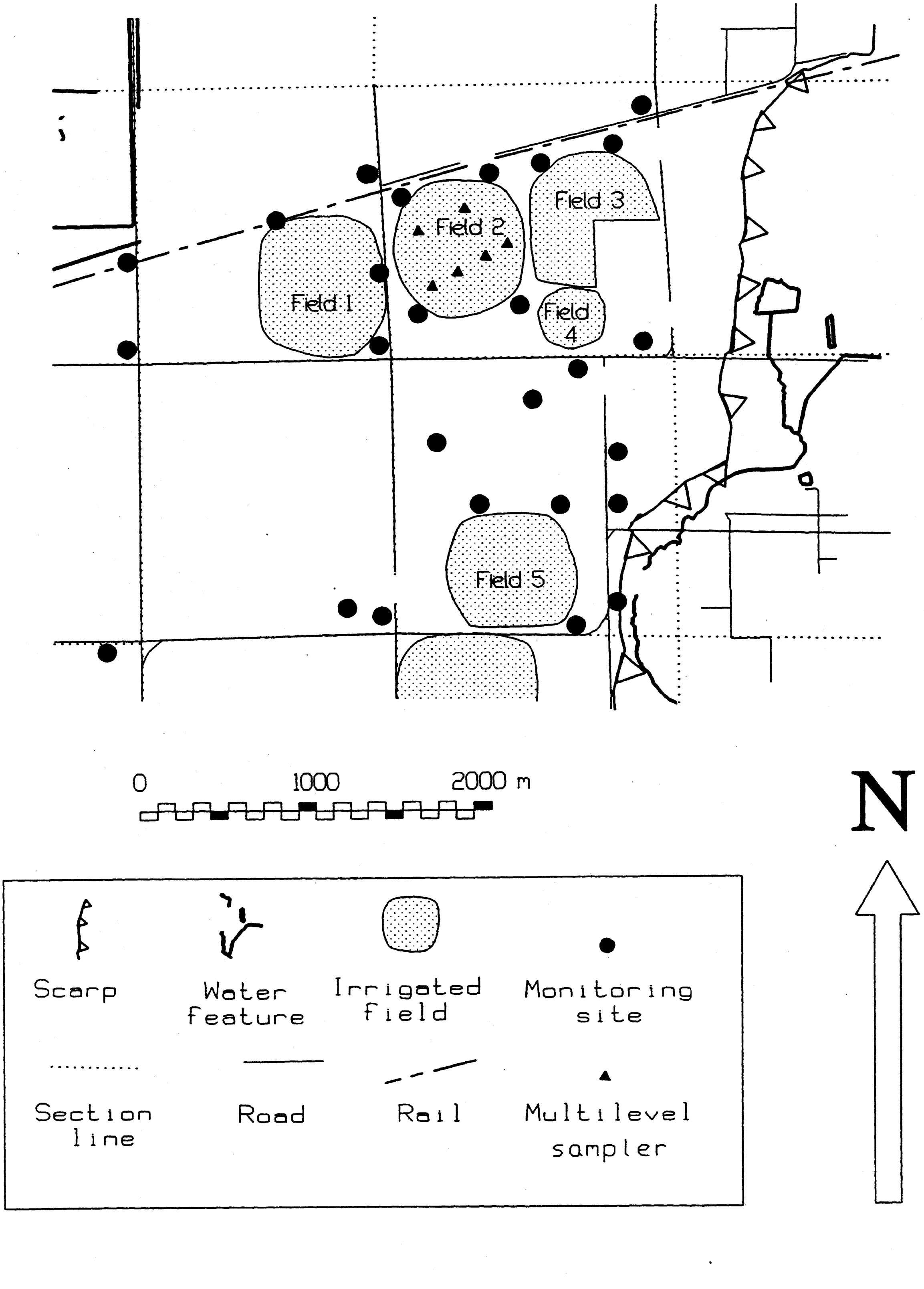


Figure 1-2. Major features of the study area.

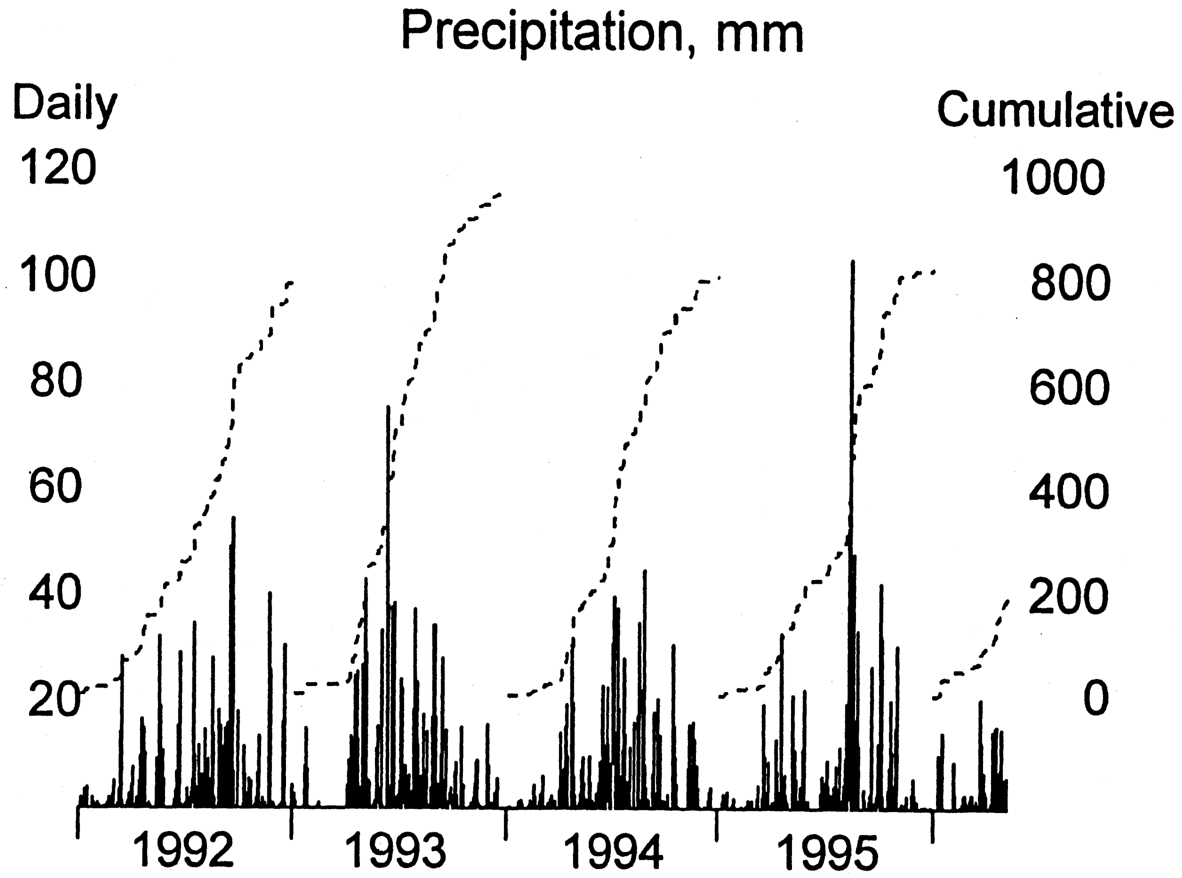


Figure 1-3. Daily precipitation (bars) and cumulative annual precipitation (dashed curves). 1992 data are from the Hancock, Wisconsin automated weather station, and later years are from the Nekoosa station.

seasons were slightly shorter than average. The 1994 growing season was cool, and the 1995 growing season was about average. Of the years immediately preceding this study, 1992 had a short, cool growing season and 1993 was very wet, especially April through June.

Table 1-1. Monthly total precipitation (mm), 1988–1995 at Wisconsin Rapids, the closest reporting station to the study area. USEDS, 1988-93; P. Knox (Wis. State Climatologist), written commun. (1996).

	1988	**1989**	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**
January	39	11	14	24	16	32	31	18	61
February	8	15	19	10	12	6	16	9	30
March	44	59	55	74	63	32	17	64	40
April	59	25	111	50	70	142	104	59	78
May	15	224	140	89	60	150	30	85	
June	29	26	30	142	33	197	60	43	
July	58	67	97	171	97	142	187	60	
August	102	75	47	175	65	116	90	271	
September	133	31	76	73	206	83	93	59	
October	37	103	63	51	29	40	44	125	
November	66	35	112	16	99	25	51	43	
December	20	8	35	44	54	26	8	20	
TOTAL	611	679	798	919	803	990	733	856	

Hydrology

The study area receives about 77 cm of precipitation in an average year, interpolating between Stevens Point and Hancock averages (Weeks and Stangland, 1971). Forests, grasslands, and non-irrigated cropland in the Central Sands evapotranspire about 53 cm, and 3 cm or less runs off, leaving about 21-23 cm for groundwater recharge (Weeks et al., 1965). Kraft et al. (1995) estimated 46 cm of ET in an irrigated field in 1992, and 50 cm in 1993.

Table 1.2. Growing season length (days above freezing), 1988–1995. USEDS, 1988-93; automated weather station records (1994-5).

	1988	**1989**	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**
Dates	5/25- 10/3	5/12- 9/23	5/11- 10/2	5/4- 9/19	5/27- 9/1	5/19- 9/24	6/9- 10/4	5/7- 9/23
Length	131	134	144	138	97	128	117	138
Deviation from average of 133 days	−2	1	11	5	−36	−5	−17	−8
Celsius degree-days (18°C base)	469	201	191	321	106	198	217	373

Chapter 2
Geology and Hydrogeology

The geology and hydrogeology of the study area were previously investigated by Kraft et al. (1995). This chapter updates that investigation with new information collected in the current study.

The geology of the Port Edwards Groundwater Priority Watershed consists predominantly of Pleistocene nearshore and offshore deposits of the Big Flats Formation overlying Hillslope sediment, Cambrian sandstone, or Precambrian crystalline rock (Clayton, 1991). Kraft et al. (1995) found that the Pleistocene deposits were typically about 70 ft thick (21 m) and composed of moderately well-sorted to well-sorted medium sand with one or more interbedded subunits of silt and clay. An extensive silty layer called the New Rome member of the Big Flats Formation (Attig et al., 1988) was found about 20-32 ft (6-10 m) below the surface in the previous study. The Pleistocene unit is the only significant aquifer.

During the present study, we drilled and logged boreholes at eight new and five preexisting locations. Two boreholes were installed specifically to determine depths to the aquifer base, the remainder for installing monitoring wells (Figure 2-1, Appendix 2-1). Borehole logs included descriptions of cuttings, drilling vibration and resistance, and materials retrieved on auger flights when drill strings were being withdrawn.

New boring-log information confirmed the conclusions of the earlier study and also provided new information. At previously established borehole locations, the drillers often found they could continue drilling 30-60 ft (10-20 m) beyond the earlier-reported contact with what was believed to be Hillslope deposits. These deeper materials seemed to consist mainly of alternating bands of compliant and resistant materials, which (with the help of cuttings retrieved on the augers) were interpreted as alternating sand and silt or clay. Silty and clayey bands were dominant. The bands appeared generally thin (2-3 ft; ~1 m), but a few may have been thicker, up to 10 ft (3 m). This deep banded material could be Hillslope deposits; in any

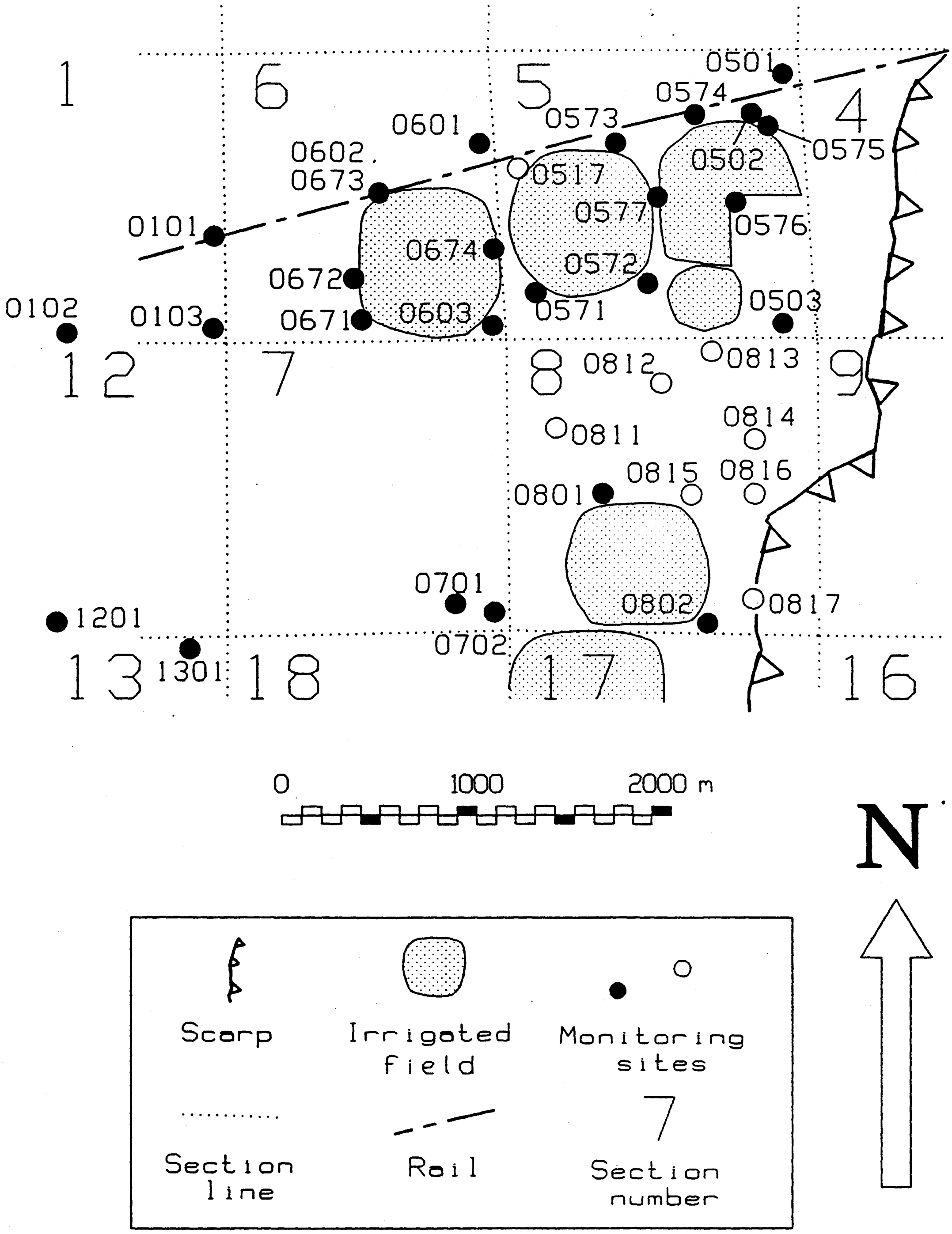


Figure 2-1. Monitoring locations used in the current study and their ID numbers. Open circles indicate locations where wells were newly installed.

case its generally fine texture indicated it has a low hydraulic conductivity compared to the overlying Pleistocene deposits.

The new boreholes also provided information on bedrock surface elevations. The bedrock was beyond drilling range in two new boreholes near the southeast corner of section 6 and the southeast corner of section 8. There, the bedrock surface elevation was less than 860 ft (262 m) MSL.

Available data confirmed the conclusions of Kraft et al. (1995) that the Pleistocene unit is typically about 70 ft (20-21 m) thick (Figures 2-2 through 2-7, Appendix 2-1). The range is 37 to 87 ft (11-27 m). It is mainly moderately well- to well-sorted medium sand with interbedded silt and clay subunits. The silty New Rome layer is nearly continuous throughout the study area. It is generally 1-5 ft thick (0.3-1.5 m) with lowest and highest extremes of 953 and 974 ft (290 and 297 m) MSL. The New Rome is absent at some locations, most commonly along the east edge of the study area near the scarp (e.g., location 0817, Figure 2-5). Other fine-textured lenses exist, but are localized and discontinuous.

The base of the Pleistocene unit is usually near 930 ft (284 m) MSL, but sometimes deeper than 903 ft (275 m) MSL. Below the Pleistocene unit and overlying bedrock lie Hillslope deposits or perhaps another unconsolidated unit of unknown origin. These unconsolidated materials are 0 to more than 60 ft thick (0 to >20 m). We encountered bedrock as high as 952 ft (290 m) MSL; at some locations we drilled to 860 ft (262 m) MSL without reaching bedrock. Bedrock appeared to be crystalline except at location 1301, where sandstone was found. The limited extent of sandstone and its low hydraulic conductivity compared to the Pleistocene unit (Clayton, 1986) make it inconsequential to this study.

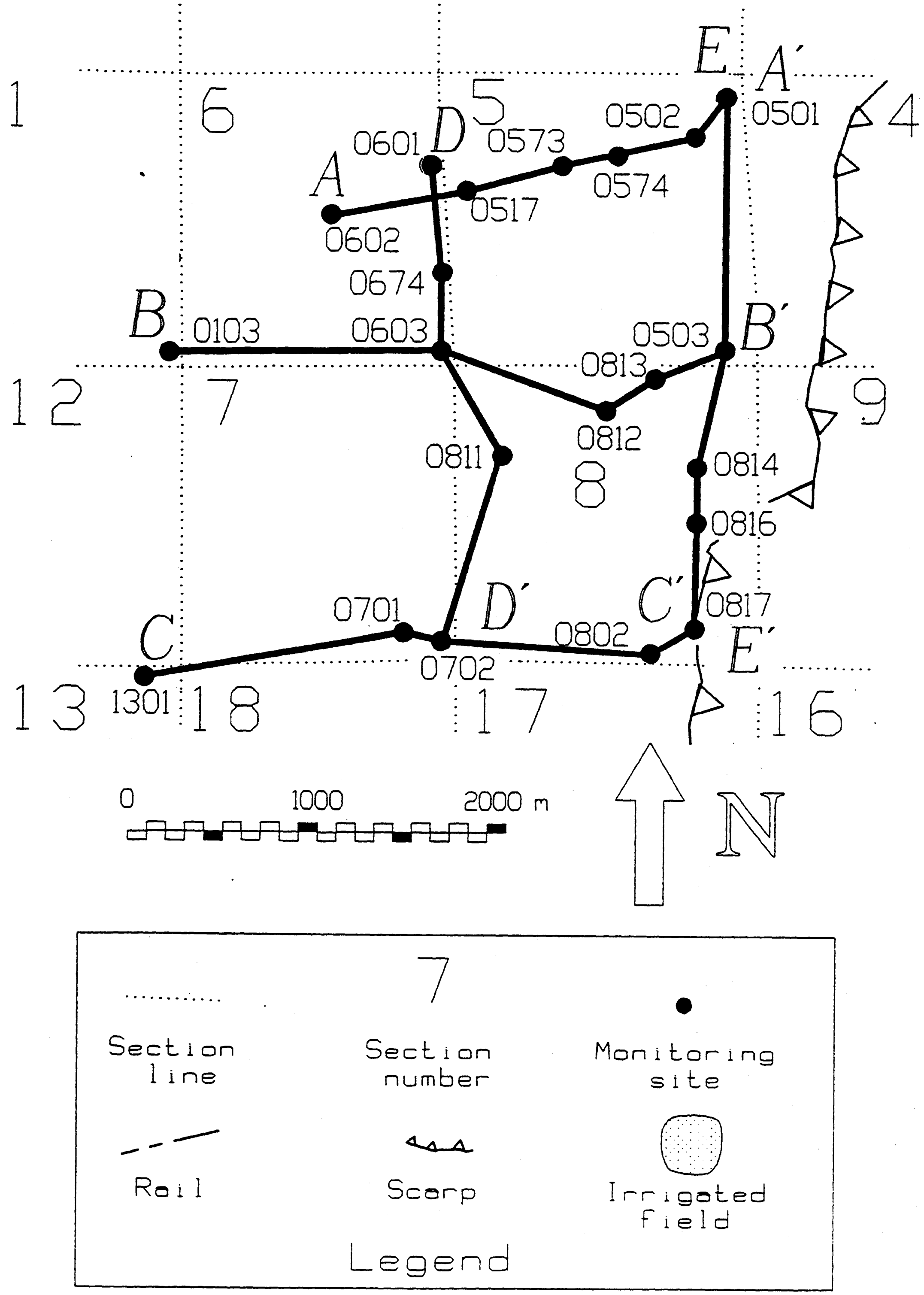


Figure 2-2. Locations of cross-sections.

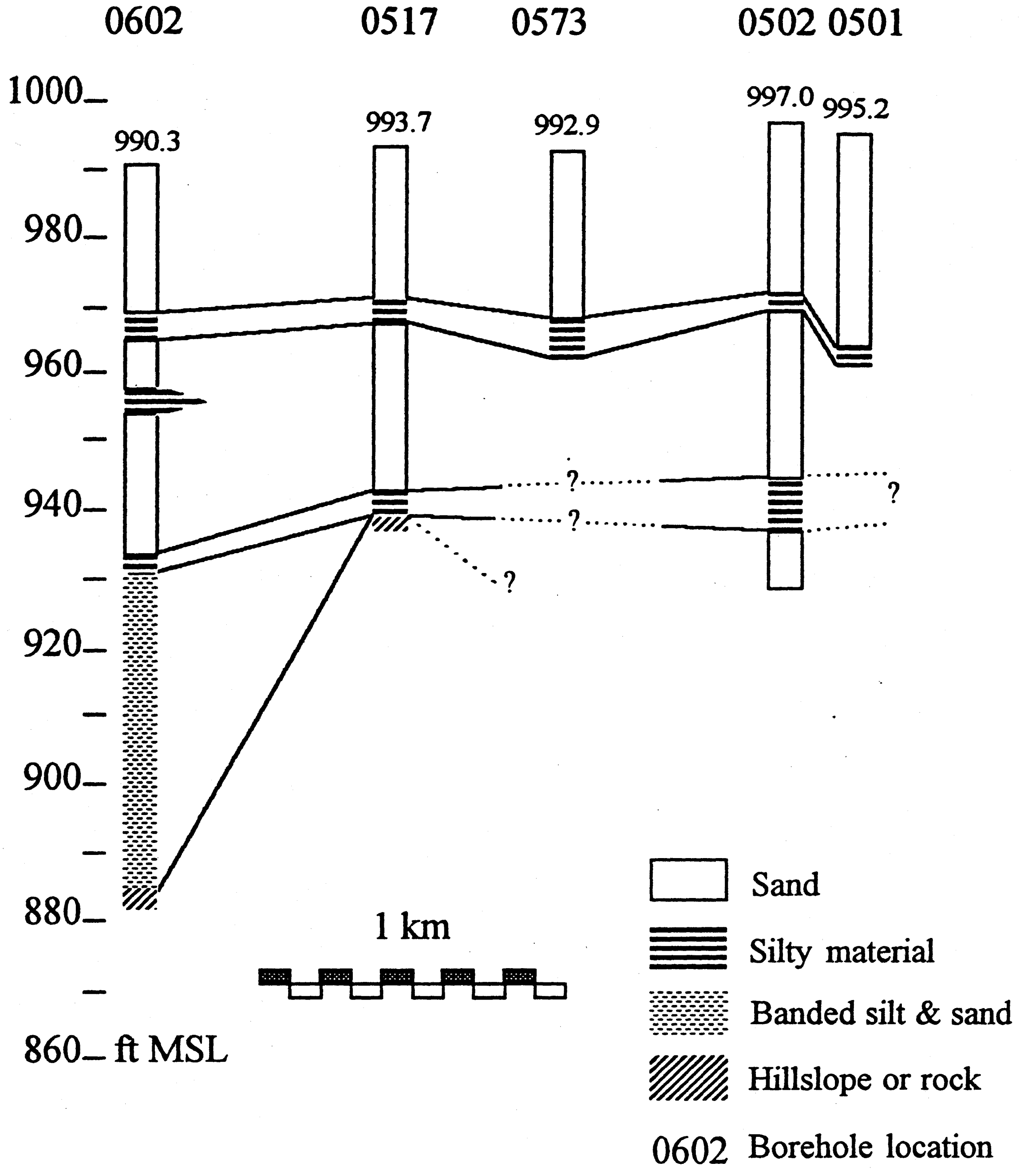


Figure 2-3. Cross-section A – A′, west to east along the north side of the study area. West is to the left.

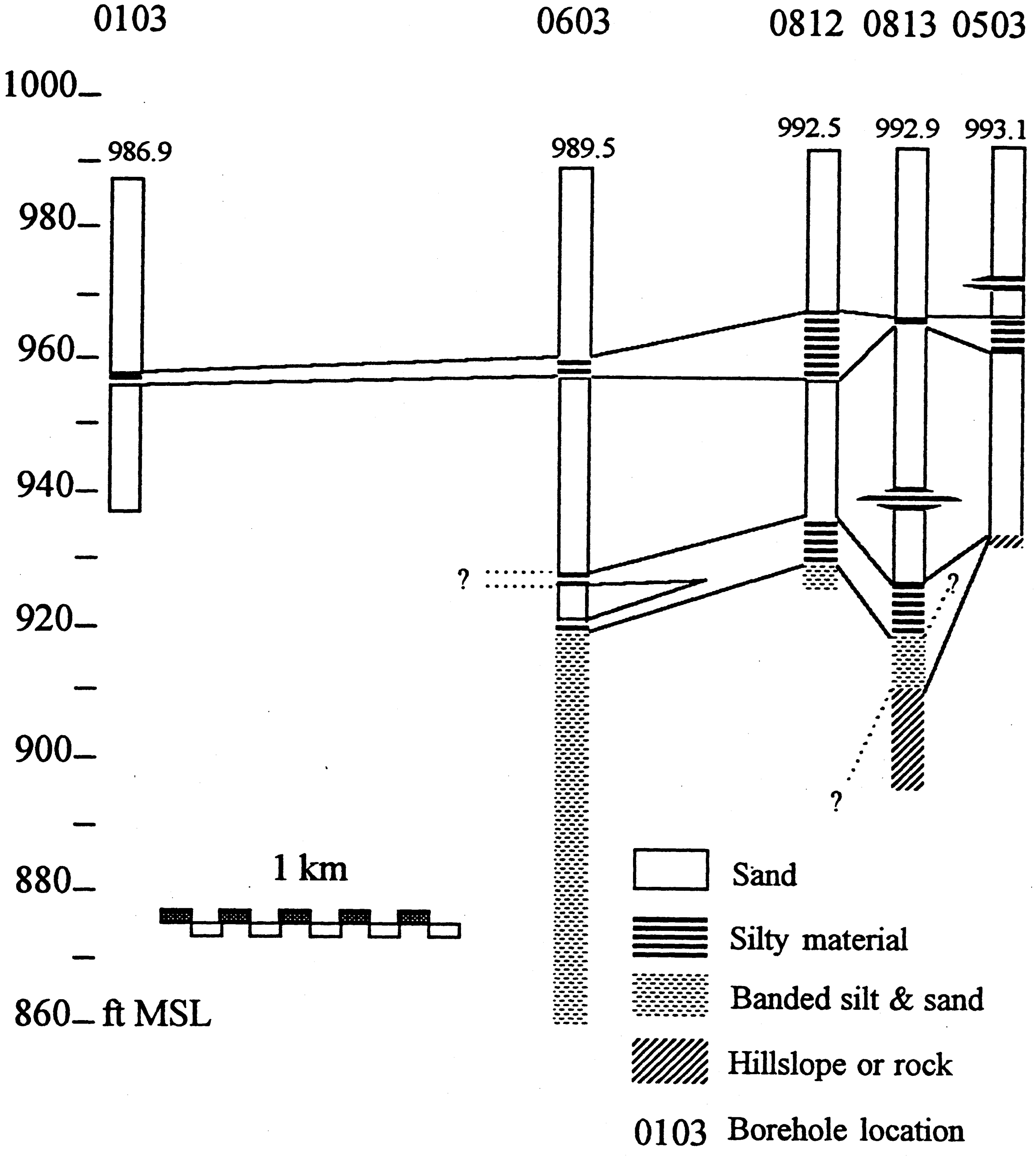


Figure 2-4. Cross-section B – B′, west to east through the middle of the study area. West is to the left.

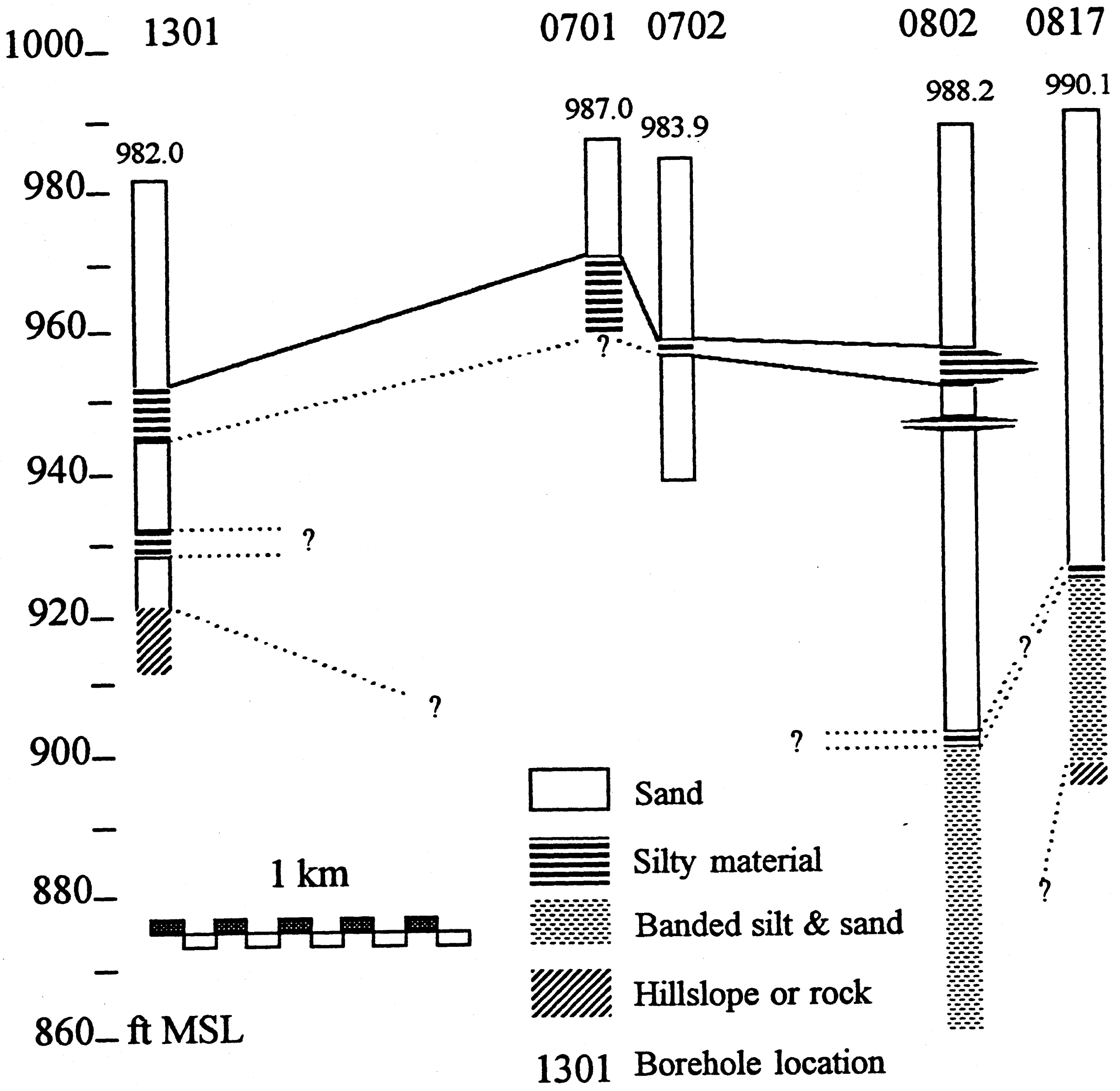


Figure 2-5. Cross-section C – C′, west to east along the south side of the study area. West is to the left.

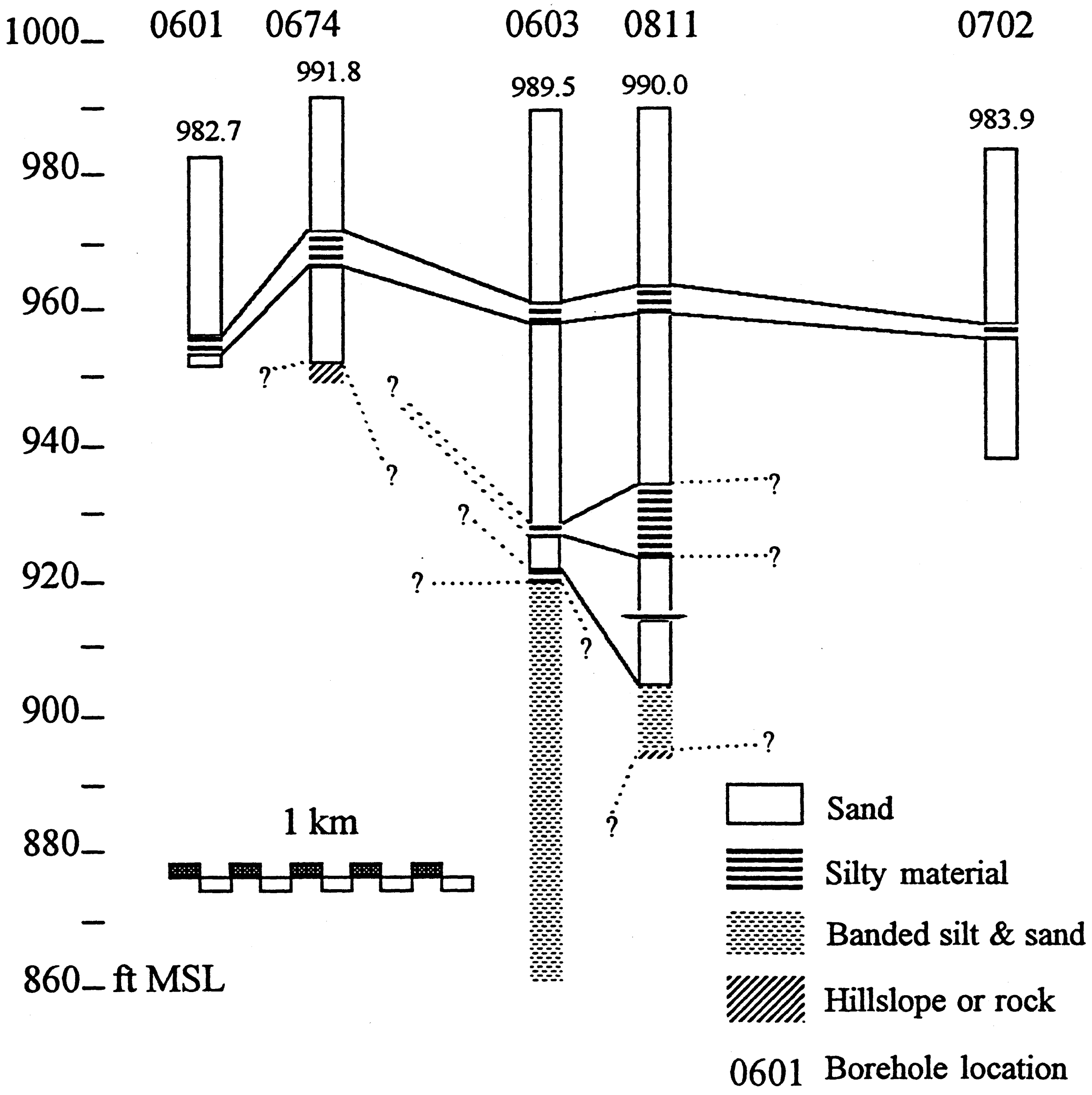


Figure 2-6. Cross-section D – D′, north to south through the middle of the study area. North is to the left.

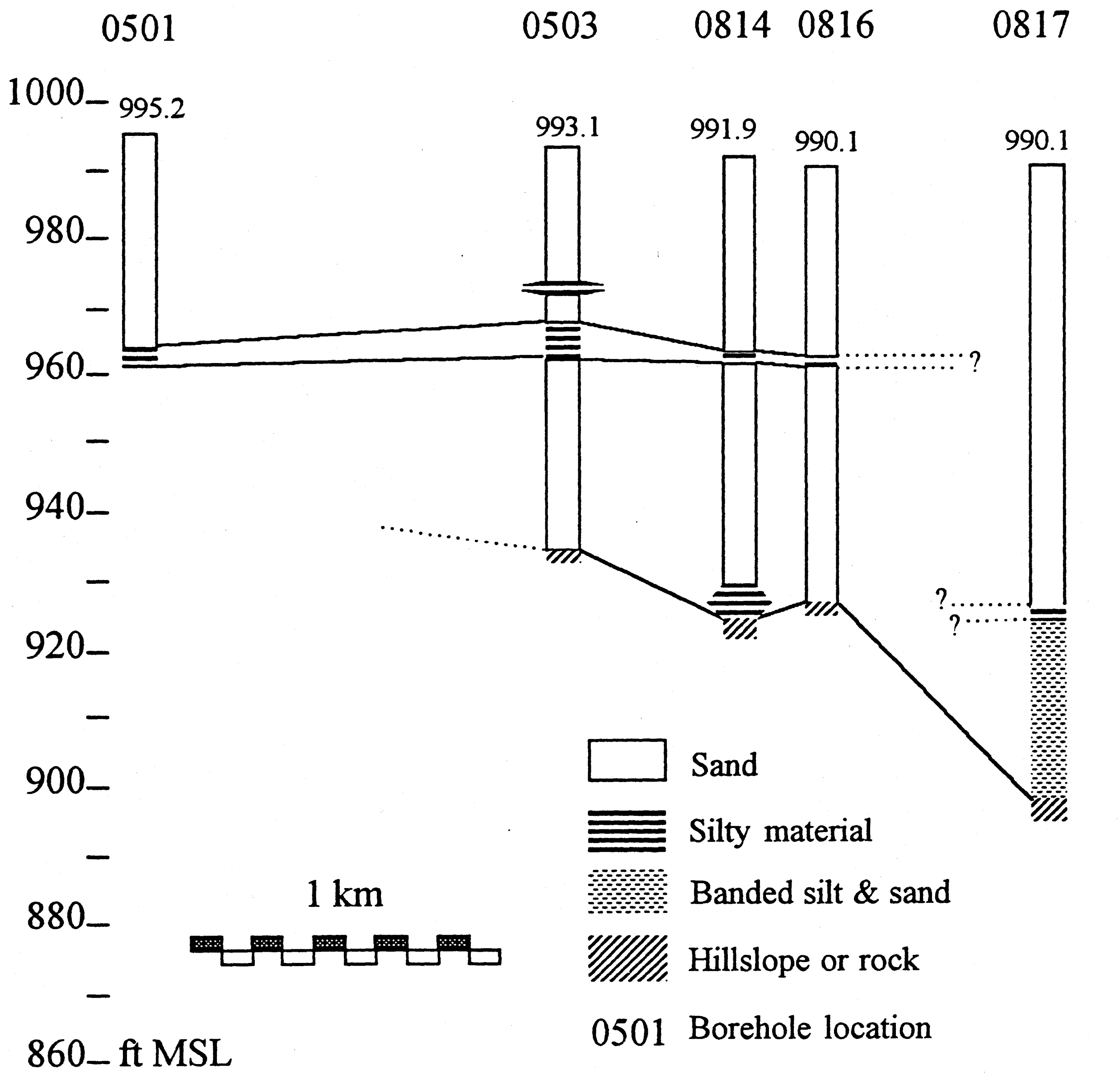


Figure 2-7. Cross-section E – E′, north to south along the east side of the study area. North is to the left.

Hydrogeology

Monitoring well network and head measurements

The monitoring well network installed by Kraft et al. (1995) was present at the start of this study. It consisted of 61 monitoring wells at 25 locations in and near the study area, plus multilevel piezometers (MLPS) in the four irrigated fields in sections 5 and 6. These wells were of 1-inch or 2-inch (2.5-cm or 5-cm) diameter PVC (polyvinyl chloride) pipe, with 5-ft (1.5-m) screens except at location 0701, where all five screens were 2 ft (0.6 m) long. Twelve existing locations with wells completed near the water table were used only for hydraulic head measurements. Thirteen existing locations containing 1-5 screens each were used to monitor water quality and hydraulic head. The MLPs in the irrigated fields will be discussed in Chapter 3.

Forty-two new monitoring wells were installed in July 1995 at 11 locations – three existing and eight new. Well locations were selected to intercept plumes migrating from the irrigated fields in sections 5 and 6. Screen depths were determined by an exploration technique. Exploration was done by installing monitoring wells as deep as possible, then gradually pulling them up, stopping every 1 ft (0.3 m) to analyze groundwater specific conductance. High conductance is a sign of possible agricultural impact on groundwater, and indicated where to install well screens to bracket agriculturally-impacted groundwater at 2-4 depths. All screens were 5 ft (1.5 m) long. Well-installation methods were the same as described by Kraft et al. (1995), and construction details for newly installed wells are in Appendix 2-2.

The elevations of all wells were determined by differential leveling with a laser level. Hydraulic heads in the monitoring wells were measured approximately monthly using an electronic tape.

Groundwater conditions

The Pleistocene deposits form the only aquifer important for the purposes of this study. Groundwater flows mainly in the sand layers and is restricted somewhat by the silty

New Rome member and fine textured lenses. We believe the influence of the less extensive fine-textured lenses is minimal. The effective aquifer base usually is around 70 ft (21 m) below the surface but ranges from 37 to 87 ft.

The water table occurred between about 6 ft (in the northwest) and 28 ft (in the southeast) below the land surface (2-8.5 m). The saturated thickness is typically 50-65 ft (15-20 m), and ranges from 30 to 83 ft (13-25 m). It is greatest at location 0811, where the base of the Big Flats formation is unusually deep and least at location 0674, where crystalline bedrock was uncommonly close to the surface. The saturated thickness was also small near the scarp at the east edge of the area.

Groundwater originates as underflow or recharge and flows southeast, discharging to marshes and streams immediately to the east (Figure 2-8). Water levels fluctuated about ±2.2 ft (0.65 m) about a mean during the 1992-6 period of record (Figure 2-9). Levels were lowest when monitoring began in early 1992, and peaked in June 1993 coincident with record rains experienced that spring. Water levels decreased from that peak and remained fairly steady during 1994 and 1995. The water-table elevation in the study area changed more or less in unison, so that the direction and speed of flow were approximately constant.

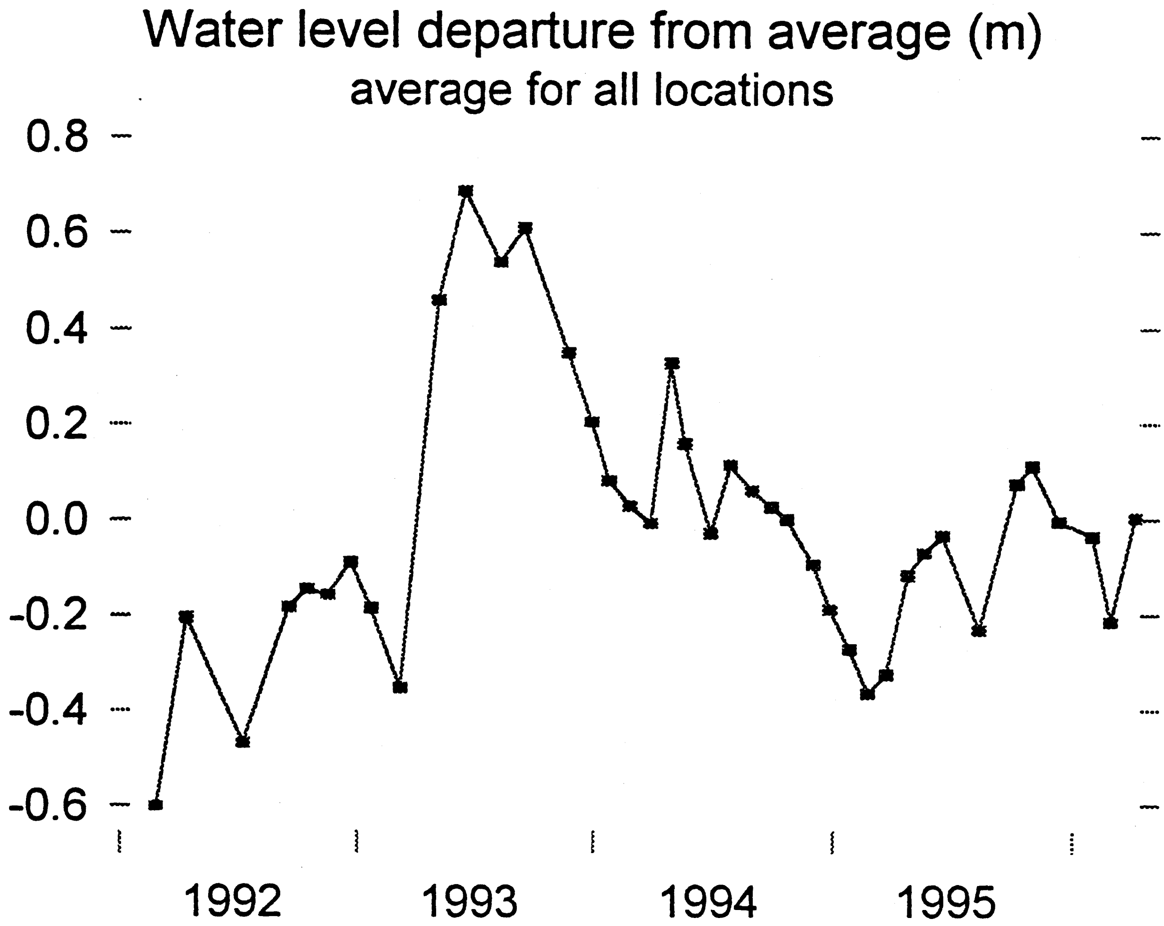


Figure 2-8. Water level departure from 1992-1996 mean. Each point represents the average for all locations on the given date.

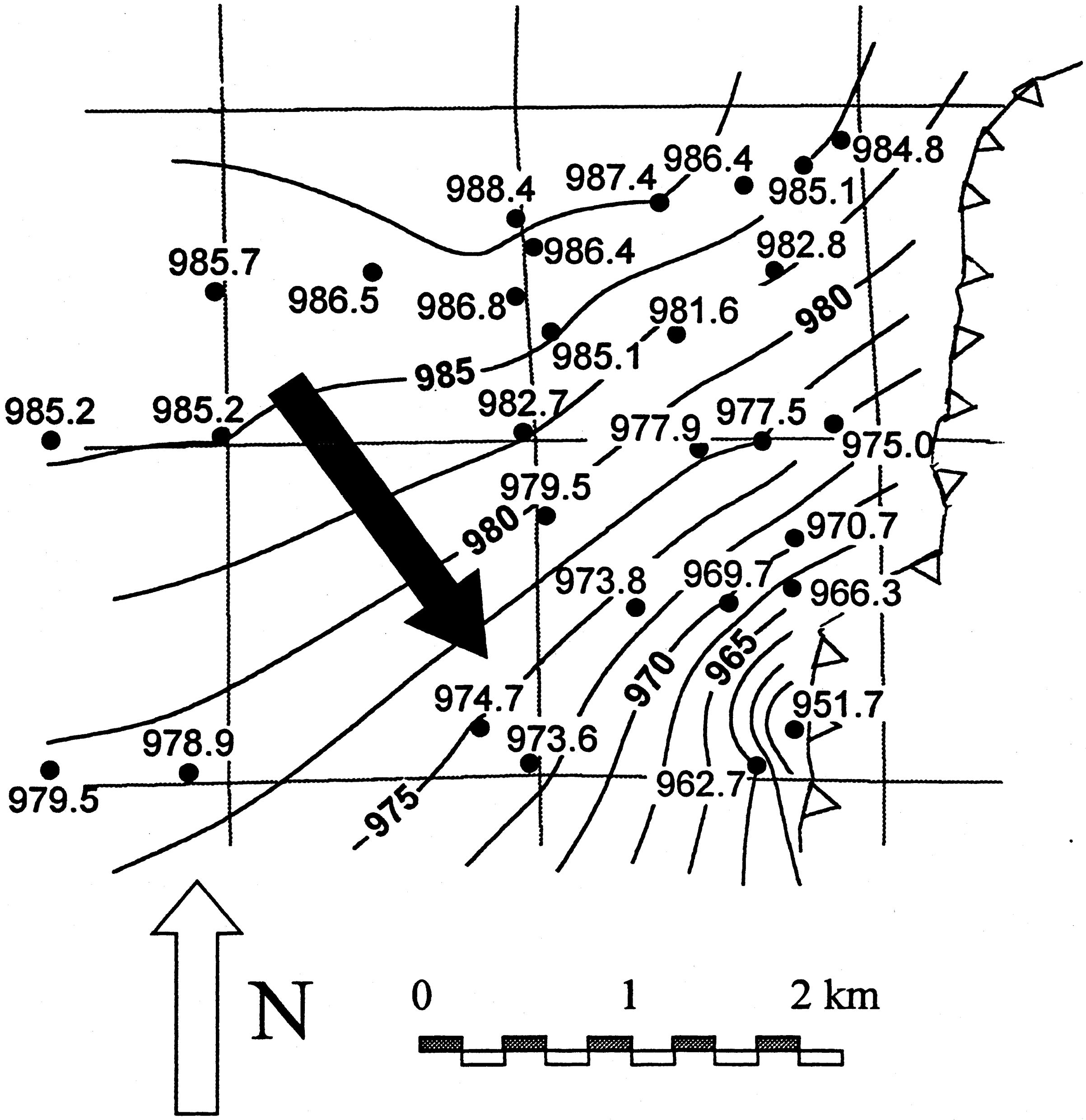


Figure 2-9. Water table elevations (ft MSL) on 9 April 1996. The gray arrow shows the approximate direction of groundwater flow.

Chapter 3
Groundwater quality under an irrigated vegetable field

This chapter describes groundwater monitoring results for Field 2, located in the north part of the study area (Figures 1-2, 3-1). The field was previously monitored by Kraft et al. (1995) from January 1992 through April 1994. The present study extended the monitoring through March 1996. Monitoring objectives were to

- compare groundwater quality under the field to upgradient groundwater quality
- evaluate rates of NO_3 loading to groundwater
- determine the effectiveness of reduced agrichemical inputs for protecting or maintaining acceptable groundwater quality
- examine pesticide leaching to groundwater.

Field 2 is 44 ha (108 acres) in size. Its soil is mapped as Plainfield loamy sand, and its underlying geology consists of about 70 ft (21 m) of Pleistocene deposits overlying relatively impermeable rocks or sediments. The Pleistocene deposits are predominantly medium sand typical of most of the study area, with the silty New Rome member about 20-32 ft (6-10 m) below the surface. The New Rome is approximately 1-5 ft thick (0.3-1.5 m). The water table is about 8-10 ft (2.5-3 m) below the surface, and groundwater flows to the south-southeast at an estimated 0.1 m d^{-1}. Upgradient land use is primarily forest. A town road lies about 30 m north of the upgradient field edge.

Field history and inputs

This field was probably brought into irrigated production between 1975 and 1978 (Kraft et al., 1995). The prior land use was about 40% forest and 60% non-irrigated crop-land. The known crop history before this study is 1993 potato, 1992 and 1991 sweet corn (*Zea mays* L.), 1990 potato (*Solanum tuberosum* L., var. Russet Burbank), 1989 and 1988 sweet corn, and 1987 pea (*Pisum sativum* L.).

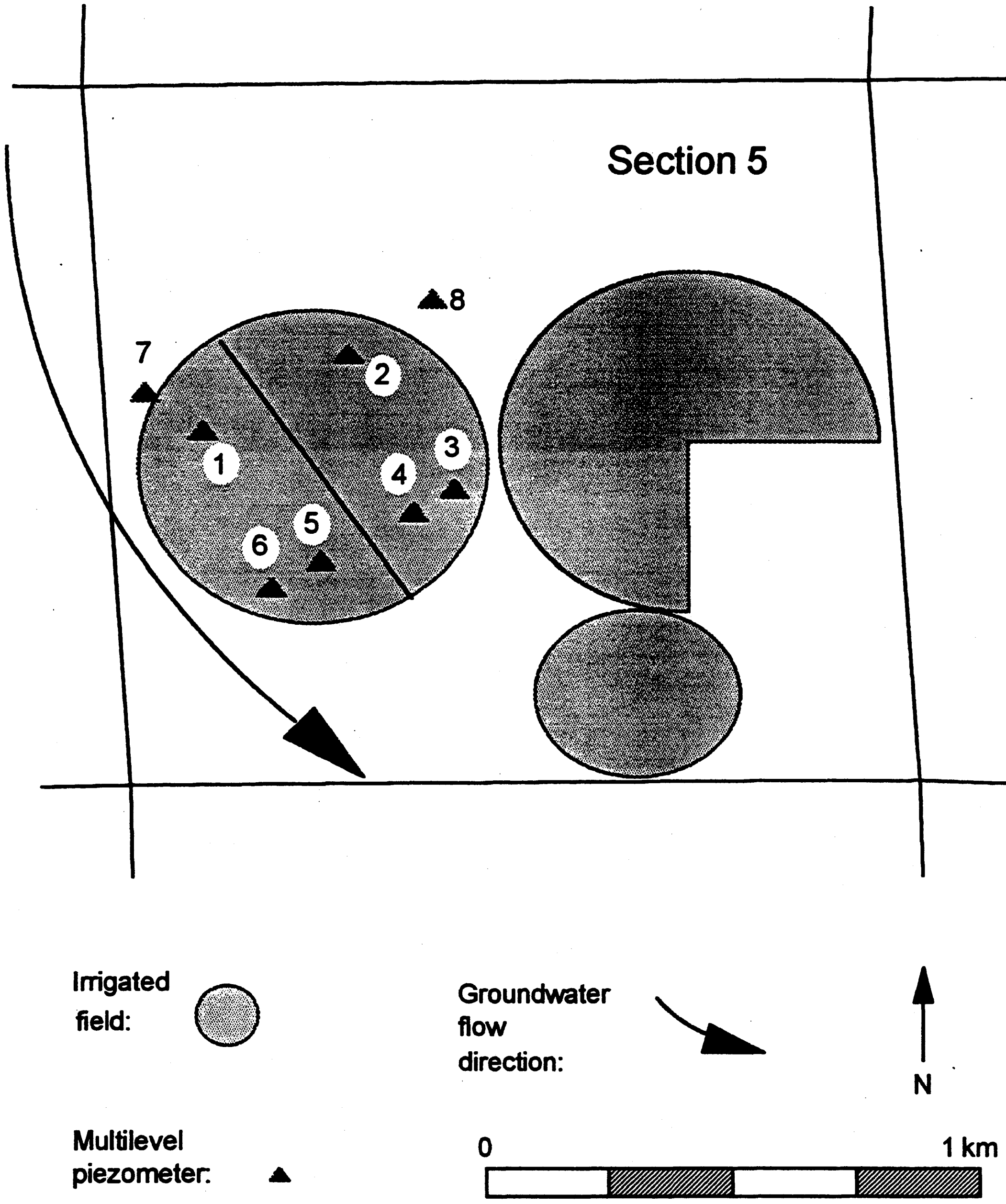


Figure 3-1. Location and numbering of multilevel piezometers in Field 2 and immediately upgradient, with approximate groundwater flow direction.

Sweet corn was grown in both 1994 and 1995. In 1994, the variety was “More”. The crop was planted on 30 May and harvested 2 September. In 1995, 72% of the field was planted with “Heritage” and 28% with “Eliminator”. The planting date was 15 May, and harvest was on 16 August. The sweet corn yield was 21.1 Mg ha^{-1} in 1994 and 20.0 Mg ha^{-1} in 1995. According to grower records, 70 mm of irrigation water was applied in 1994, and the amount was not recorded in 1995. Seventy mm is a surprisingly small irrigation total and may not be accurate; a comparable experimental field at Hancock received 184 mm in 1994. Agrichemical applications are detailed in Tables 3-1 and 3-2, and Table 3-3 summarizes inputs in

Table 3-1. Chemical inputs to study field, 1994. Amounts are kg ha^{-1} of the element or active ingredient.

Date	Material	Amount	Remarks
9 March	K	120	215 lb/A of 0-0-60
	Cl	109	215 lb/A of 0-0-60
28 May	N	16.2	130 lb/A of 11.1-14.7-14.7
	P	9.4	
	K	17.8	
	Cl	16.2	
	S	14.1	
	Mg	10.8	
	Zn	2.0	
	B	0.15	
1 June	Alachlor	1.68	West half
	Atrazine	1.12	West half
	Metolachlor	1.68	East half
	Atrazine	0.84	East half
17 June	N	52	Fertigation
27 June	N	77	Fertigation
15 July	N	33	Fertigation
24 July	N	29	Fertigation
6 August	methyl parathion	?	Penncap-M
Total	N	207	
	Cl	125	

1992 and 1993. Fertilizer-N applications were greater than University of Wisconsin – Extension recommendations during the 1992-5 period, except for 1995 (Table 3-4). Nitrogen applications to sweet corn were 176 to 250 kg ha^{-1}, compared to the University recommendation (Kelling et al., 1991) of 179 kg ha^{-1}. Applications to potato in 1993 were 297 (north) or 357 (south) kg ha^{-1}, compared to the University recommendation of 258 kg ha^{-1}.

Table 3-2. Chemical inputs to study field, 1995. Amounts are kg ha^{-1} of the element or active ingredient. Exact early-season dates are not known.

Date	Material	Amount	Remarks
March/April	K	117	K fertilizer, 200 lb/acre
	Cl	106	
May	N	17	starter, 172 lb/acre
	P	10	9.0-11.6-17.4-12.7 S -
	K	28	1.1 Zn - 5.7 Mg - 0.1 B
	Cl	30	
	S	25	
	Zn	2	
	Mg	11	
	B	0.2	
	Atrazine	1.0	
	Alachlor	1.68	
7 June	N	36	Fertigation
14 June	N	58	Fertigation
19 June	N	30	West: Fertigation
		40	East: $(NH_4)_2SO_4$
28 June	N	36	West: Fertigation
		22	East: Fertigation
Total	N	178	West
		174	East
	Cl	137	

Table 3-3. Summary of chemical inputs to study field, 1992 and 1993. Amounts are kg ha^{-1} of the element or active ingredient.

		Nutrient Applications		Other
	Crop	No.	Total NPK	Applications
1992	Sweet corn	7	250N, 12P, 109K	atrazine metolachlor(W) alachlor(E)
1993	Potato	6	north 297N, 54P, 389K south 357N, 54P, 389K	metolachlor metribuzin carbofuran sulfur esfenvalerate chlorothalonil methamidophos diquat

Table 3-4. Comparison of fertilizer recommendations from the University of Wisconsin - Extension and actual N fertilizer applications (kg ha^{-1}) to Field 2.

Year	Crop	Recommendation	Application	Excess
1992	Sweet corn	179	250	71
1993	Potato	258	297 / 357[1]	39 / 99[1]
1994	Sweet corn	179	207	28
1995	Sweet corn	179	176[2]	-3

[1] North and south halves, respectively

[2] Average of 174 (east) and 178 (west)

Instrumentation

The study field was instrumented by Kraft et al. (1995) in 1991. A dividing line was laid out along a groundwater flow line (Figure 3-1), so that different agricultural practices could be implemented on each half. Three multilevel piezometers (MLPs) were installed in each field half, for a total of six "in-field" MLPs. Two "upgradient" MLPs were installed off the north edge of the field.

MLPs initially consisted of 18 polypropylene tubes of 6-mm inside diameter attached to a 32-mm (1¼ in. nominal) "backbone" PVC pipe (Figure 3-2). Each tube terminated in a nylon-mesh screen, as did the PVC pipe, providing a total of 19 ports. Ports were numbered from 0 at the bottom to 18 at the top. The ports were 15, 20 or 40 cm long, providing continuous screening over a length of 340 cm. In early 1994, five new, shallower ports were added to each in-field MLP after the water table rose due to heavy 1993 rains. In-field MLPs then had 24 ports each, spanning 425 cm.

Methods

Sampling

MLPs were sampled 17 times during the current study, beginning on 3 March 1994 and ending 30 March 1996. The samples were collected by suction with a peristaltic pump. The sampling interval was approximately monthly, except during the coldest part of the year, and when the MLPs were buried to allow farming operations. Samples were collected from the six in-field MLPs and one of the two upgradient MLPs (the two were alternated on each sampling date). Even-numbered ports were sampled, plus port nos. 1, 17, 19, 21, and 23. If a desired port could not be sampled, sampling was attempted from an adjacent port. The top few ports could only be sampled when the water table was sufficiently high.

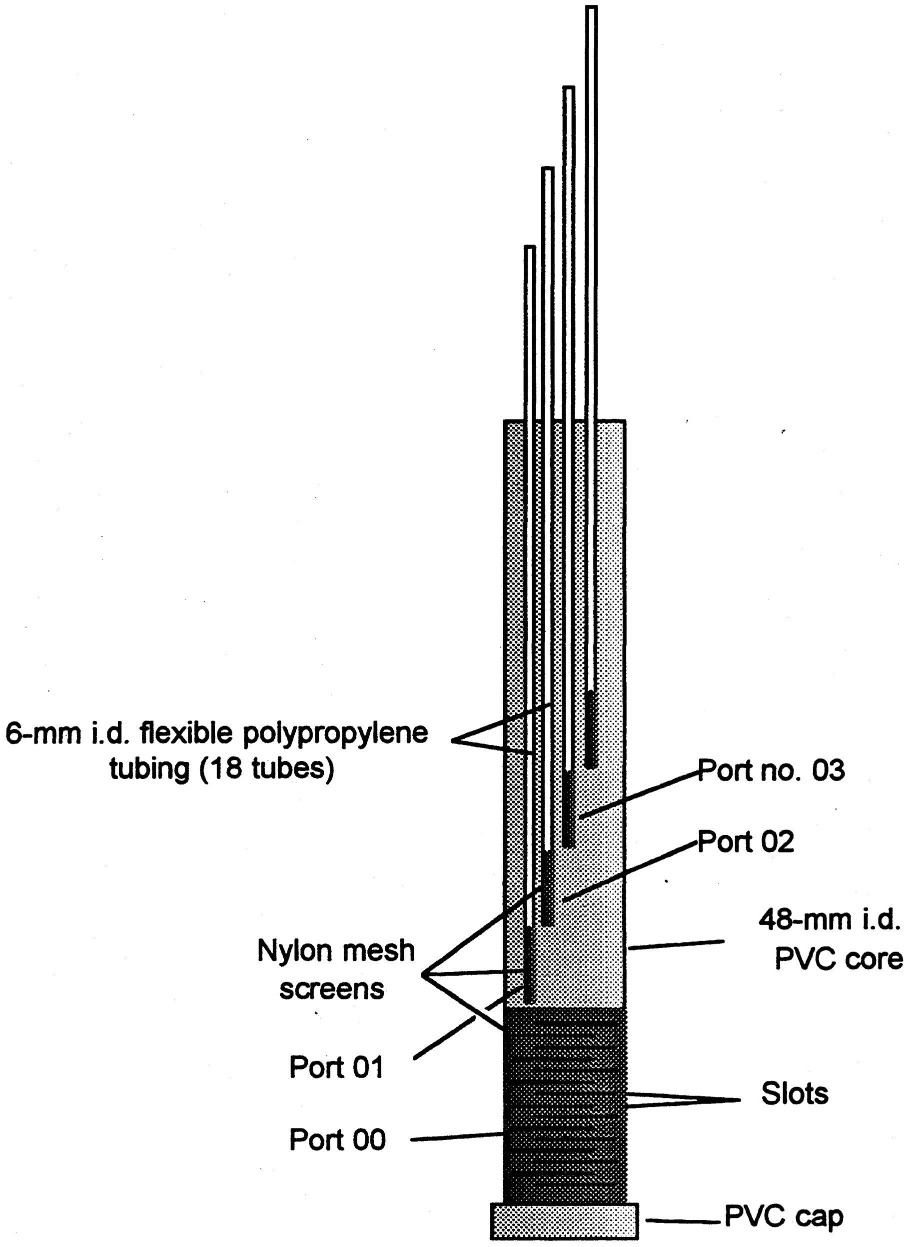


Figure 3-2. Schematic drawing of a multilevel piezometer. Only five ports are shown.

Analyses

Details of inorganic analyses are given in Kraft et al. (1995). All MLP samples were analyzed for NO_3, Cl, pH, and conductance. Conductance and pH were measured in the field; *ex situ* sample temperatures were measured to adjust conductance measurements to a 25 °C basis. Nitrate and Cl analyses were done by automated colorimetry (Lachat, 1992; 1991) at the UW-SP Environmental Task Force Laboratory (ETF). The reporting limits were 0.2 mg L^{-1} for NO_3-N and 1 mg L^{-1} for Cl. Samples for pesticide analysis were collected 30 November 1995. Pesticide analyses were also done at the ETF, using a modified version of EPA method 8270 (USEMSL, 1988) capable of detecting and quantifying 27 pesticide residues (Table 3-5).

Table 3-5. Method detection limits and limits of quantitation for pesticide analyses.

Compound	MDL	LOQ	Detected?	Compound	MDL	LOQ	Detected?
Alachlor	0.1	0.3		Metribuzin	0.1	0.4	*Y*
Atrazine	0.1	0.4	*Y*	Molinate	0.1	0.4	
Desethylatrazine	0.1*e*	0.5*e*	*Y*	Oxidiazon	0.1	0.2	
Desisopropylatrazine	0.2*e*	1.0*e*	*Y*	Oxyfluorfen	0.1	0.3	
Benfluralin	0.1	0.4		Pebulate	0.1	0.3	
Bromacil	0.3	1.1		Pendimethalin	0.1	0.3	
Butylate	0.1	0.3		Profluralin	0.1	0.3	
Carbofuran	0.2	0.5	*Y*	Propachlor	0.1	0.3	
Cyanazine	0.2	0.7		Propazine	0.1	0.3	
Cycloate	0.1	0.3		Simazine	0.1	0.5	
EPTC	0.1	0.4		Terbacil	0.3	1.0	
Hexazinone		1.0*e*		Trifluralin	0.1	0.3	
Isopropalin	0.1	0.3		Vernolate	0.1	0.2	
Metolachlor	0.1	0.5	*Y*				

e - estimated limit

Y - Pesticide residue was detected beneath the field

Many analyses resulted in a finding that the sample concentration was below the limit of quantitation for the analytical method; such concentrations are hereafter called "nondetects". This occurred commonly with NO_3 and Cl in upgradient MLPs. When several of the observations in an MLP were well above the reporting limit, statistical calculations are barely affected whether nondetects are assigned a value of 0, the reporting limit, or some intermediate value. We chose one-half the reporting limit in such cases. With pesticides, however, nondetects were very common to dominant. Therefore, means for pesticide residues were calculated for detects only.

Statistical tests such as analysis of variance were evaluated at a Type I error rate of 5% ($P < 0.05$). The statistical method of "contrasts", an elaboration of analysis of variance (Sokal and Rohlf, 1981), was used to perform means testing for NO_3 and Cl concentrations in MLPs. A set of six contrasts was calculated in order to compare each one of the in-field MLPs to all five others. The quantities compared were depth-weighted average concentrations for each MLP on each sampling date (225 degrees of freedom). If the nominal Type I error rate of 5% is to apply, only five contrasts can be calculated with six data groups (six MLPs). Taking six contrasts, as here, causes the Type I error rate to be larger than indicated in the calculations, but the increased error rate is modest (Sokal and Rohlf, 1981). We compensated by requiring significant P values to be less than 0.025 rather than 0.05.

Depth-weighted concentration averages were determined by assigning each observation a weight according to its depth increment. A given port's increment extended upward and downward from the center of the port halfway to the center of the next port from which a sample had been collected. Time weights were calculated analogously, with divisions halfway between successive sampling dates.

Results

The MLPs in Field 2 were sampled 17 times during the current study (3 March 1994 to 30 March 1996) and 22 times by Kraft et al. (1995), for a total of 39 sampling events beginning 21 January 1992. Nitrate-N, Cl, pH, and specific conductance were analyzed on nearly every sampling event. More detailed suites of inorganic parameters were analyzed on 3 occasions by Kraft et al. (1995). Pesticides were analyzed once during the current study, and three times in the previous study.

Inorganic groundwater quality

Analyte values displayed time, depth, and locational dependencies. Nitrate-N concentrations in individual samples ranged from <0.2 to 50.5 mg L^{-1}, and Cl from <1 to 119 mg L^{-1}. The pH range was 3.72 to 7.27, and specific conductance was 1.8 to 73.8 mS m^{-1}. Nitrate and Cl concentrations were higher under the field than upgradient, and NO_3 concentrations were greater in shallow groundwater than deep.

Field averages

Average solute concentrations in in-field MLPs are a useful measure of groundwater quality in the monitored zone (approximately the upper 3 m of groundwater) beneath the field during the monitoring period. We calculated time- and depth-weighted analyte means for each in-field MLP, and then averaged the six means to yield a grand mean representative of groundwater conditions for the period of record.

The time- and depth-weighted average NO_3-N concentration for each of the six in-field MLPs was between 15.2 and 22.3 mg L^{-1} for the period of record (Table 3-6), and averaged 20.5 mg L^{-1}. Statistical contrast analysis showed NO_3 (and Cl) concentrations were low in MLP 2, probably arising from that MLP's position closest to the upgradient edge of the field. Water sampled from the deepest ports in MLP 2 was quite low in NO_3 and Cl compared to other MLPs. We interpret this as evidence of an "underflow" effect, that is, water in the deepest ports of MLP 2 originated upgradient of the field, and thus was unaffected by agricultural chemicals (see the "Identifying solute pulses" section below). Nitrate-N concentrations in the lower ports of MLP 1 also showed some effects of underflow, but only intermittently and in fewer ports. We attribute the smaller underflow effect in MLP 1 to its being twice as far from the upgradient field edge as MLP 2. Underflow was not seen at all in the other four in-field MLPs. Contrasts also showed NO_3 concentrations were large in MLPs 1, 4, and 6, for unknown reasons. It may be uneven fertilizer application, or variability in topography, soil properties, or the vadose-zone flow regime.

Table 3-6. Time- and depth-weighted-average NO_3-N and Cl concentrations (mg L^{-1}) for each in-field MLP, and contrasts showing approximate (see text) significance (*P*) of the deviation of individual MLP means from the grand mean. Pooled variance estimates were used.

	NO_3-N		Cl	
MLP	conc.	*P*	conc.	*P*
1	21.8	*0.016	18.4	0.113
2	15.2	*0.000	13.9	*0.000
3	21.6	0.027	24.9	*0.000
4	22.1	*0.009	22.3	0.124
5	19.9	0.105	21.3	0.296
6	22.3	*0.000	20.6	0.862
All	20.5		20.2	

*Significant at approximately the 0.025 level.

The time- and depth-weighted average Cl concentrations for each of the six in-field MLPs varied from 13.9 to 24.9 mg L^{-1}, and the mean of the six was 20.2 mg L^{-1} for the period of record. Contrast analysis showed that Cl concentrations were significantly low in MLP 2, presumably due to underflow, and high in MLP 3 possibly because of variable field properties or uneven fertilizer application.

The time- and depth-weighted average pH in the six in-field MLPs ranged from 4.23 to 5.90, and the mean among MLPs was 4.90. Specific conductance ranged from 18 to 47 mS m^{-1} (time- and depth-weighted, by MLP), and averaged 33.2 mS m^{-1}.

Upgradient MLPs

Nitrate, Cl and specific conductance were much less in upgradient MLPs than in-field MLPs (Table 3-7). The NO_3-N depth- and time-weighted average for both upgradient MLPs combined was 0.7 mg L^{-1}. Depth-weighted average NO_3-N concentrations did not exceed 1.4 mg L^{-1} on any sampling date in either upgradient MLP; the greatest single-port concentration was 8 mg L^{-1}.

Table 3-7. Time- and depth-weighted average concentrations of inorganic analytes in upgradient MLPs, compared to in-field MLPs for the period of record.

	Upgradient MLPs		
Parameter	MLP 7	MLP 8	In-field average
NO_3-N (mg L^{-1})	0.6	0.8	20.5
Cl (mg L^{-1})	1.8	0.9	20.2
Spec. conductance (mS m^{-1})	5.7	5.7	33.2
pH	4.93	5.17	4.90

The upgradient time- and depth-weighted average Cl concentration was 1.4 mg L^{-1}. Cl concentrations at MLP 7 usually averaged 2 mg L^{-1} or less, but in early 1995 a strong pulse containing concentrations up to 26 mg L^{-1} was recorded, and the depth-weighted average reached 12 mg L^{-1} on one sampling date. In MLP 8, the average Cl concentration was seldom greater than 1 mg L^{-1}, and no distinctive peak was seen. Upgradient specific conductance values were low and fairly steady. In each upgradient MLP the time- and depth-weighted average was 6 mS m^{-1}, the maximum was 8, and minima were 3 and 4. pH values were somewhat greater in the upgradient MLPs (5.07) than in-field MLPs (4.90).

The only marked difference between the two upgradient MLPs was the Cl pulse in MLP 7. Both MLPs are within 30 m of a rural road, but MLP 7 is near and downgradient from a corner and a rail crossing, which received more road salt in winter. The rest of the road received very little salt.

Temporal changes

Depth-weighted average NO_3-N and Cl concentrations varied substantially with time (Figure 3-3). For each sampling date, depth-weighted average NO_3-N and Cl concentrations were calculated at each in-field MLP. These ranged from 9.1 to 32.8 mg L^{-1} of NO_3-N, and 4.4 to 46.5 mg L^{-1} of Cl during the period of record. Averaging together the depth-weighted NO_3-N concentration in the six in-field MLPs on each date (Figure 3-3) produced a range of 16.2 to 28.1 mg L^{-1} and a time-weighted mean of 20.5 mg L^{-1} for the period of record. The Cl concentration averaged over all six in-field MLPs on each date ranged from 10.3 to 36.4 mg L^{-1} (time-weighted mean, 20.2 mg L^{-1}).

Chloride concentrations were more variable with time than NO_3 (Figure 3-3). Probable reasons for this greater variability are first, that the amount of Cl applied varied more than N from year to year (Tables 3-1 – 3-3). The very large Cl application in 1993 was reflected as a marked increase in 1994 Cl concentration. Second, Cl was only applied early in the year, whereas nitrogen fertilizer was applied several times during the growing season, leading to a steadier NO_3-N concentration in the recharge water. Third, nitrogen is much more subject to retention in biomass. These reasons may explain how the large 1993 N application caused only a moderate increase in average NO_3-N concentration in 1994 (Figure 3-3).

Depth-weighted average NO_3-N concentrations in upgradient MLPs on each sampling date ranged from 0.3 to 1.4 mg L^{-1}. Nitrate concentrations did not change strikingly through time (Figure 3-3). The range of upgradient Cl depth-weighted average concentrations was from undetectable to 11.5 mg L^{-1}. Chloride concentrations were much greater in 1995 than other times. Although road salt is believed to be the source of the Cl, it is unclear why the pulse was so large in 1995.

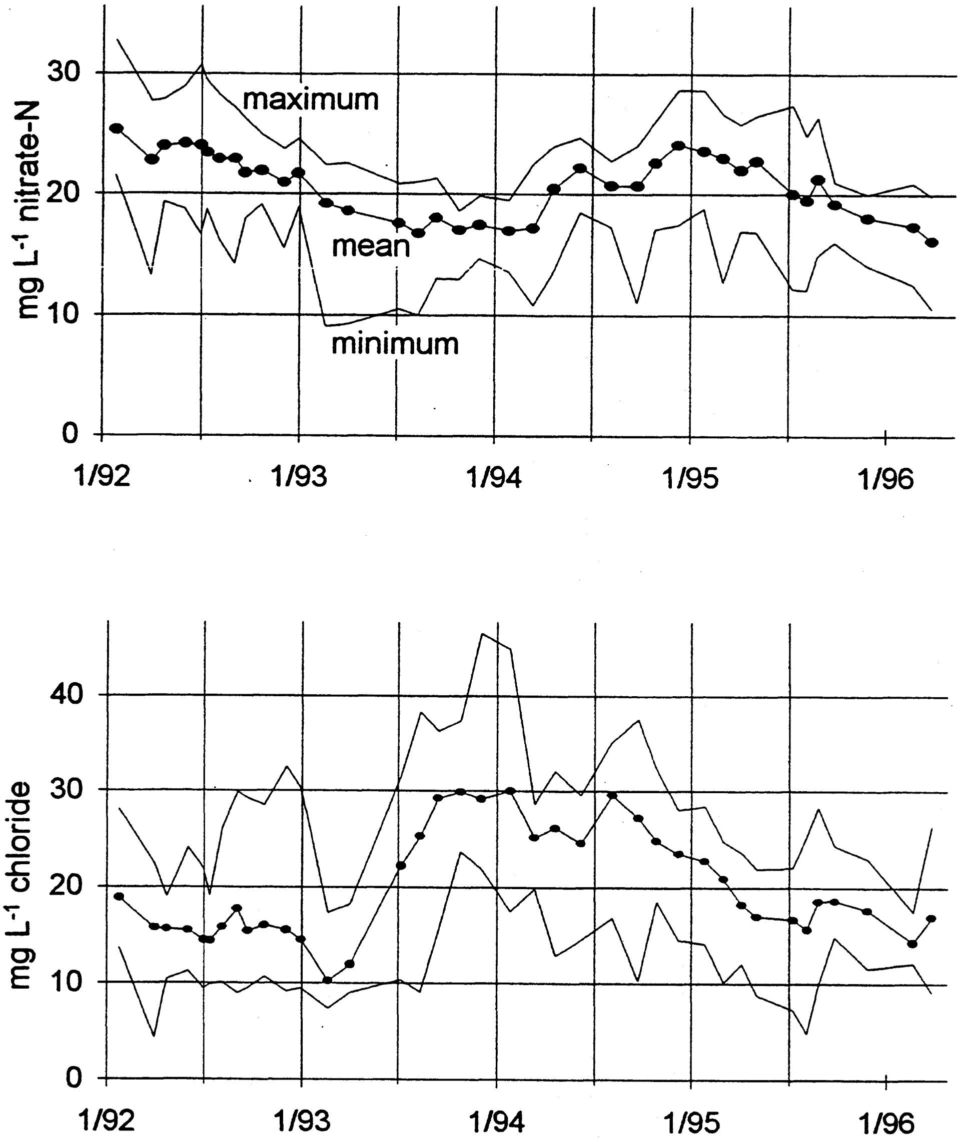
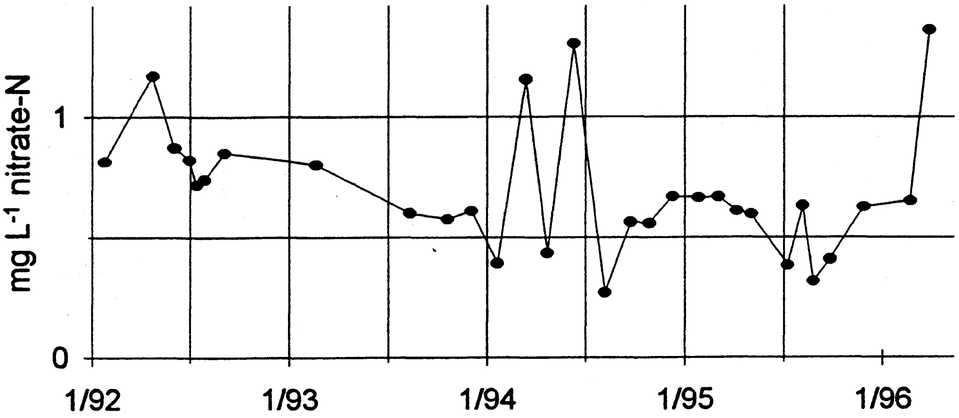


Figure 3-3. Maximum, mean, and minimum depth-weighted average concentrations of NO_3-N and Cl on each sampling date for in-field MLPs.



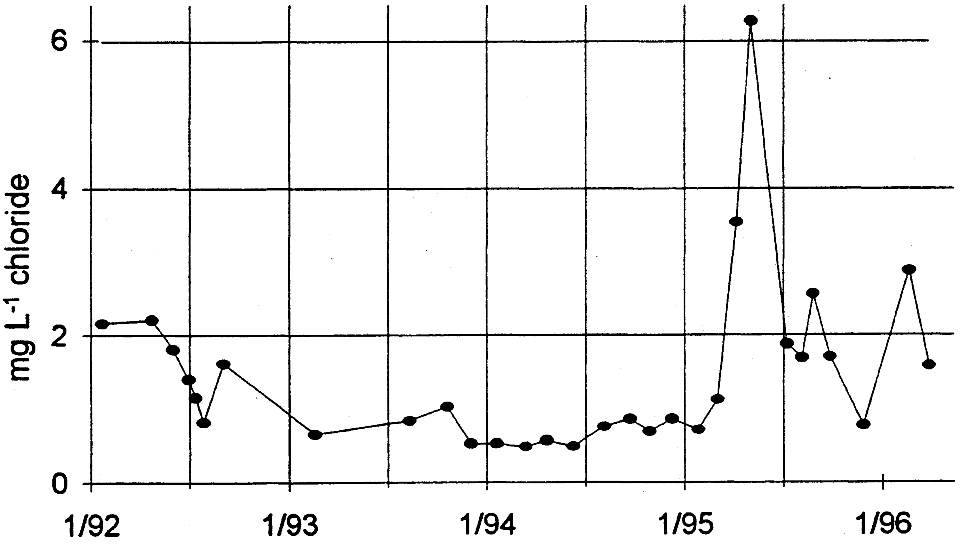


Figure 3-3, continued. Mean depth-weighted average concentrations of NO_3-N and Cl in upgradient MLPs on each sampling date. Nondetects were assigned one-half the detection limit.

Concentration-depth relationships

To detect depth variations in analyte concentrations within the field, we calculated a time-weighted average concentration for every in-field MLP port (i.e., constant relative elevation). We excluded ports sampled fewer than 10 times. These time-weighted concentrations were regressed against elevation (relative to the installation datum). Nitrate-N concentrations declined significantly and rather steadily with depth (Figure 3-4). The regression coefficient predicts a decrease of 2.8 mg L^{-1} per meter of depth for NO_3-N. The decreasing concentration with depth is likely due to dispersion of solute-laden water originating from the field into low-solute water originating upgradient of the field. Chloride concentrations varied with depth, but not as linearly (Figure 3-4), and the regression was not significant. Nitrate-N and Cl concentrations did not vary with depth in upgradient MLPs. Specific conductance and pH did not vary with depth in the field or in upgradient MLPs.

Identifying solute pulses

A profile of solute concentration vs. depth is a snapshot of concentration-depth relations at one sampling time. A series of profiles can show vertical movement of a solute pulse over time. MLP 6 was chosen to illustrate solute movement (Figure 3-5). The April 1993 profile shows a pulse of Cl from the 1992 growing season between the tick marks at elevations 1.75 and 3 m. Water below 1.75 m was recharged in 1991; 1993 water had not yet appeared. The 1993 pulse arrived by September 1993, and 1991 water had probably disappeared below the sampler. In June 1994, most of the sampling depth was occupied by 1993 water, and a 1994 Cl pulse had appeared above about 3 m. By August 1995, the 1995 Cl pulse was clearly present, and the previous two years' bands could be discerned below.

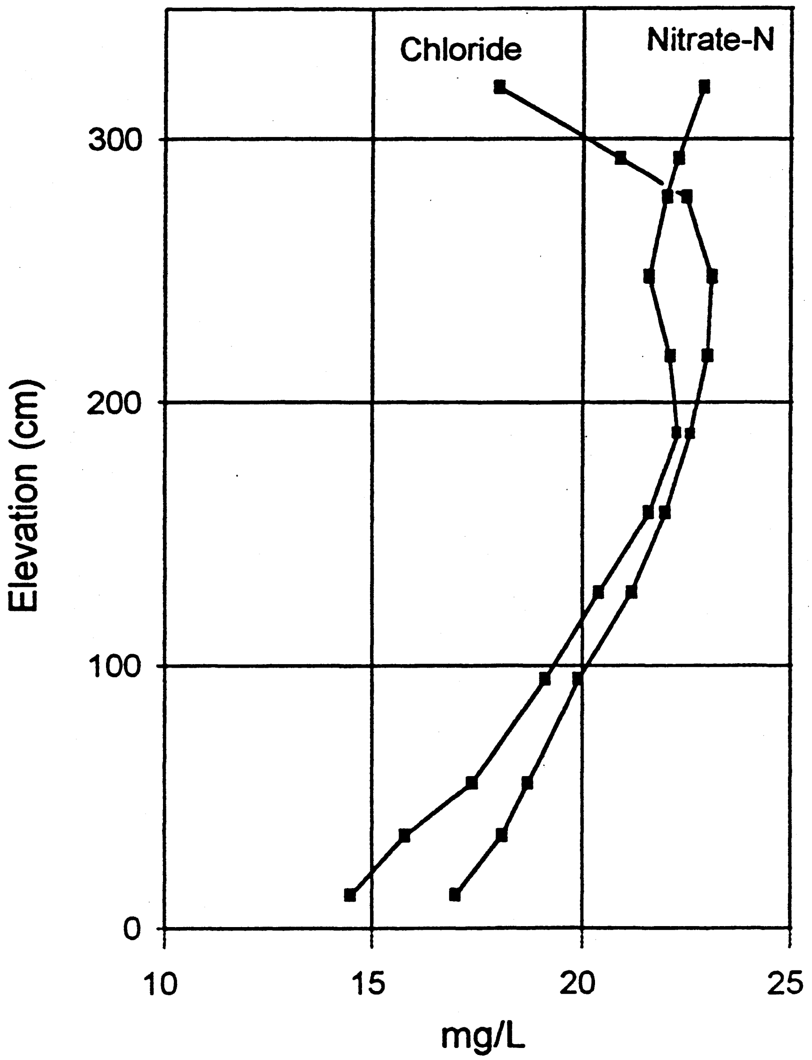


Figure 3-4. Profiles of time-weighted average NO_3-N and Cl concentrations. Averages for each port (from the six in-field MLPs) were combined into a mean to represent the concentration at that relative elevation. Only ports that were sampled at least 10 times are included.

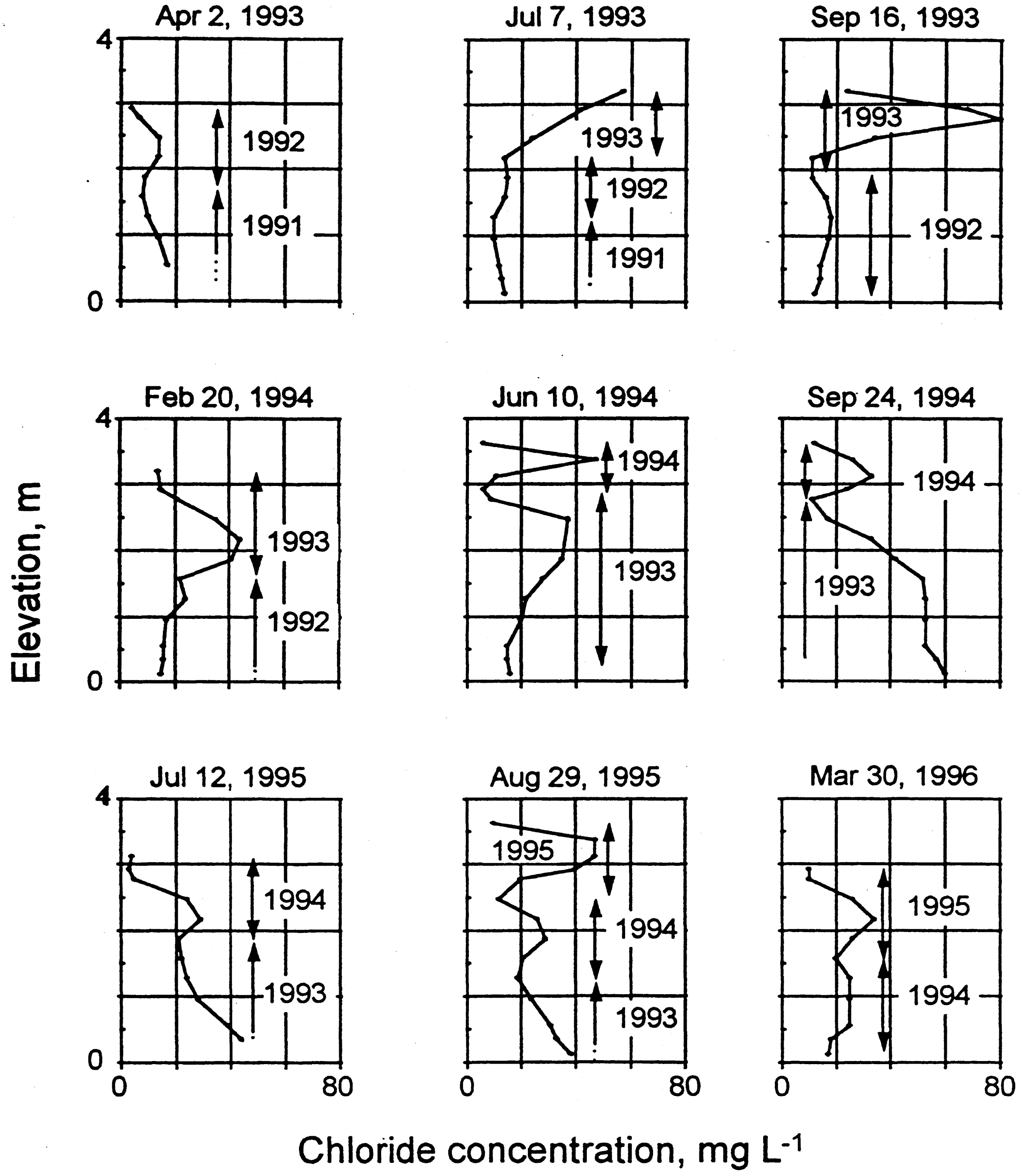


Figure 3-5. Chloride concentration profiles in MLP 6, showing downward movement of pulses. Elevations are relative to the bottom of the sampled zone. Arrows indicate bands of annual recharge.

Chloride and NO_3-N concentrations at each relative elevation, averaged over the six in-field MLPs and by quarter-year, are shown for 1994-1996 in Figures 3-6 and 3-7. The pattern of downward solute movement with time is apparent in the figures. The most important features in a Cl profile are the concentration minima that demarcate annual pulses, because annual pulses can be used to measure the annual loading of solutes to groundwater (see Chapter 4). In the first profile (1994, quarter 2), a Cl minimum was present 3 m above the base datum. By the next quarter (1994, quarter 3), it moved down to about 2.4 m elevation, and in the next two quarters it is near 2.0 m; little vertical movement occurred during winter. The same minimum may have been present a bit below 2 m elevation in the second quarter of 1995, but then it faded out. A new minimum was appearing at an elevation of at least 3.5 m in the third quarter of 1995, and it moved down to about 3.2 m, then 3.0 m, in the subsequent two quarters.

Maxima and minima in the quarterly NO_3 profiles are generally harder to determine (Figure 3-7). This can be attributed to the same processes that make NO_3 less temporally variable than Cl: multiple annual inputs, less variability in annual application rates, and more biological retention.

Individual MLPs' NO_3-N and Cl concentrations through the period of record are presented as contour plots in time and depth in Figures 3-8 and 3-9. Downward movement of solute pulses can be seen as dark regions extending downward and to the right.

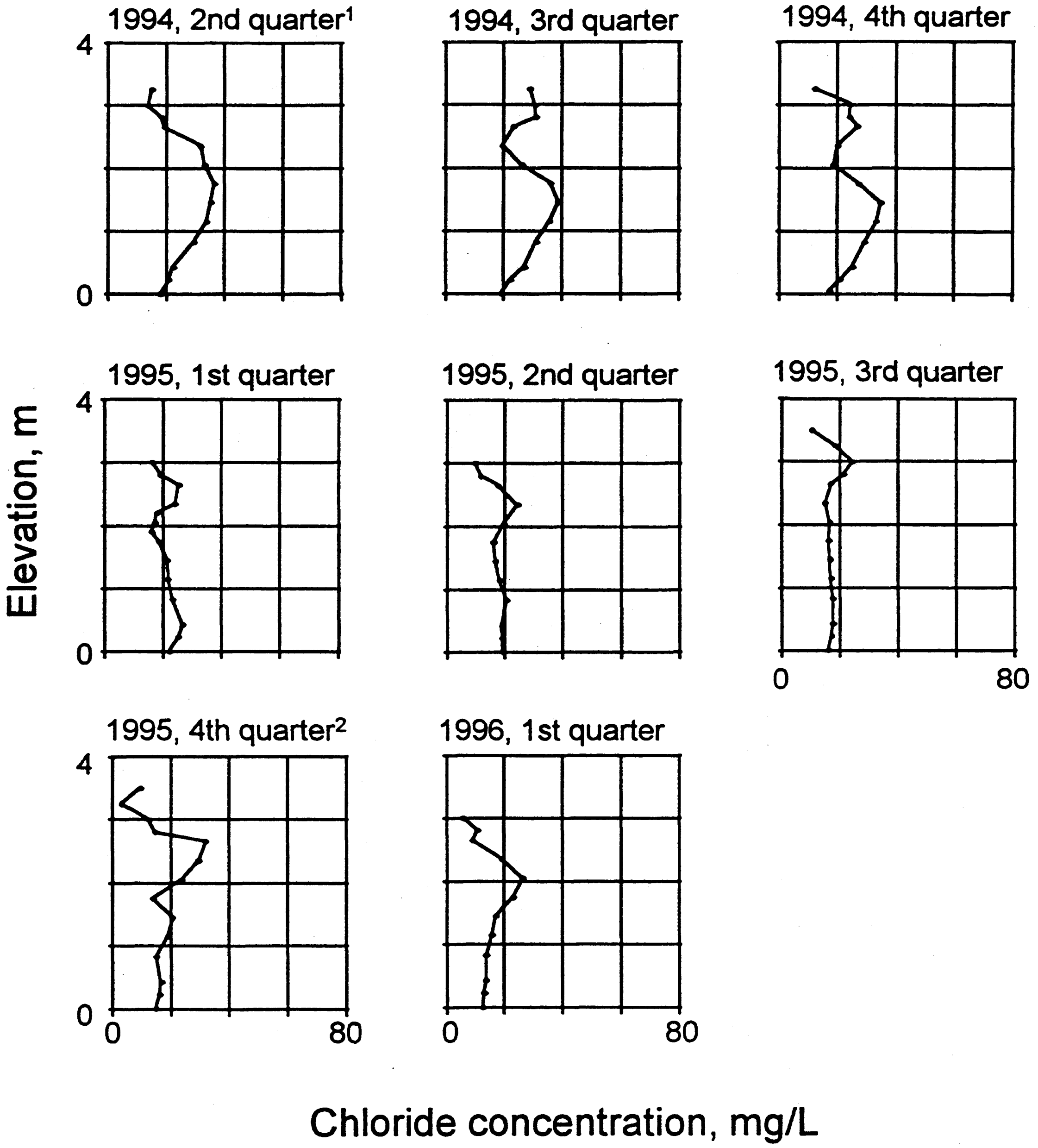


Figure 3-6. Chloride concentration profiles. The elevation datum was 3 m below the water table at the time the MLPs were installed. Grid divisions are 1 meter.
[1] Includes one late-March sampling
[2] Only one sampling date in this quarter

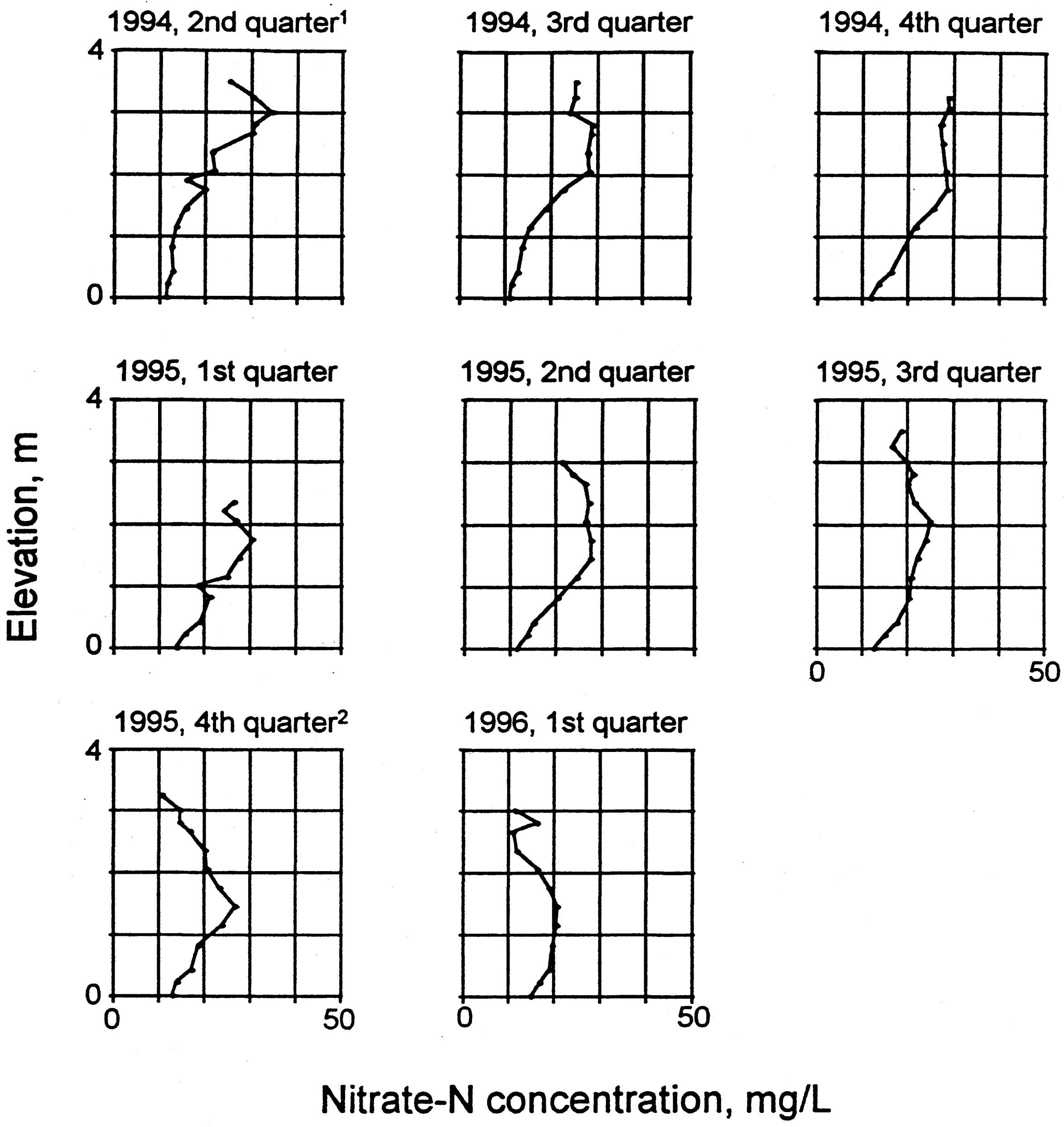


Figure 3-7. Nitrate-N concentration profiles. The elevation datum was 3 m below the water table at the time the MLPs were installed. Grid divisions are 1 meter.
[1]Includes late March sampling
[2]Only one sampling date in this quarter

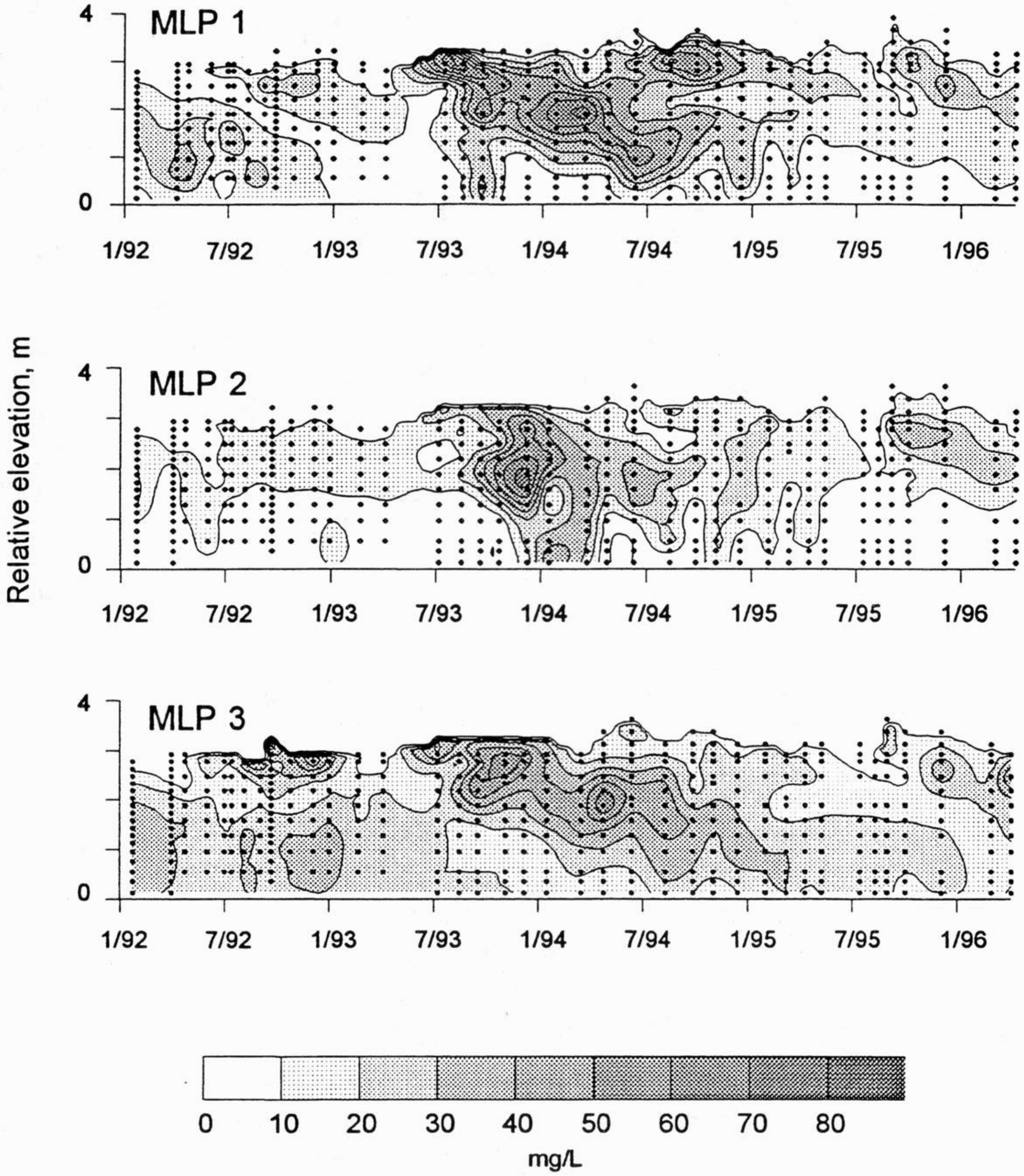


Figure 3-8. Contour plots of Cl concentration in depth and time for MLPs 1-3. Sample depths are indicated with dots.

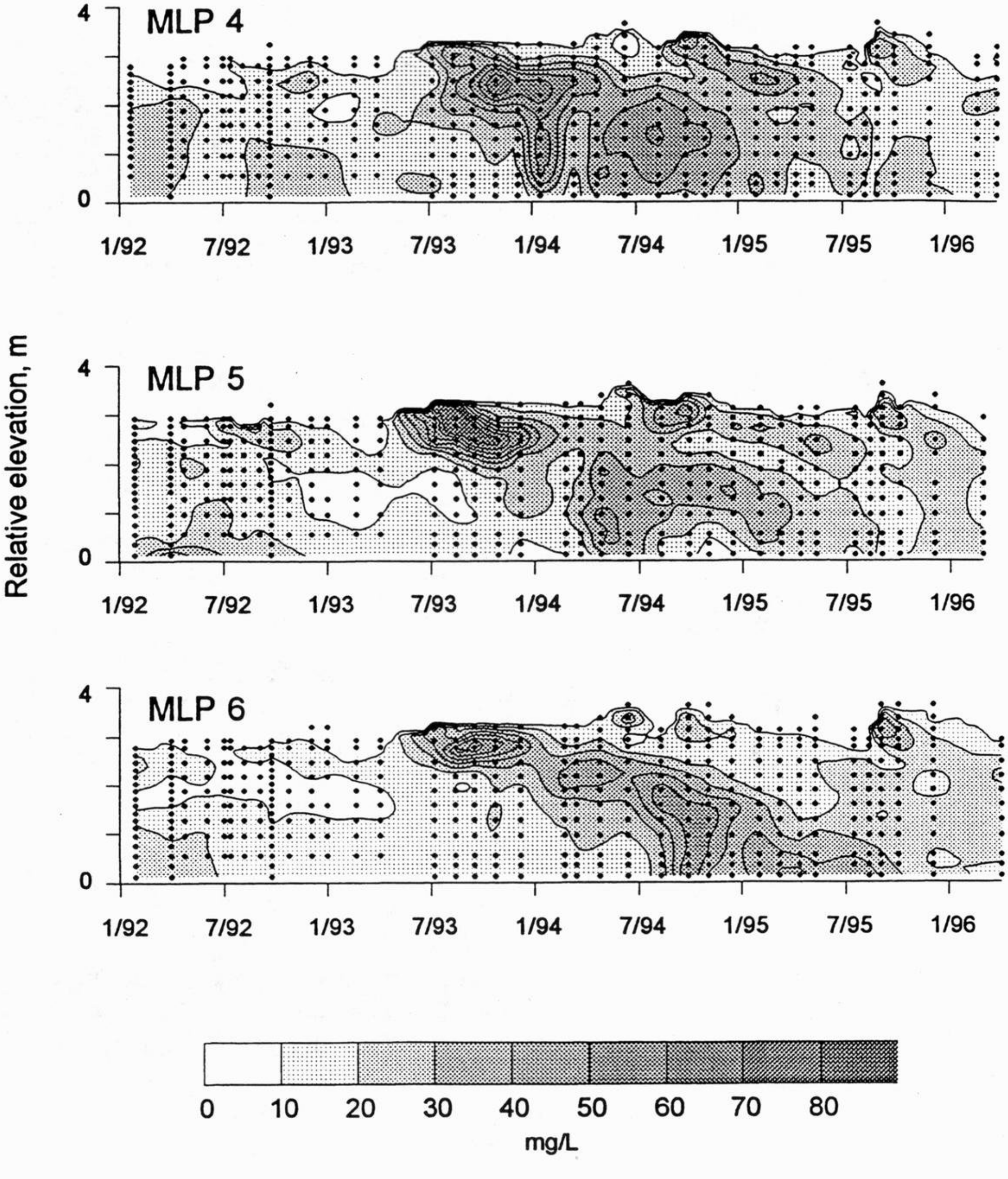


Figure 3-8, continued. Contour plots of Cl concentration in depth and time for MLPs 4-6. Sample depths are indicated with dots.

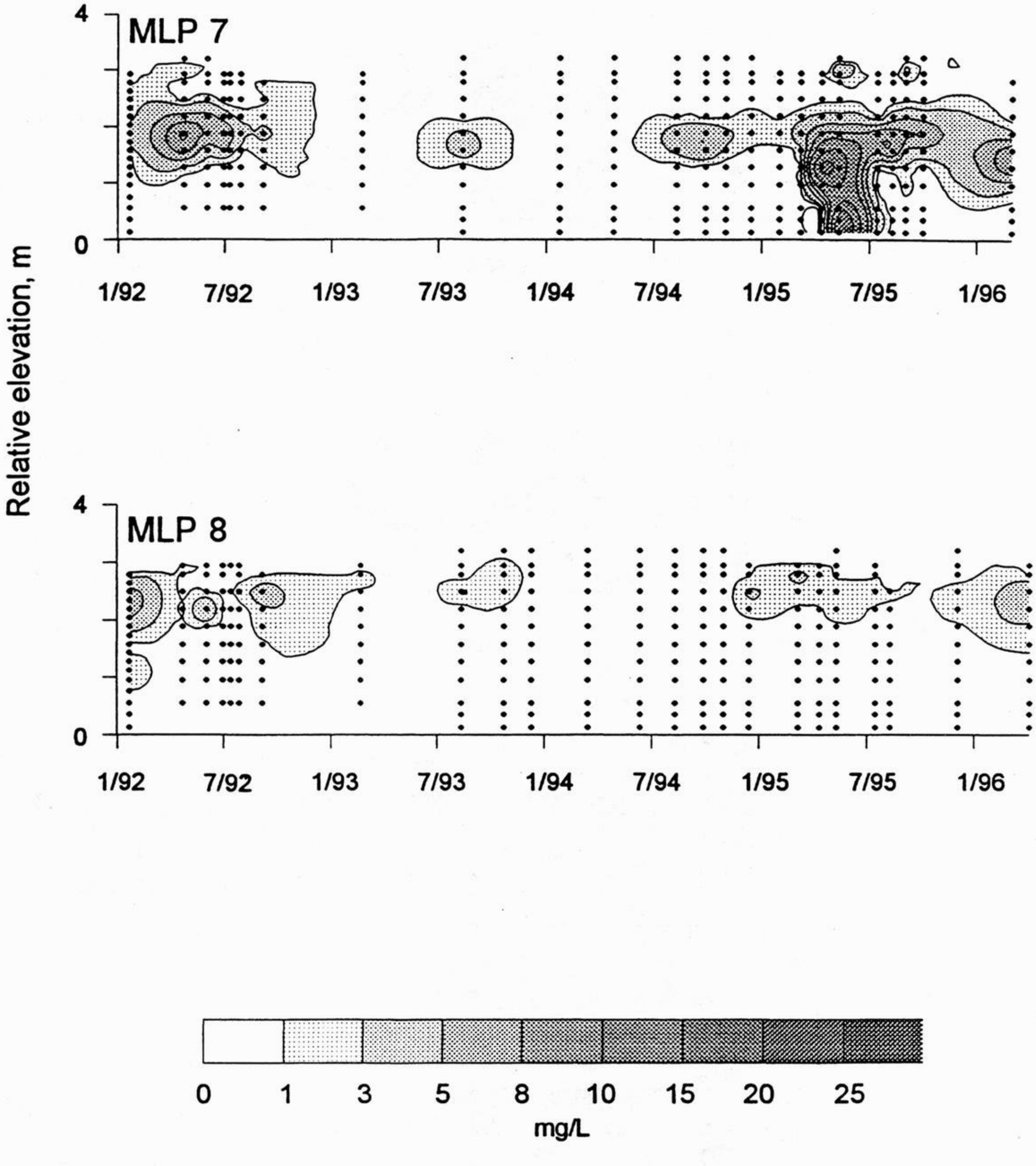


Figure 3-8, continued. Contour plots of Cl concentration in depth and time for MLPs 7-8. Sample depths are indicated with dots. Scale differs from previous pages.

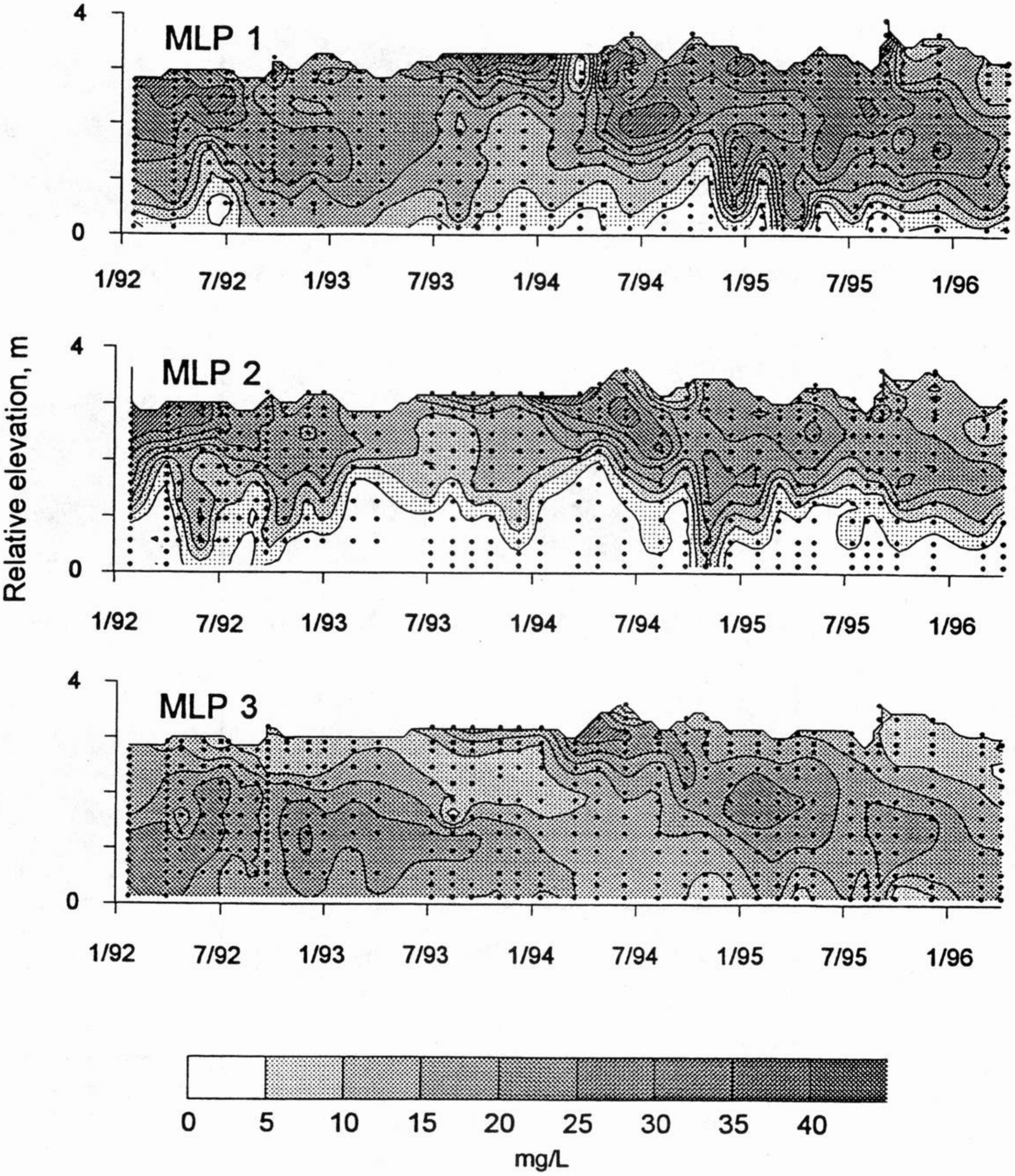


Figure 3-9. Contour plots of NO_3-N concentration in depth and time for MLPs 1-3. Sample depths are indicated with dots.

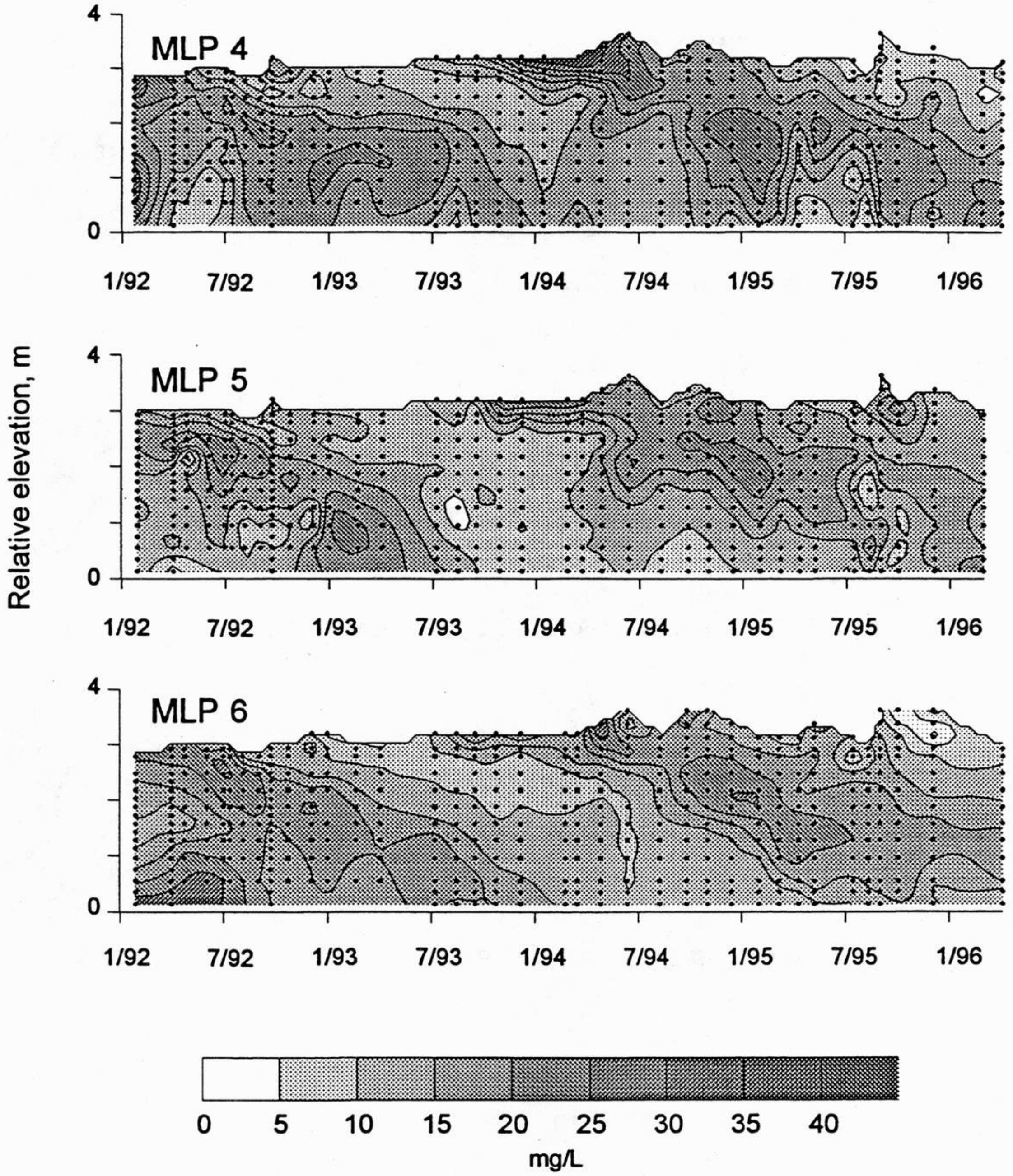


Figure 3-9, continued. Contour plots of NO_3-N concentration in depth and time for MLPs 1-8. Sample depths are indicated with dots.

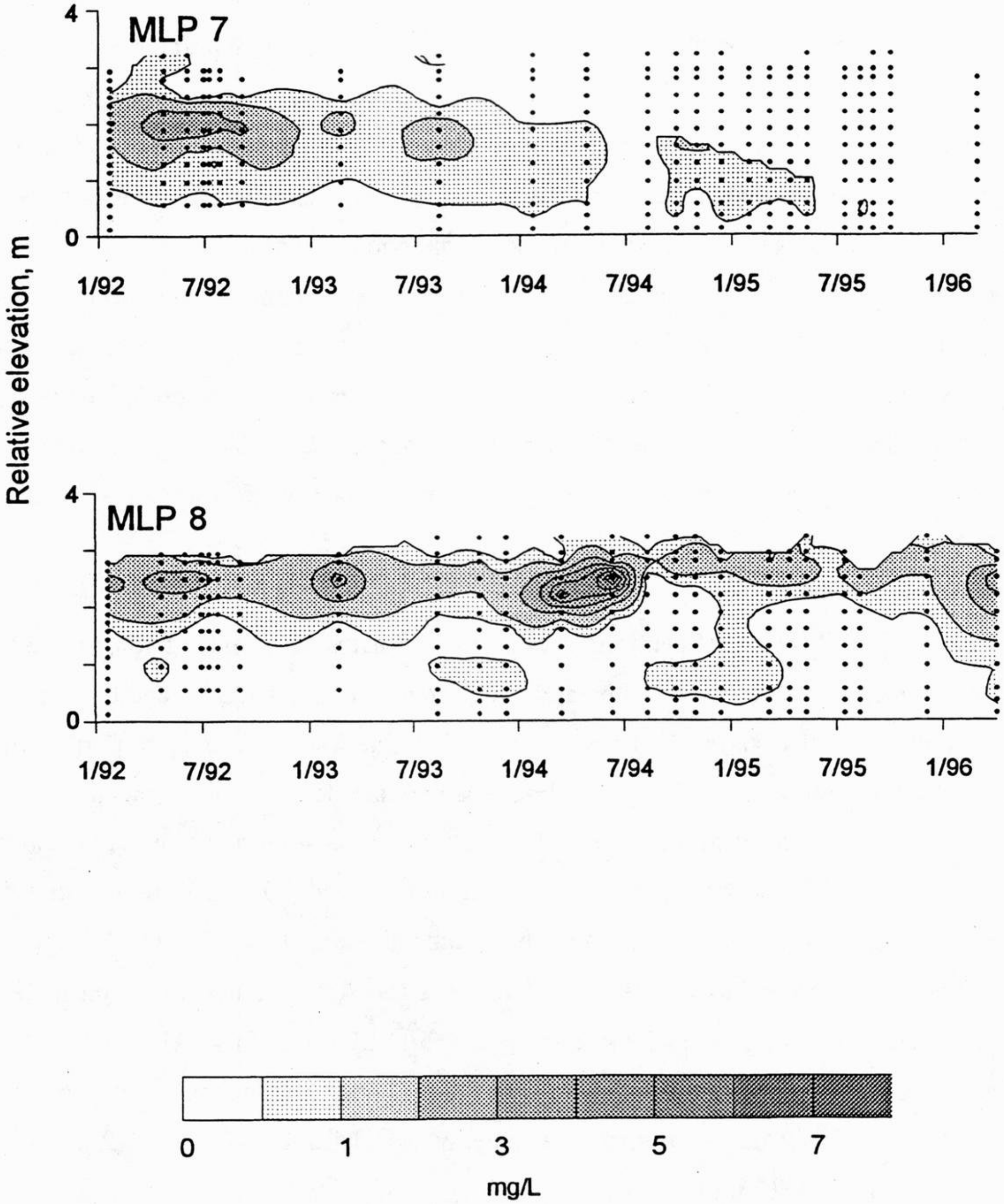


Figure 3-9, continued. Contour plots of NO_3-N concentration in depth and time for MLPs 1-8. Sample depths are indicated with dots. Scale differs from previous pages.

Pesticides in groundwater

The study field has been sampled for pesticides three times by Kraft et al. (1995) and once during the current study. In the current study, we sampled 19 ports in MLPs 3-6 in November 1995, and detected pesticide residues in all but one. We identified the same pesticide residues as Kraft et al., except no alachlor: atrazine, desethylatrazine, desisopropylatrazine, carbofuran, metolachlor, and metribuzin. Other pesticide residues may have been present, but are not detectable by our analytical method or those of Kraft et al. For example, the alachlor breakdown product alachlor ESA is found more commonly in groundwater than its parent compound (Kolpin et al., 1996). Since alachlor was previously detected under this field, alachlor ESA may well have been present. Other pesticides used on the field but not detectable by our analytical method include methyl parathion, methamidophos, and chlorothalonil. These pesticides have been detected in other groundwater studies (Williams et al., 1987; McLean et al., 1988).

Table 3-8 shows concentrations of pesticides that were found in two or more MLPs in November 1995. Atrazine and its breakdown products desethylatrazine and desisopropylatrazine were found in 17 of the 19 sampled ports. Atrazine was detected in 16 ports with a mean detection concentration of 0.2 $\mu g\ L^{-1}$, desethylatrazine in 15 ports with a mean of 0.3 $\mu g\ L^{-1}$, and desisopropylatrazine in only one port at 0.1 $\mu g\ L^{-1}$. Total atrazine-residue (atrazine + desethylatrazine + desisopropylatrazine) detections averaged 0.6 $\mu g\ L^{-1}$, double its Wisconsin preventive action limit (PAL), and had a maximum value of 1.1 $\mu g\ L^{-1}$. The PAL for atrazine residues was exceeded in 10 ports. Metribuzin was found in 17 ports, with a mean detection concentration of 0.9 $\mu g\ L^{-1}$ and a maximum of 3.2 $\mu g\ L^{-1}$, 6% of its PAL. Metolachlor was detected in 11 ports with a mean detection of 0.2 $\mu g\ L^{-1}$ and a maximum concentration of 0.3 $\mu g\ L^{-1}$, 20% of its PAL. Carbofuran was only found in MLP 6, at 0.5 and 0.2 $\mu g\ L^{-1}$, much less than its PAL of 10 $\mu g\ L^{-1}$.

We found lower pesticide concentrations and fewer PAL exceedances in November 1995 than did Kraft et al. (1995) in 1993. In November 1993, pesticides were sampled extensively enough in Field 2 to provide comparisons for the 1995 data. In 1993, atrazine exceeded its PAL in all the ports that were resampled in 1995, when there was a PAL exceedance in only one port.

Table 3-8. Concentrations (μg L^{-1}) of pesticide residues found in two or more MLPs in November, 1995, with standards for comparison. Nondetects are shown with dashes.

Standards		Atrazine	Desethylatrazine	Sum[1]	Metolachlor	Metribuzin
US MCL		3.0				
Wis. ES				3.0	15	250
Wis. PAL				0.3	1.5	50

MLP	Elevation	Atrazine	Desethylatrazine	Sum[1]	Metolachlor	Metribuzin
3	338	0.3	0.3	0.6	0.2	0.1
	188	0.1	0.1	0.2	0.2	0.6
	95	0.1	–	0.1	–	0.1
	12	0.2	0.2	0.5[2]	–	0.1
4	338	–	–	–	–	–
	278	0.1	0.1	0.2	–	0.1
	188	0.2	0.5	0.7	0.2	0.8
	95	–	0.2	0.2	0.1	1.0
	12	0.1	0.2	0.3	0.1	–
East-½ no. of detections		7 (of 9)	7	8	5	7
mean of detections		0.2	0.2	0.4	0.2	0.4
5	338	0.2	0.4	0.6	–	–
	278	0.3	0.3	0.6	0.1	0.4
	188	0.2	0.2	0.4	0.1	2.3
	95	–	–	–	–	0.7
	12	0.2	0.2	0.4	0.1	0.2
6	338	0.1	0.1	0.2	–	0.2
	278	0.1	–	0.1	–	0.2
	188	0.3	0.4	0.7	0.1	2.1
	95	0.3	0.6	0.9	0.2	3.2
	12	0.4	0.7	1.1	0.3	1.9
West-½ no. of detections		9 (of 10)	8	9	6	9
mean of detections		0.2	0.4	0.6	0.2	1.2
Mean for field		0.2	0.3	0.6	0.2	0.9

[1] Sum of atrazine residues

[2] Includes 0.1 μg L^{-1} of desisopropylatrazine

Detections of total atrazine residues, metolachlor, and metribuzin, averaged by MLP, were greater in 1993 than 1995 in nine out of 12 cases (Table 3-9).

Table 3-9. Average pesticide detections in the 1993 and 1995 samplings. Atrazine, desethylatrazine, and desisopropylatrazine are summed.

		MLP			
Compound	Year	3	4	5	6
Sum of atrazine residues	1993	1.0	0.8	1.0	0.5
	1995	0.4	0.4	0.5	0.6
Metolachlor	1993	0.9	0.5	n.d.	0.5
	1995	0.2	0.1	0.1	0.2
Metribuzin	1993	1.8	1.7	0.8	1.6
	1995	0.2	0.6	0.9	1.5

Pesticides did not generally vary systematically with depth. The depth distribution of atrazine concentrations (both species and their sum) was fairly even, except in MLP 6 where concentrations increased markedly with depth. Metolachlor also increased slightly with depth in MLP 6. Analysis of variance was used to test for effects of depth on concentration of each pesticide. A natural-log transformation was applied, because the concentration distributions were skewed to the right. The only significant difference between depths was for metribuzin, with greater concentrations at 95- and 188-cm elevations.

West field-half pesticide-residue detections (MLPs 5 and 6) were usually greater than those on the east, but there were too few observations for statistical tests. Average summed atrazine-residue detections were 0.6 $\mu g\ L^{-1}$ on the west field-half and 0.4 on the east. Recent atrazine applications were also slightly greater on the west (Table 3-10), which may have contributed to the difference in average detections.

Table 3-10. Pesticide applications to Field 2, 1991-1995. Amounts are kg ha^{-1}.

Year	Material	West-side rate	East-side rate
1992	Atrazine	0.84	0.84
	Alachlor	0	1.68
	Metolachlor	1.68	0
1993	Metolachlor	1.68	1.68
	Metribuzin	0.56	0.56
1994	Atrazine	1.12	0.84
	Alachlor	1.68	0
	Metolachlor	0	1.68
1995	Atrazine	1.12	1.12
	Alachlor	1.68	1.68

Metribuzin concentrations, like atrazine, were greater on the west field-half; concentrations strictly increased from east to west. Metribuzin was last applied to this field in 1993 at a uniform 1.1 kg ha^{-1}, so differences in treatment do not explain the difference in detections. There may have been a gradient of some property that affects the movement or persistence of metribuzin. For example, water-table depth increases from west to east. Perhaps metribuzin on the west side had reached the groundwater and passed out of our sampled volume, while on the east side it was still in the system. The presence of up to 3.2 μg L^{-1} of metribuzin more than two years after the last application (comparable to the largest concentration found in this field in 1993, 5.7 μg L^{-1}) demonstrated the persistence of this compound.

Metolachlor concentrations did not exhibit a spatial pattern. Average metolachlor detections were equal on the two field halves, 0.11 μg L^{-1}, even though the material was applied only to the east half in 1994 (Table 3-10). None was applied in 1995, and the application was uniform in 1993. The 1993 application, then, persisted with concentrations up to 0.4 μg L^{-1} in late 1995. In 1992, metolachlor was applied only to the west half, also at 1.68 kg ha^{-1}. It is possible enough 1992 metolachlor remained to balance the 1994 application, producing equal detections on both sides of the field.

Atrazine has persisted in this system for more than one year after its application. Kraft et al. (1995) found a Field 2 peak concentration of 1.9 µg L^{-1} atrazine species in November 1993, atrazine having been applied early in the 1992 growing season.

Pesticide concentrations in this field were generally well below Wisconsin ESs and PALs, except for atrazine. The sum of atrazine residues exceeded the PAL in 10 of 19 samples, but did not exceed the ES. Of the other detected pesticides, the closest approach to any standard was where metolachlor reached 20% of its PAL in one port. Concentrations of atrazine residues and metribuzin were usually greater in the west field-half than the east. There is not a long-term bias toward greater concentrations in the west field-half; Kraft et al. (1995) found greater concentrations in the east side of Field 2 for atrazine residues, metribuzin, and metolachlor. The cross-field difference in atrazine residues could be related to different application rates, but a similar metribuzin difference is not explainable by application rates.

Conclusions

This study extended monitoring of Field 2 and immediately upgradient MLPs to just over four years. Analyte values varied with time, depth, and location in the field. Nitrate-N concentrations ranged from <0.2 to 50.5 mg L^{-1}, and Cl from <1 to 119 mg L^{-1}. The pH range was 3.72 to 7.27, and specific conductance ranged from 2 to 75 mS m^{-1}. Residues of four pesticides were identified. Solute concentrations were much greater under the field than in upgradient MLPs.

The mean concentration of NO_3-N in the monitored groundwater zone (approximately the uppermost 3 m of groundwater) beneath the field was 20.5 mg L^{-1} during the period of record. Averages for other analytes were Cl, 20.2 mg L^{-1}; specific conductance, 33.2 mS m^{-1}; and pH, 4.90. Upgradient from the field, the average NO_3-N concentration was 0.7 mg L^{-1}, and Cl was 1.4 mg L^{-1}; specific conductance was 6 mS m^{-1}. This demonstrates that virtually all of the solutes in groundwater below the field originated from farming practices. Pulses of chloride and NO_3 moved downward in the saturated zone beneath the field, and were identifiable for months to years.

Fertilizer N applications were slightly below the UW-Extension recommendation in 1995, and not greatly in excess of the recommendation in 1994. Despite the grower's efforts to reduce N input, the upper groundwater below the field contained twice the enforcement-standard concentration of NO_3-N.

Depth-averaged Cl concentrations in the monitored zone beneath the field varied substantially with time. A particularly heavy 1993 application of KCl fertilizer boosted Cl for one year and contributed to this variability. Nitrate concentrations did not show such large time variations, even though there were large differences in annual application rates.

Nitrate and Cl concentrations within the field apparently decreased with depth, but only the NO_3 trend was statistically significant. This decrease was most likely due to vertical mixing of agriculture-affected shallow groundwater with low-solute water from upgradient. One MLP close to the upgradient edge of the field had unusually low solute concentrations in the lower ports. Apparently, underflow from outside the agriculture-affected area was being sampled.

Pesticide concentrations in the current study were generally less than in the previous study (Kraft et al., 1995) and, except for atrazine, were much less than their PALs or ESs. Average atrazine-residue concentrations in MLPs often exceeded the PAL. Atrazine and metribuzin detections were usually greater on the west half of the field than the east half. The difference in atrazine corresponded with a small difference in atrazine application rates to the two sides of the field. Metribuzin concentrations did not reflect application rates, but increased from east to west. The trend is probably related to a gradient in some environmental property such as water-table depth.

Average metolachlor detections were the same on the two field halves, even though none was applied in 1995, and in 1994 metolachlor was applied only to the east half. The last time metolachlor was applied to the west half was 1992.

Chapter 4
Nitrogen loading to groundwater in an irrigated vegetable field

Nitrate loading to groundwater for the 1994 and 1995 sweet corn crop in Field 2 (Figure 3-1) was estimated using a nitrogen budget, and a method employing available groundwater monitoring data. The budget method estimates NO_3-N loading as the difference between N inputs and outputs. The method utilizing monitoring data ("water-year method") estimates NO_3 loading from concentrations observed in a given year's groundwater recharge. Both methods were used by Kraft et al. (1995) to estimate NO_3 loading from Field 2 in 1992 and 1993.

Nitrogen budget

Theory

We calculated an N budget for the study field for 1994 and 1995, using an approach similar to that of Meisinger and Randall (1991). The budget can be expressed as

$$N_{pl} = N_{input} - N_{output} - \Delta N.$$

N_{input} and N_{output} are the N components (other than leaching) that enter and exit the field within the crop canopy and soil root zone. ΔN is the change in stored N, and N_{pl} is N potentially available for leaching to groundwater. We use N_{pl} as an estimate for NO_3-N loading to groundwater, which is justified because NO_3 is the only N form that leaches significantly in this agricultural system.

A steady state ($\Delta N = 0$) is assumed with respect to inorganic N, that is, the inorganic N content is the same on the first and last days of the budget year. In addition, for accounting purposes, we treat the mineralization of soil organic matter as an input. As a result, the N budget becomes

$$N_{pl} = N_{input} - N_{output}.$$

Inputs

Nitrogen inputs considered by Meisinger and Randall (1991) include fertilizer, precipitation, dry deposition, soil organic matter mineralization, microbial fixation, irrigation water, and crop seed. We used a subset of these (Table 4-1). Organic matter N mineralization for a Plainfield loamy sand has been measured at 45 kg N ha^{-1} y^{-1} during the growing season by Oberle and Bundy (1987), however, their study did not account for wet nor dry N deposition. We used a mineralization rate of 15 kg N ha^{-1} y^{-1}, which is 45 minus our estimate of 30 (Table 4-1) for wet + dry deposition. N fixation was negligible for the crops and conditions of 1994 and 1995, and was disregarded. Irrigation water analyses indicated this is a minuscule source of nitrogen and can be neglected. Corn seed adds less than 1 kg N ha^{-1} and hence is also negligible.

Table 4-1. Nitrogen inputs to the study field in 1994 and 1995. Values are kg ha^{-1}.

Source	1994	1995	References
Fertilizer	207	176[1]	Grower's records
Precipitation	15	15	Andraski and Bundy (1990)
Dry deposition	15	15	Andraski and Bundy (1990), Schepers and Mosier (1991), Meisinger and Randall (1991)
Mineralization	15	15	Oberle and Keeney (1990); all the above
Total	252	221	

[1] Average of 178 kg ha^{-1} on the west field-half and 174 kg ha^{-1} on the east.

Outputs

N outputs considered by Meisinger and Randall (1991) include harvested crop, ammonia volatilization, denitrification, soil erosion and runoff, and miscellaneous gaseous losses (Table 4-2). Their review indicates a common harvested sweet corn N concentration of 4.3 g

N kg^{-1} (fresh), with a range of 3.8-4.9. The "common" value applies to crops grown under good management and weather conditions. The sweet corn yields of 21 Mg ha^{-1} in 1994 and 20 Mg ha^{-1} in 1995 give harvested N estimates of 91 kg ha^{-1} (range 79 to 102) and 86 kg ha^{-1} (range 76 to 97), respectively.

Table 4-2. Estimated N outputs other than leaching, 1994 and 1995.

N loss category		1994	1995
Miscellaneous gaseous loss, kg ha^{-1}		3	2
Senescent gaseous loss, kg ha^{-1}		4	4
Harvest			
Crop yield, Mg ha^{-1}		21	20
Crop N content, g kg^{-1}	Low	3.78	3.78
	Common[1]	4.32	4.32
	High	4.86	4.86
Harvested N, kg ha^{-1}	Low	79	76
	Common	91	86
	High	102	97
Total output, kg ha^{-1}	Low	86	82
	Common	98	92
	High	109	103

[1] As defined by Meisinger and Randall (1991).

Ammonia volatilization occurs under certain conditions when ammonium-yielding fertilizers are not incorporated. However, the acidity of Plainfield soil apparently limits ammonia volatilization (Saffigna et al., 1977), and in addition, as little as 2.5 mm of moisture within 4 days after urea application virtually eliminates ammonia loss (Oberle and Keeney, 1987). As the study fields were irrigated regularly, ammonia volatilization losses were deemed negligible. Denitrification is also negligible in these well-drained sandy soils (Saffigna et al, 1977). Wind and water erosion losses were not evident, and so such potential losses are neglected.

Miscellaneous gaseous losses include N_2O evolution during nitrification, decomposition of nitrous acid, and reactions of nitrous acid with soil minerals and organic constituents. We used the suggestion by Meisinger and Randall (1991) that these losses be approximated by 1% of the total N inputs.

Estimates for the gaseous loss of plant N due to senescence have been given as 2-8% of total aboveground plant N as ammonia and volatile amines (Meisinger and Randall, 1991). We assumed the midpoint of this range, 5%. The pre-harvest aboveground sweet-corn N was taken to be 180 kg ha^{-1} (Olson and Kurtz, 1982), therefore, senescent N loss would have been approximately 4 kg ha^{-1}. This estimate may be high since sweet corn is harvested before grain maturity. A review of research by Parker (1962) and Wagger et al. (1985) shows post-harvest volatile loss of N from sweet corn residue is probably negligible.

Results

Nitrogen inputs, outputs, and NO_3 loading for the 1994 and 1995 sweet corn crops are shown in Table 4-3. Ranges of NO_3 loading are based on the range of crop N content. The NO_3 loading estimates for 1994 and 1995 are 154 and 129 kg ha^{-1}, respectively.

Table 4-3. Nitrogen-budget inputs, outputs, and NO_3-N loading for a range of crop N content. Values are kg ha^{-1}.

	Crop N content	1994	1995
Total N input		252	221
	Low	86	82
Total N output	Common[1]	98	92
	High	109	103
	Low	143	118
NO_3-N loading	Common[1]	154	129
	High	166	139

[1] As defined by Meisinger and Randall (1991).

Discussion

Budget-derived NO_3-N loading estimates for Field 2 are available in this study for the 1994 and 1995 sweet corn crops, and from Kraft et al. (1995) for the 1992 sweet corn and 1993 potato crops (Table 4-4). The loading for the 1993 potato crop has been revised using new information on potato N concentration. Kraft et al. (1995) previously used a concentration of 0.4% N for fresh harvested potato based on the review contained in Meisinger and Randall (1991). Wisconsin-specific data suggest that 0.24% N is more appropriate (Saffigna et al., 1977; Wilner et al., 1997; Bundy et al., 1997).

Table 4-4. Fertilizer N input, crop yield, and budget-derived NO_3-N loading during the period of record for Field 2. Values are kg ha^{-1}.

	1992	1993	1994	1995	Average
Crop	S. corn	Potato	S. corn	S. corn	
Crop yield	25 000	46 000	21 000	20 000	
			N inputs		
Fertilizer	250	297 / 357	207	176	240[1]
Atmosphere, soil	45	45	45	45	45
Seed	0	9	0	0	2
Total N input	295	351 / 411	252	221	288[1]
			N outputs		
Harvest	106	110	91	86	98
Other	7	10	7	6	8
Total N output	113	120	98	92	106
			NO_3-N loading		
	182	231 / 291	154	129	182[1]
As % of fertilizer N	73	78 / 82	74	73	76
As % of total N	62	66 / 71	61	58	63

[1] A single mean value was used for 1993 so as not to count the year twice.

The budget-derived annual NO_3-N loading for the three years of sweet corn during the period of record varied from 129 to 182 kg ha^{-1} (Table 4-4). Sweet-corn loading increased strictly with N fertilizer amount. The loading was lowest with sweet corn and a fertilizer N input close to the University of Wisconsin – Extension recommendation (Table 4-5) of 179 kg ha^{-1} (Kelling et al., 1991). Yet NO_3-N loading was still 129 kg ha^{-1} that year.

Table 4-5. Comparison of budget-derived NO_3-N loading with the excess of fertilizer over UW–Extension recommendations (Kelling et al., 1991). Values are kg ha^{-1}.

	1992	1993	1994	1995	Avg.
Crop	S. corn	Potato	S. corn	S. corn	
Fertilizer N	250	297 / 357	207	176	251[1]
UWEX recommendation	179	258	179	179	199
Excess	71	39 / 99	28	-3	41[1]
NO_3-N loading	182	231 / 291	154	129	182[1]

[1] A single mean value was used for 1993 so as not to count the year twice.

The 1993 potato NO_3-N loading was 231 kg ha^{-1} for the north half of the field, and 291 in the south, reflecting differing fertilization rates (Table 4-4). These NO_3 loading amounts were greater than any sweet corn loading, even though the fertilizer N applied to potato in excess of UW–Extension recommendations was less than the sweet-corn excess, in some cases.

Water year method

The water-year method estimated NO_3 loading from analyses of groundwater collected at the six monitoring locations in Field 2 (Chapter 3). The crux of the method is to use the analyses to identify the upper and lower bounds of a given year's recharge in water quality profiles, and then to calculate the NO_3 mass between those bounds.

Methods

Identifying a year's recharge water in the saturated zone is possible if an annually-applied tracer is available to mark the beginning and end of the recharge year ("water year"). In this study, Cl originating mainly from KCl fertilizer applied annually in March or April met this need. Early KCl applications accounted for more than 85% of the Cl applied to the field, and far exceeded background Cl inputs. Since this application occurred well before crop root development, much of the Cl leached to groundwater. The resulting Cl pulse could be observed to grow and pass through a monitored interval in the saturated zone, with concentration minima marking boundaries between successive water years (Figure 3-5).

The mass of a solute loaded to groundwater during a water year can be estimated once the upper and lower boundaries of the water year are delineated in concentration-depth profiles. The mass of solute per area of field can be calculated by integrating the product of concentration and porosity over the thickness of the solute plume (Kraft et al., 1995), which is expressed as

$$m = \int_a^b \theta C dz$$

where m = mass of solute leached during the water year per unit area of field
a = elevation of the top of the water-year band
b = elevation of the bottom of the water-year band
θ = aquifer porosity
C = concentration
z = elevation.

This approach is valid so long as the water-year profile contains recharge originating only from the study field. Such is the case when monitoring locations are at least one year's groundwater travel distance from upgradient field edges. At monitoring locations closer to an upgradient field edge, water-year profiles would contain recharge water from both the field and the upgradient land use, precluding an accurate estimate of NO_3-N loading from the field.

In this study, most of the Cl in water-year profiles would be expected to have originated at the soil surface during the same water year. This is due to Cl being applied early in the year and being highly leachable from soil and crop residues. Chloride concentration profiles over time (Figure 3-5) demonstrate clear breakthrough curves, supporting this expectation. The situation is more complex with NO_3, because N is more subject to biocycling and bioretention. The NO_3 loaded to groundwater during a given water year originated from that year's fertilizer, soil organic matter mineralization, and rainfall, and likely also from the previous year's crop residue. Hence, crop residue might buffer the system from one year to the next. The result is that the NO_3-N loading determined for a given water year may not tell specifically how much NO_3 resulted solely from the previous year's practices. The water-year method should perform well for evaluating loading over a period of several years, because in time, such a buffering effect would average out.

Loading estimates were calculated for sampling dates on which complete water-year profiles were identifiable in the saturated zone (Appendix 4, Table A4-1). Profiles that were clearly bounded by Cl concentration minima were considered complete. Profiles that preceded the appearance of a local minimum at the top, or whose lower-boundary minimum had already moved past the bottom of the sampled zone, were also included if they contained the same breakthrough features as preceding or subsequent complete profiles.

MLP 1 (Table A4-1, page 1) provides an example of how water years were identified, and how data from some sampling dates were selected for estimating loading. The 13 March 1994 profile contained the first evidence that 1994 recharge had begun to reach the water table. On that date, a local Cl minimum at $z = 277.5$ cm (z = elevation) marked the boundary between 1994 water (above) and 1993 water (below). At later dates, the boundary moved downward in the profile, while the 1994 Cl pulse continued its breakthrough at the water table. Though the boundary between water years blurred over time due to dispersion, it persisted as a local Cl minimum. The beginning of the 1995 Cl breakthrough at MLP 1 was evident in July 1995. Comparing profiles before and after the 1995 breakthrough showed that the 1994 profile was essentially complete by April 1995 and remained complete through the July sampling. The three sampling dates from April through July, then, were used to estimate Cl and NO_3 loading in the 1994 water year. After the 12 July sampling, the bottom of the Cl pulse marking the 1994 water year moved below the sampled zone, so later dates are not usable for estimating 1994 solute loading.

The procedure at MLPs 3-6 was as described for MLP 1. Interpreting breakthrough curves was difficult at MLP 2, because of the mixing of underflow water with field recharge in the lower part of the sampled zone. Elevations of water-year boundaries at the other MLPs on corresponding dates were used to help infer boundaries.

Results

Chloride loading estimates for the six MLPs in Field 2 (Table 4-6) averaged 116 kg ha^{-1} (95% confidence interval, 98-133 kg ha^{-1}) in the 1994 water year, and in 1995, 138 kg ha^{-1} (95% c.i. 108-169). The average loadings were close to the total Cl input: 93% of the fertilizer application in 1994 and 102% in 1995. (The next-largest input, precipitation, was probably < 1 kg ha^{-1} y^{-1}; see Berner and Berner, 1987). Some Cl must have been exported with the crop. Therefore, the Cl loading to groundwater should be somewhat less than Cl inputs. If the exported amount was significant, the water-year Cl loadings are too large. Crop Cl data to evaluate this issue are not available. Agreement between Cl input and loading out-put suggests (but does not prove) that (1) fertilizer Cl largely leaches to groundwater, and (2) the water-year method accurately measures solute loading to groundwater.

The average NO_3-N loading among the six MLPs was 171 kg ha^{-1} (95% c.i. 109-233) in 1994 and 132 kg ha^{-1} (95% c.i. 97-168) in 1995 (Table 4-6).

Spatial variability was probably the principal cause of variation among MLPs in water-year loading determinations. Variability in chemical application rates, uneven redistribution after application, or uneven recharge could cause differences in profiles measured at different times.

Table 4-6. Chloride and NO_3-N loading in 1994 and 1995 at the six in-field MLPs. Chloride is also shown as the percentage of fertilizer Cl recovered.

								95% Confidence bounds	
MLP:	1	2	3	4	5	6	Mean	Lower	Upper
					1994				
Mean NO_3-N loading	285	181	148	142	122	148	171	109	233
Mean Cl loading	140	99	99	128	112	115	116	98	133
Percent of Cl input	112	79	79	103	89	92	93	79	106
					1995				
Mean NO_3-N loading	142	184	115	153	109	91	132	97	168
Mean Cl loading	121	159	186	121	108	135	138	108	169
Percent of Cl input	89	117	137	89	80	99	102	79	124

Nitrate loading varied more among MLPs than Cl loading, as shown by wider confidence intervals (Table 4-6). This may result from the added sources of variability for NO_3, such as variations in soil organic matter concentration and mineralization rates.

1992-1995 water year results

Water-year calculations of annual NO_3-N loading for 1992-1995 ranged from 132 to 178 kg ha^{-1} for the three years of sweet corn, and was 195 or 271 (average 233) kg ha^{-1} for the single year of potato (Table 4-7). Averages of 65% of the total N input and 72% of fertilizer N appeared as NO_3 in groundwater. Nitrate loading increased with the fertilizer input, and if the two crops are considered separately, NO_3 loading also increased with the amount of fertilizer in excess of UW–Extension recommendations (Table 4-8).

Table 4-7. Comparison of fertilizer and total N inputs to water-year NO$_3$-N loading. Values are kg ha^{-1}.

	1992	1993	1994	1995	Avg.[1]
Crop	S. corn	Potato	S. corn	S. corn	
Crop yield	25 000	46 000	21 000	20 000	
Fertilizer N input	250	297 / 357	207	176	240
Total N input	295	357 / 411	252	221	288
Water-year NO$_3$-N loading	178	195 / 271	171	132	178
Loading, % of fertilizer N	71	56 / 66	83	75	74
Loading, % of total N	60	66 / 76	68	60	62

[1] A single mean value was used for 1993 so as not to count the year twice.

Table 4-8. Comparison of water-year NO$_3$-N loading with the excess of fertilizer over UW–Extension recommendations (Kelling et al., 1991). Values are kg ha^{-1}.

	1992	1993	1994	1995	Avg.
Crop	S. corn	Potato	S. corn	S. corn	
Fertilizer N	250	297 / 357	207	176	251[1]
UWEX recommendation	179	258	179	179	199
Excess	71	39 / 99	28	-3	41[1]
Water-year N loading	178	195 / 271	171	132	178[1]

[1] A single mean value was used for 1993 so as not to count the year twice.

Discussion

Budget and water-year NO$_3$ loading estimates compared well for the 1992-1995 growing seasons (Table 4-9). The good agreement between the two methods supports the utility of both. It was noted that the NO$_3$ loading from the water-year method may not reflect only that year's management practices, because of buffering by crop-residue N across years. The consistency of the two methods suggests the buffering effect was small.

The main advantage of the budget method is that many of the data can be obtained from the literature and growers' records, so monitoring data are not needed. But the budget method requires field validation for a wide range of crop and field conditions under which it might be employed. The water-year method requires substantial data collection, but is based on concrete measurements for a specific crop and field. Further verification is needed, but case-by-case validation is not required.

Table 4-9. Comparison of water-year estimates of NO_3-N loading to N-budget N_{pl} (both kg ha^{-1}) in Field 2 for the period of record. The N budget was chosen as the basis for calculating discrepancies.

	1992	1993	1994	1995	Mean
N budget	182	261	154	129	182
Water year	178	233	171	132	178
Discrepancy	-2%	-11%	11%	2%	-2%

Conclusions

Two methods were used to estimate NO_3 loading to groundwater in the study field. An N-budget method estimated NO_3-N loading as the difference between N inputs and outputs. The water-year method estimated NO_3 loading from groundwater monitoring data.

The only field data required by the N budget are crop yields (measurement of crop N concentration would also be helpful), but it requires validation for particular crops and field conditions. The water-year method depends on intensive monitoring, but assumes fewer parameter values, and thus does not depend on extensive validation.

The two methods agreed within 11% each year, and for the four-year period of record they agreed to 2%. In addition to the agreement between the two methods on NO_3 loading, the water-year Cl loading agreed well with the Cl applied in fertilizer. But if a significant amount of Cl was exported in the crop, the agreement may be spurious. More information on crop Cl concentrations is needed to decide.

Variation among MLPs in the field was considerable, and is a potential problem for the water-year method. The 95% confidence interval for Cl loading spanned ±18 kg ha^{-1} in 1994, or 15% of the mean, and ±30 kg ha^{-1} in 1995 (28%). The widths of NO_3-N confidence intervals were ±62 kg ha^{-1} in 1994 (36%) and ±36 kg ha^{-1} in 1995 (27%).

Because the water-year and N-budget methods gave similar results, we arbitrarily select the N-budget loading rates for discussion. The NO_3-N loading rates for the three years of sweet corn were 129-182 kg ha^{-1} y^{-1}, and for potato 231 or 291 kg ha^{-1} y^{-1} (north or south field-half). These loading rates are reasonable for typical conditions, except the value of 291 kg ha^{-1} y^{-1}, which resulted when the grower attempted to replace N believed to have leached out of the root zone after very heavy rains. Nitrate-N loading in the four-year period averaged 75% of fertilizer N or 62% of total N input. Heavier N fertilizer applications were associated with greater NO_3 loading rates, but even with a fertilizer application rate slightly less than the UW–Extension recommendation (Kelling et al., 1991), the annual NO_3-N loading to groundwater was 129 kg ha^{-1}.

These four years of data can be used to estimate the annual N-budget loading in a rotation of potato-sweet corn-sweet corn, which is common in the Central Sands. Taking the average loading for each crop and weighting for 2 y of sweet corn and 1 y of potato, the loading rate was 190 kg ha^{-1} y^{-1}. Since our potato data came from the atypical crop year of 1993, this rotation might ordinarily load less NO_3 to groundwater. A more typical potato N budget like that of Table 4-4 can be calculated using the UW-Extension fertilizer recommendation of 258 kg ha^{-1} and a yield of 39 Mg ha^{-1} (1990-1994 average yield; WASS, 1995). This budget predicts 208 kg ha^{-1} y^{-1} N loading, and implies a three-year average annual loading of 173 kg ha^{-1}.

Chapter 5
Groundwater quality downgradient from irrigated vegetable fields

Chapters 3 and 4 of this report and the previous work of Kraft et al. (1995) described four years of groundwater conditions beneath an irrigated vegetable field (Field 2). To summarize, the upper 3 m of groundwater beneath the field averaged 20 mg L^{-1} NO_3-N and 20 mg L^{-1} Cl. The sum of detectable pesticides was typically about 1 µg L^{-1} (maximum of 8.8 µg L^{-1}), with atrazine and its progeny products being most common, followed by metribuzin and metolachlor. This chapter describes an investigation into the fate of agrichemical solutes (NO_3, Cl, and pesticides) as they advected from vegetable fields toward groundwater discharge areas. This was accomplished by monitoring groundwater quality in the study area, particularly along groundwater flow paths downgradient from the fields. The monitoring network consisted of 77 wells in nests of 1-6 at 21 locations (Figure 5-1). Forty-two wells were installed during the current study, as described in Chapter 2. The remainder were installed for the study of Kraft et al. (1995). The pre-existing wells were sampled three times during the current study and up to four times during the previous study.

Methods

Sampling procedures consisted of well purging, sample collection, sample preservation, and field analysis. Procedures generally followed the guidelines in Lindorff et al. (1987), and complied with NR 149, Wisconsin Administrative Code. Wells were purged and sampled with an inertial pump. Analyses were for Cl, NO_3-N, pH, and specific conductance, using the methods described in Chapter 3.

In calculating analyte means for individual wells over the various sampling dates, nondetects of inorganic parameters were taken as one half the laboratory detection limit, as discussed in Chapter 3. Simple averages over sampling dates were used. Comparisons among groups of wells employed the Mann-Whitney test for equality of medians, a nonparametric

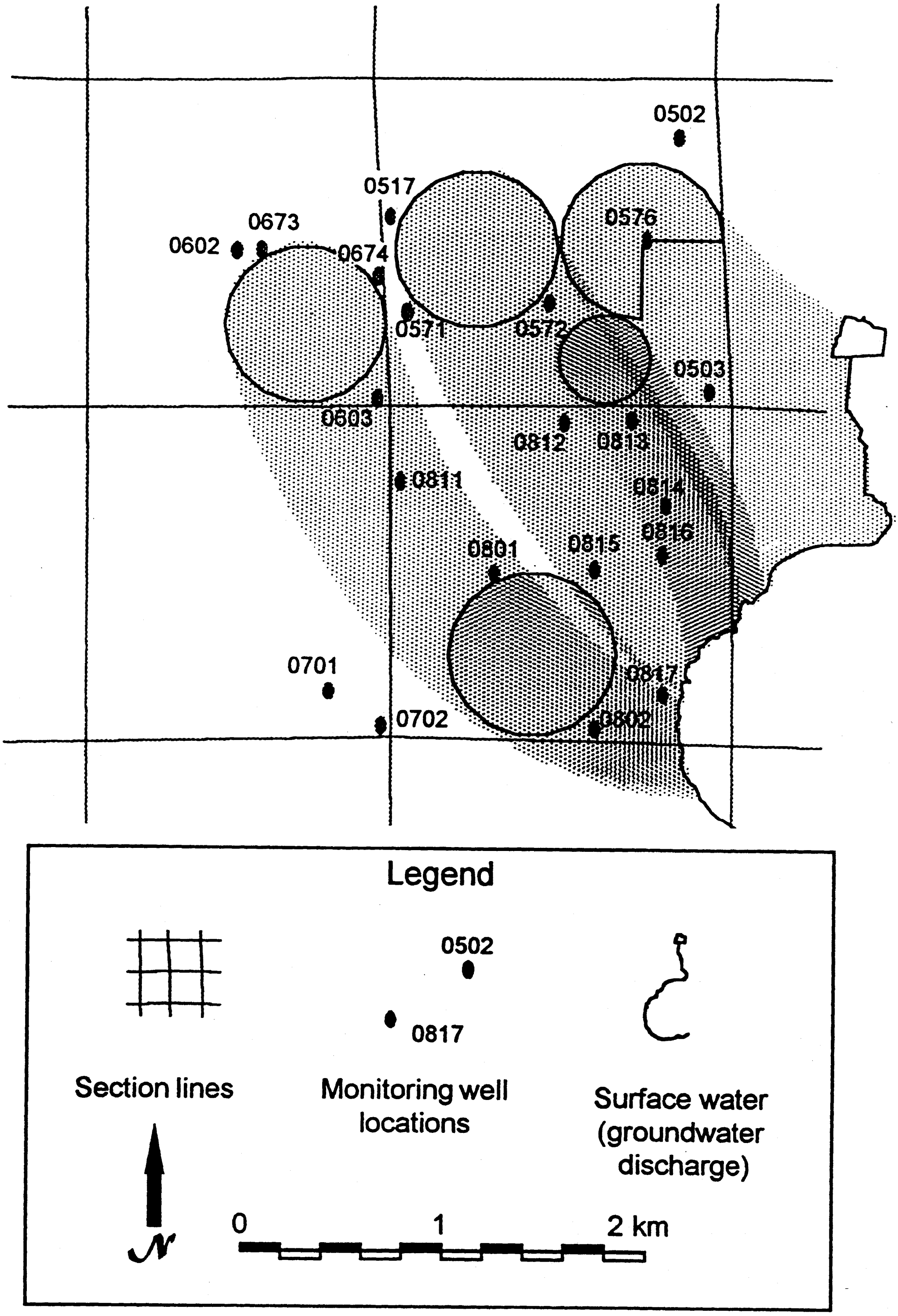


Figure 5-1. Monitoring-well locations in the current study and plan view of simulated plumes.

analogue of Student's *t*. For pesticides, nondetects were so common in vegetable fields that means were taken only among samples where residues were detected.

Groundwater flow and particle tracking models were constructed to simulate advective transport from fields, define flow paths, and outline plumes of groundwater affected by agricultural contaminants. A one-layer flow model was constructed using the computer code MODFLOW (McDonald and Harbaugh, 1988). It employed a variably-spaced grid consisting of 28 north-south columns and 36 east-west rows, extending about 300 to 1600 m beyond the boundaries of the study area. Boundary conditions were constant-head in the north, west, and southeast, and no-flow in the southwest. Constant head values were assigned from measured or extrapolated head measurements, or from surface water elevations. A hydraulic conductivity (K) was assigned to each cell as the thickness-weighted K of the units in the cell. A K of 9.0×10^{-4} m s^{-1} (240 ft day^{-1}) was adopted for the sandy sediments based on field measurements and a model calibration in the nearby Buena Vista groundwater basin (Bradbury et al., 1992). A recharge rate of 0.23 m yr^{-1} was used, which is the area-weighted average of recharge on irrigated lands (0.15 m yr^{-1}) and all other lands (0.25 m yr^{-1}; Weeks et al., 1965). The model was calibrated to heads measured in January 1993, as these data appeared to best represent steady-state conditions. We calibrated by increasing head at some constant-head boundaries, especially near the main discharge zone. Details of the model and its calibration are archived at the Central Wisconsin Groundwater Center.

The particle-tracking code PATH3D (Zheng, 1989) utilized the MODFLOW output to determine groundwater flow paths and plumes advecting from fields. Plumes from fields were delineated by outlining each field perimeter with seven particles, which were traced forward in time until they reached a model boundary.

Results

Inorganic analytes

During the current study, 191 samples were taken from 72 wells, and previously, 86 samples were taken from 27 wells by Kraft et al. (1995). The record for the study area contains a total of 277 samples taken from 77 wells. Nitrate-N concentrations in individual samples over the period of record ranged from < 0.2 to 30.9 mg L^{-1}, and < 1 to 84 mg L^{-1} for Cl. The pH range in individual samples was 4.62 to 7.73, and specific conductance was 2 to 67 mS m^{-1}.

Average analyte concentrations were calculated for each well over all sampling events (Table 5-1). For the population of 77 wells, the median NO_3-N concentration was 6.8 mg L^{-1}, the mean was 8.4 mg L^{-1}, and NO_3-N was detected at ≥ 0.2 mg L^{-1} in 79% of monitoring wells. The median Cl concentration was 19 mg L^{-1}, the mean was 20, and Cl was detected at ≥ 1 mg L^{-1} in 82% of wells. The pH ranged from 4.87 to 7.39, and specific conductance spanned 5-46 mS m^{-1}.

Table 5-1. Summary statistics for inorganic solutes in 77 monitoring wells.

	NO_3-N (mg L^{-1})	Cl (mg L^{-1})	pH	Spec. cond. (mS m^{-1})
Max	29.6	84	7.39	46
Min	< 0.2	< 1	4.87	5
Median	6.8	19	6.23	24
Mean	8.4	20	6.27	23

Pesticides

In the current study, samples for pesticide analysis were collected from 30 wells at 12 locations in March 1996. Only wells vulnerable to pesticide pollution (on the basis of location and inorganic chemistry) were selected for pesticide analysis. Residues detected were atrazine, desethylatrazine, metolachlor, and metribuzin (Table 5-2). Pesticide residues were detected in 19 wells at 10 locations (Figure 5-2; Table 5-3). The sum of detectable pesticide residues was < 1 μg L^{-1} in 80% of wells sampled and ≤ 1.4 μg L^{-1} in 90%. The maximum sum of detections was 22.0 μg L^{-1} (including 20.6 μg L^{-1} metolachlor at location 0813, exceeding the 15 μg L^{-1} Wisconsin enforcement standard). Pesticide residues were not detected at sites 0801 and 0816 (Figure 5-2), the two monitoring locations farthest downgradient from any vegetable field.

Table 5-2. Summary statistics for pesticide residues in 30 wells sampled March 1996. Means include detections only. Medians of all observations, including nondetects, were less than the detection limit for each analyte.

Parameter	No. of detects	Detection limit	Max.	Mean of detects
Atrazine	9	0.1	0.1	0.1
Desethylatrazine	10	0.1	0.3	0.2
Metolachlor	15	0.1	20.6	1.6
Metribuzin	15	0.1	2.3	0.6

For the entire period of record (August 1992 through March 1996), pesticide residues were detected at the same 10 of the 12 sampled locations, in 20 of 38 sampled wells (Figure 5-2; Tables 5-3, 5-4). In addition to the pesticides detected in the current study, carbofuran was detected in one sample (0.1 μg L^{-1}). Six wells had at least one occasion when summed pesticide residues exceeded 1 μg L^{-1}.

Other pesticide residues may have been present, but were not detectable by our analytical method. Some of these residues of pesticides used in the study area and detected in other groundwater studies include alachlor ESA, methyl parathion, methamidophos, and chlorothalonil (Williams et al., 1987; McLean et al., 1988).

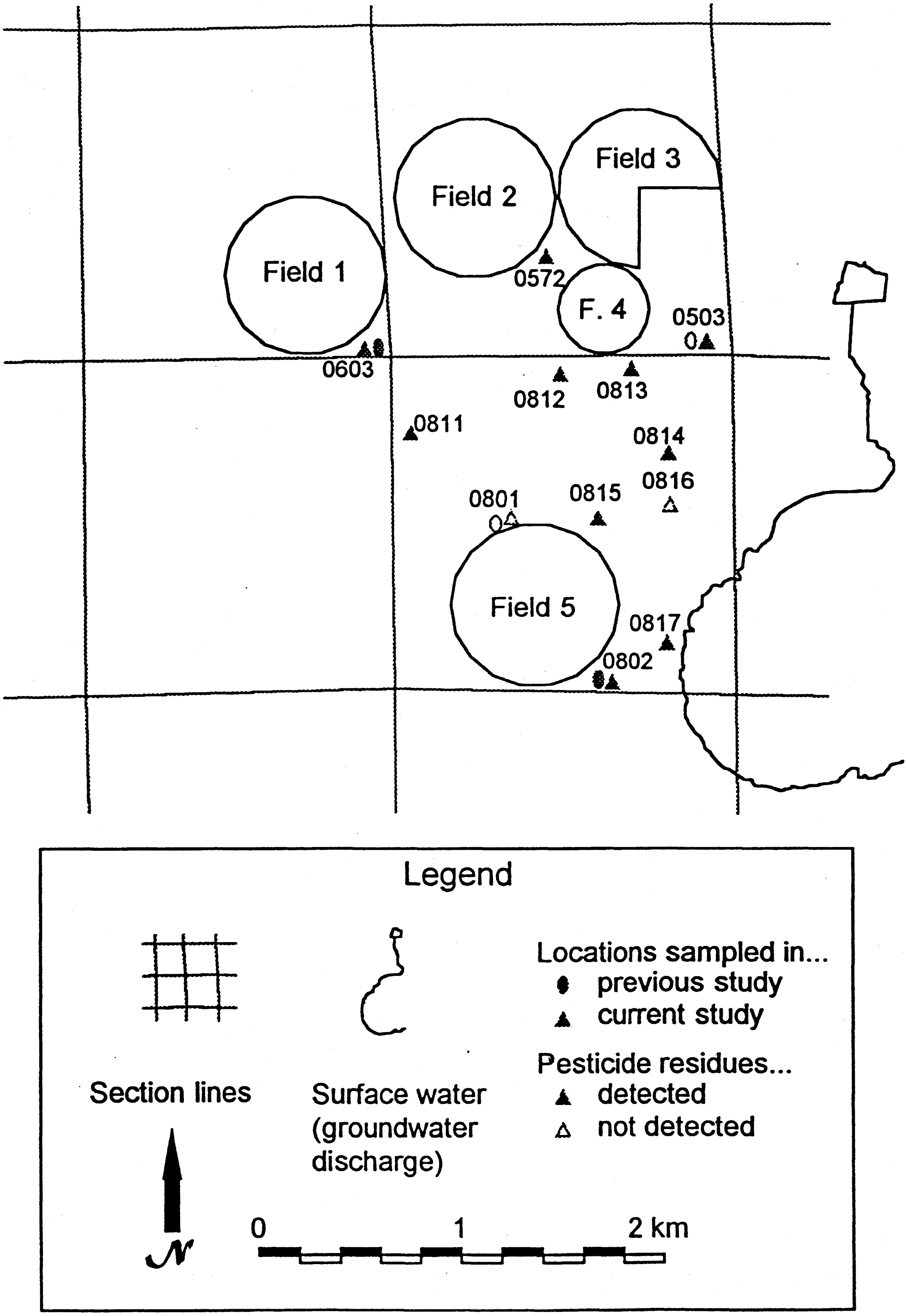


Figure 5-2. Pesticide-monitoring locations.

Table 5-3. Pesticide-residue concentrations in monitoring wells where at least one residue was detected. Some detection limits varied during the period of record.

Location	UWN	Date	Atrazine	Desethylatrazine	Carbofuran	Metolachlor	Metribuzin
0503	EU941	Mar. 96	< 0.1	**0.1**	< 0.1	< 0.1	< 0.1
0572	EU692	Mar. 96	**0.1**	< 0.1	< 0.1	< 0.1	**0.4**
0572	EU939	Mar. 96	**0.1**	**0.3**	< 0.1	**0.1**	**0.6**
0603	EU601	Aug. 92	**0.2**	**0.2**	NA[1]	< 0.4	< 0.1
0603	EU601	Mar. 93	< 0.1	**0.2**	< 0.1	< 0.4	< 0.2
0603	EU601	Mar. 96	**0.1**	**0.2**	< 0.1	**0.4**	**2.3**
0603	EU602	Mar. 93	< 0.1	< 0.1	< 0.1	**0.6**	**0.2**
0603	EU602	Mar. 94	< 0.2	**0.2**	< 0.1	< 0.4	< 0.2
0603	EU602	Mar. 96	< 0.1	**0.1**	< 0.1	**0.1**	**0.1**
0603	EU603	Mar. 93	< 0.1	**0.4**	< 0.1	< 0.4	**0.2**
0603	EU603	Mar. 94	< 0.2	**0.2**	< 0.1	< 0.4	**0.4**
0603	EU603	Mar. 96	< 0.1	< 0.1	< 0.1	**0.1**	**0.1**
0603	EU762	Aug. 92	**0.3**	< 0.1	NA	< 0.6	< 0.1
0802	EU623	Mar. 96	**0.1**	**0.1**	< 0.1	< 0.1	**0.3**
0802	EU761	Mar. 93	< 0.1	**0.8**	**0.1**	< 0.6	< 0.2
0802	EU761	Mar. 96	**0.1**	< 0.1	< 0.1	< 0.1	< 0.1
0811	EU913	Mar. 96	< 0.1	< 0.1	< 0.1	**0.1**	< 0.1
0812	EU932	Mar. 96	< 0.1	**0.1**	< 0.1	**0.1**	**0.2**
0812	EU933	Mar. 96	< 0.1	< 0.1	< 0.1	**0.1**	**0.1**
0813	EU945	Mar. 96	< 0.1	< 0.1	< 0.1	**0.6**	**0.8**
0813	EU946	Mar. 96	**0.1**	**0.1**	< 0.1	**20.6**	**1.2**
0813	EU947	Mar. 96	**0.1**	**0.2**	< 0.1	**0.3**	**0.2**
0813	EU948	Mar. 96	**0.1**	**0.2**	< 0.1	**0.1**	**0.2**
0814	EU928	Mar. 96	< 0.1	< 0.1	< 0.1	**0.1**	< 0.1
0815	EU920	Mar. 96	< 0.1	< 0.1	< 0.1	**0.1**	**0.3**
0815	EU924	Mar. 96	< 0.1	< 0.1	< 0.1	**1.0**	**1.4**
0817	EU919	Mar. 96	**0.1**	**0.2**	< 0.1	**0.1**	**0.8**

[1] Not analyzed

Table 5-4. Summary statistics for pesticide residues in 38 wells through the period of record (August 1992 - March 1996). Means include detections only. Medians of all observations, including nondetects, were less than the detection limit for each analyte.

Parameter	Number of detects	Detection limit	Max.	Mean of detections
Atrazine	10	0.1 - 0.2	0.8	0.2
Carbofuran	1	0.1	0.1	0.1
Desethylatrazine	12	0.1	0.3	0.1
Metolachlor	15	0.1 - 0.6	20.6	1.5
Metribuzin	15	0.1 - 0.2	2.3	0.5

Impacts of vegetable fields on downgradient groundwater quality

Particle-track modeling (Figure 5-1), when combined with cross sections (Figures 5-3 - 5-9), provided a pseudo-three-dimensional view of water quality downgradient of vegetable fields. We chose two cross sections – A and B (Figures 5-3 - 5-5) – to illustrate water-quality conditions along flow paths intersecting fields, and four cross sections perpendicular to flow paths – C through F (Figures 5-6 - 5-9) – to illustrate water quality upgradient of fields and at different distances downgradient.

Upgradient groundwater quality: cross section C

Groundwater upgradient of the northern tier of fields contained little NO_3-N and Cl, averaging 0.5 and 2.1 mg L^{-1} (Figure 5-6), and presumably no pesticides (pesticides were not detected by Kraft et al. (1995) in upgradient samples). Most of the Cl present in this upgradient groundwater is likely attributable to road salt and a few residences.

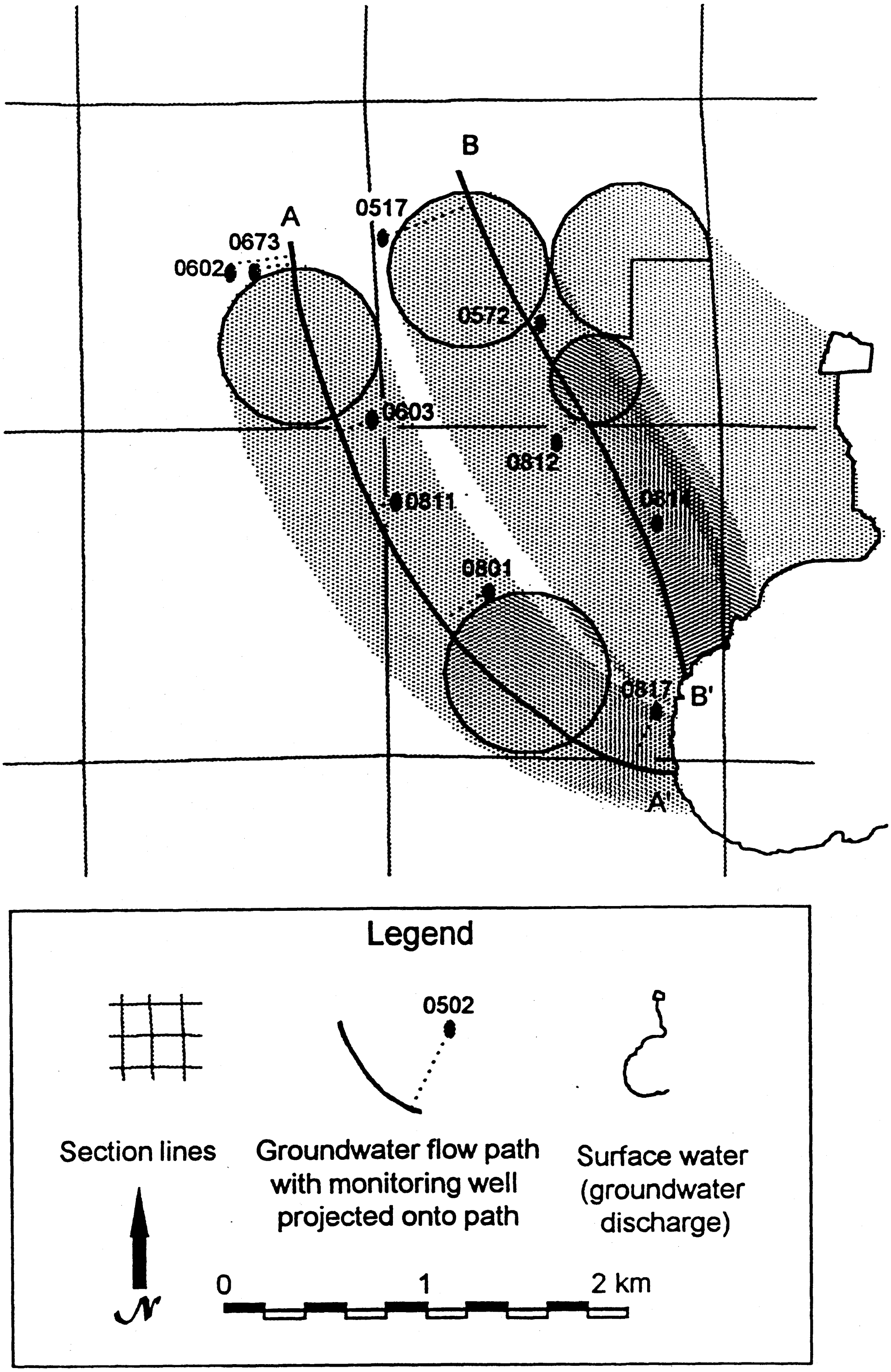


Figure 5-3. Locations of cross sections A and B.

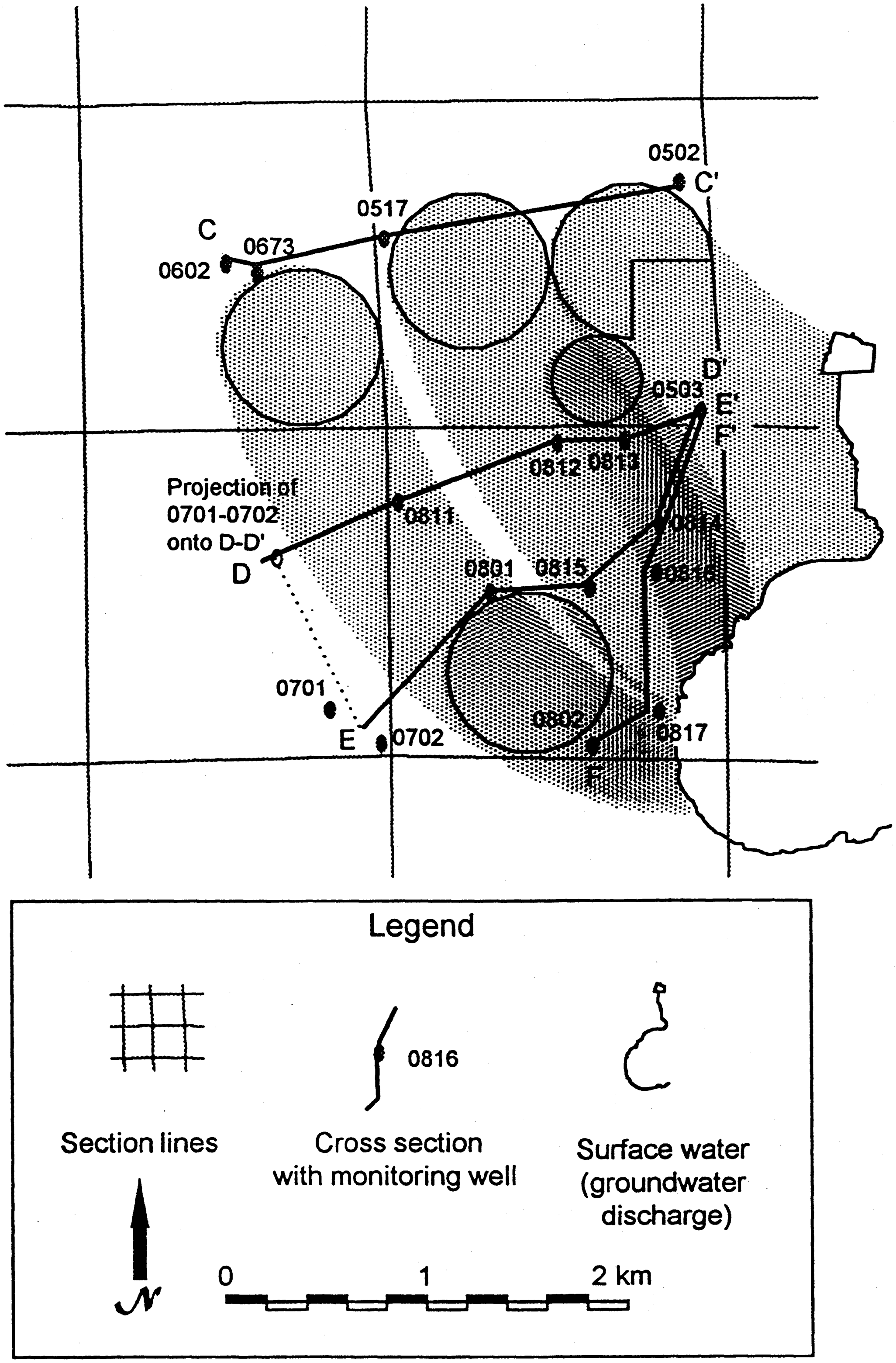


Figure 5-3, continued. Locations of cross sections C-F.

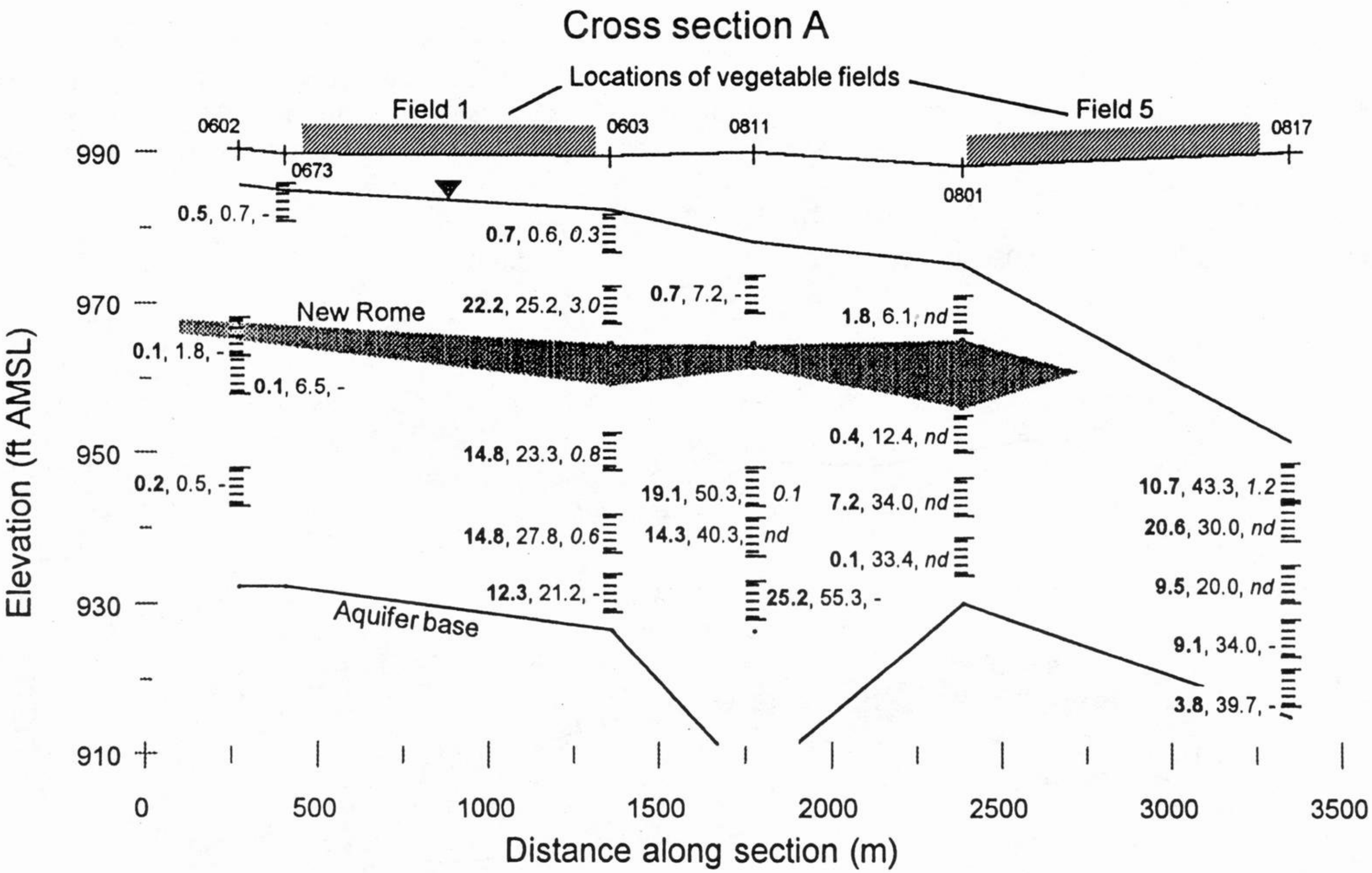


Figure 5-4. Water quality in monitoring wells along cross section A. Next to well screens, NO_3-N concentrations (mg L^{-1}) are shown in bold type, Cl (mg L^{-1}) plain, and maximum observed sum of pesticide residues (µg L^{-1}), italic. A dash (-) indicates no pesticide analysis; *nd* indicates no pesticide detection.

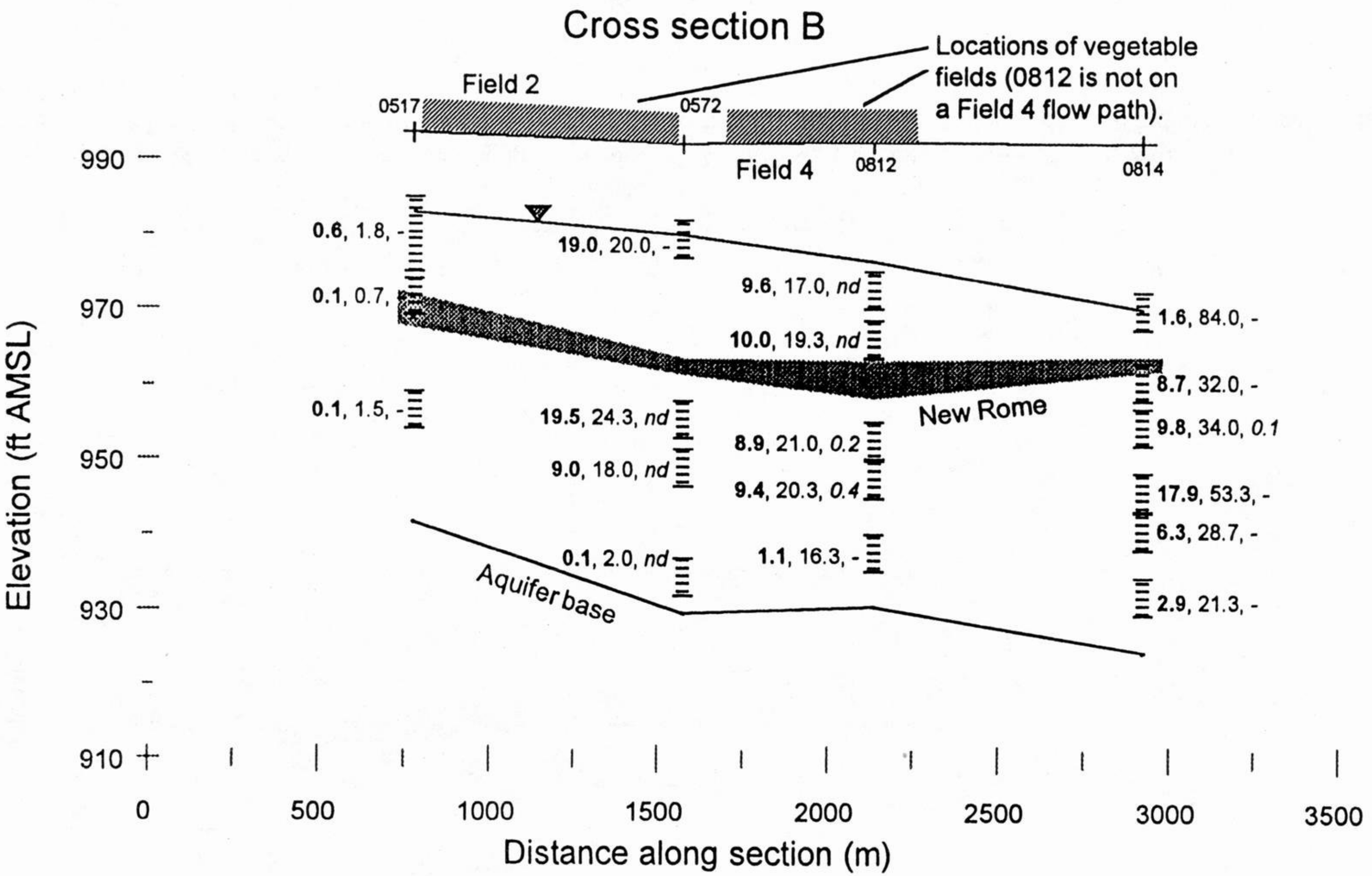


Figure 5-5. Water quality in monitoring wells along cross section B. Next to well screens, NO_3-N concentrations (mg L^{-1}) are shown in bold type, Cl (mg L^{-1}) plain, and maximum observed sum of pesticide residues (µg L^{-1}), italic. A dash (-) indicates no pesticide analysis; *nd* indicates no pesticide detection.

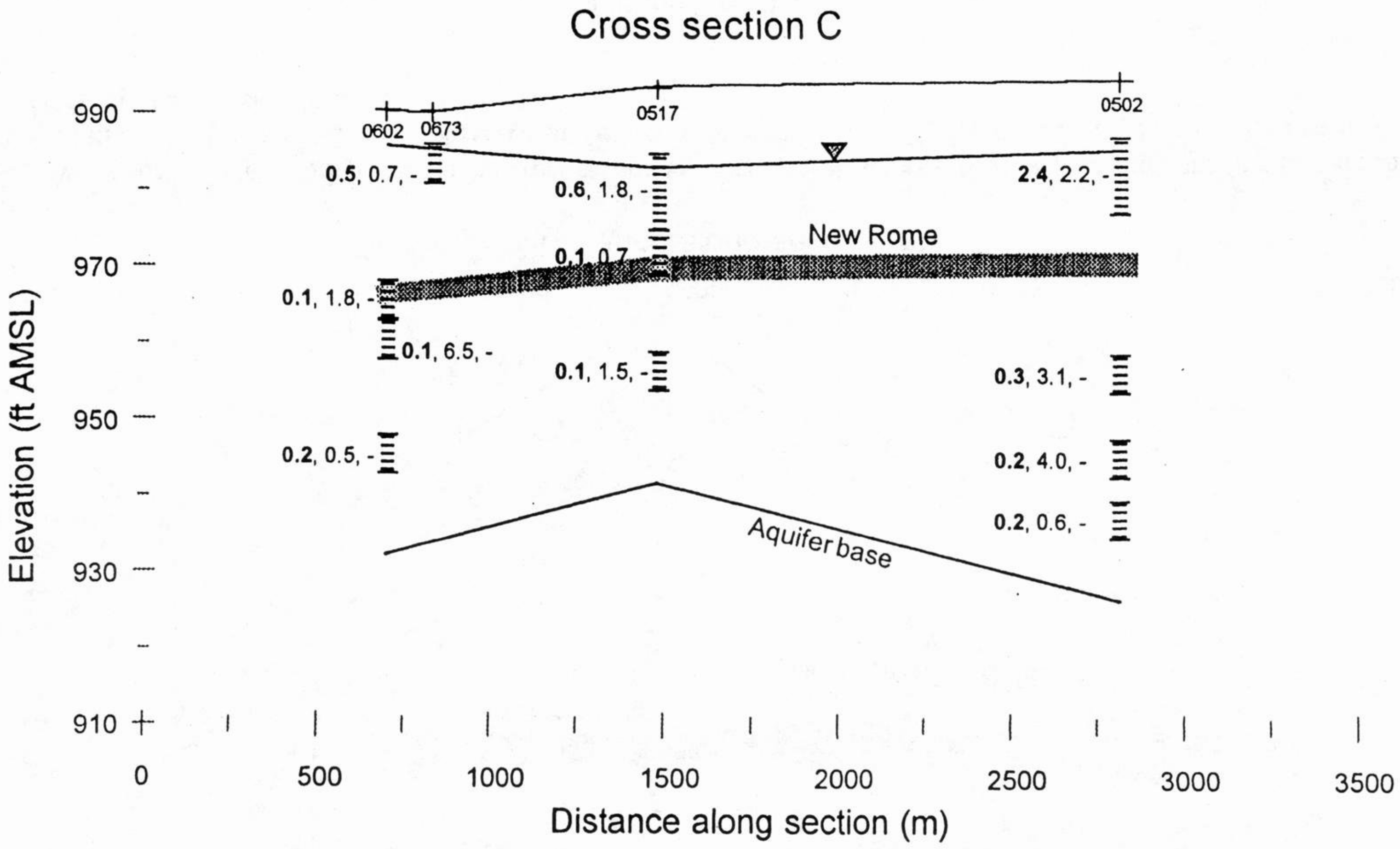


Figure 5-6. Water quality in monitoring wells along cross section C. Next to well screens, NO_3-N concentrations (mg L^{-1}) are shown in bold type, Cl (mg L^{-1}) plain, and maximum observed sum of pesticide residues (μg L^{-1}), italic. A dash (-) indicates no pesticide analysis; *nd* indicates no pesticide detection.

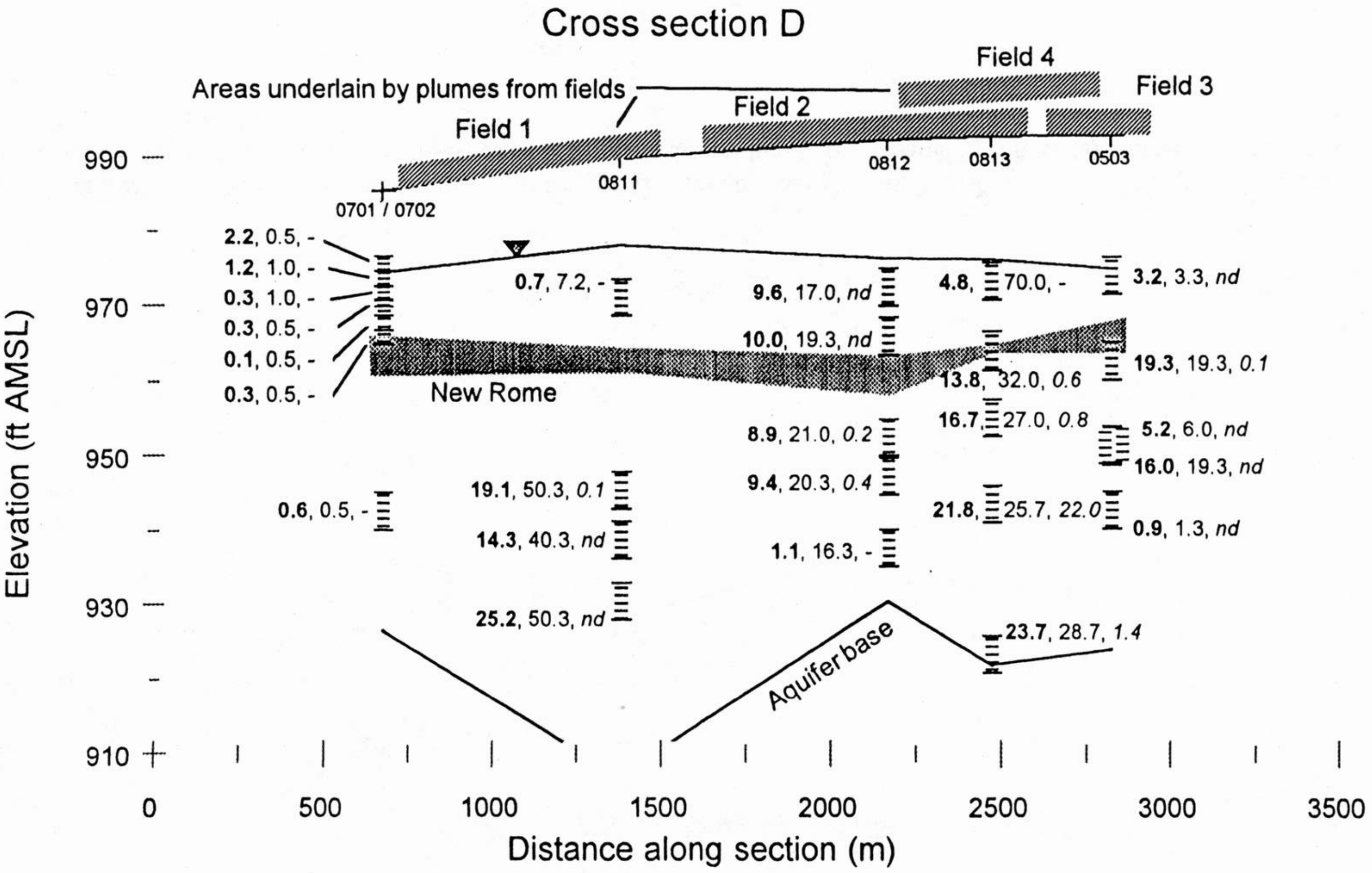


Figure 5-7. Water quality in monitoring wells along cross section D. Next to well screens, NO_3-N concentrations (mg L^{-1}) are shown in bold type, Cl (mg L^{-1}) plain, and maximum observed sum of pesticide residues (µg L^{-1}), italic. A dash (-) indicates no pesticide analysis; *nd* indicates no pesticide detection.

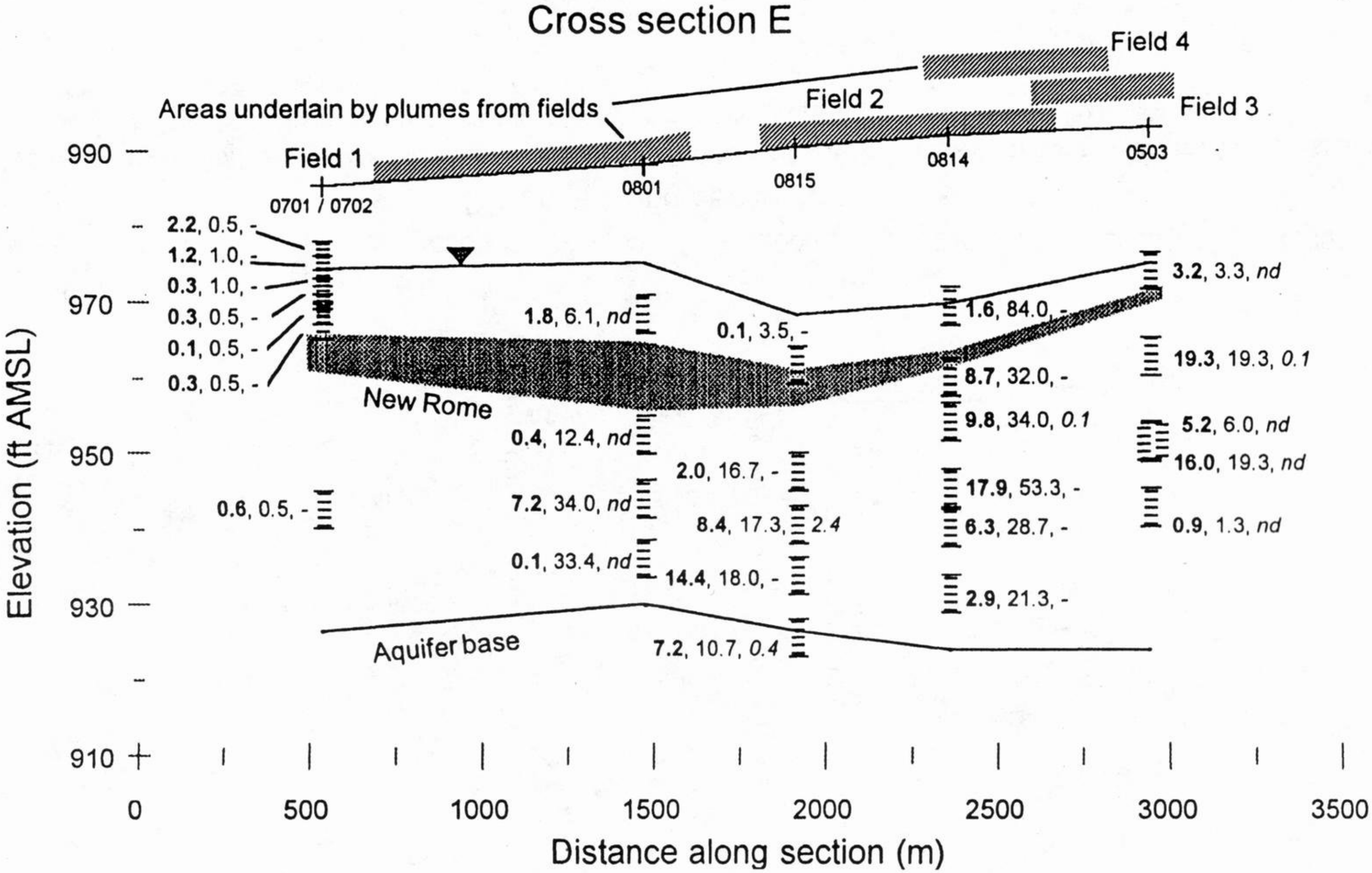


Figure 5-8. Water quality in monitoring wells along cross section E. Next to well screens, NO_3-N concentrations (mg L^{-1}) are shown in bold type, Cl (mg L^{-1}) plain, and maximum observed sum of pesticide residues (µg L^{-1}), italic. A dash (-) indicates no pesticide analysis; *nd* indicates no pesticide detection.

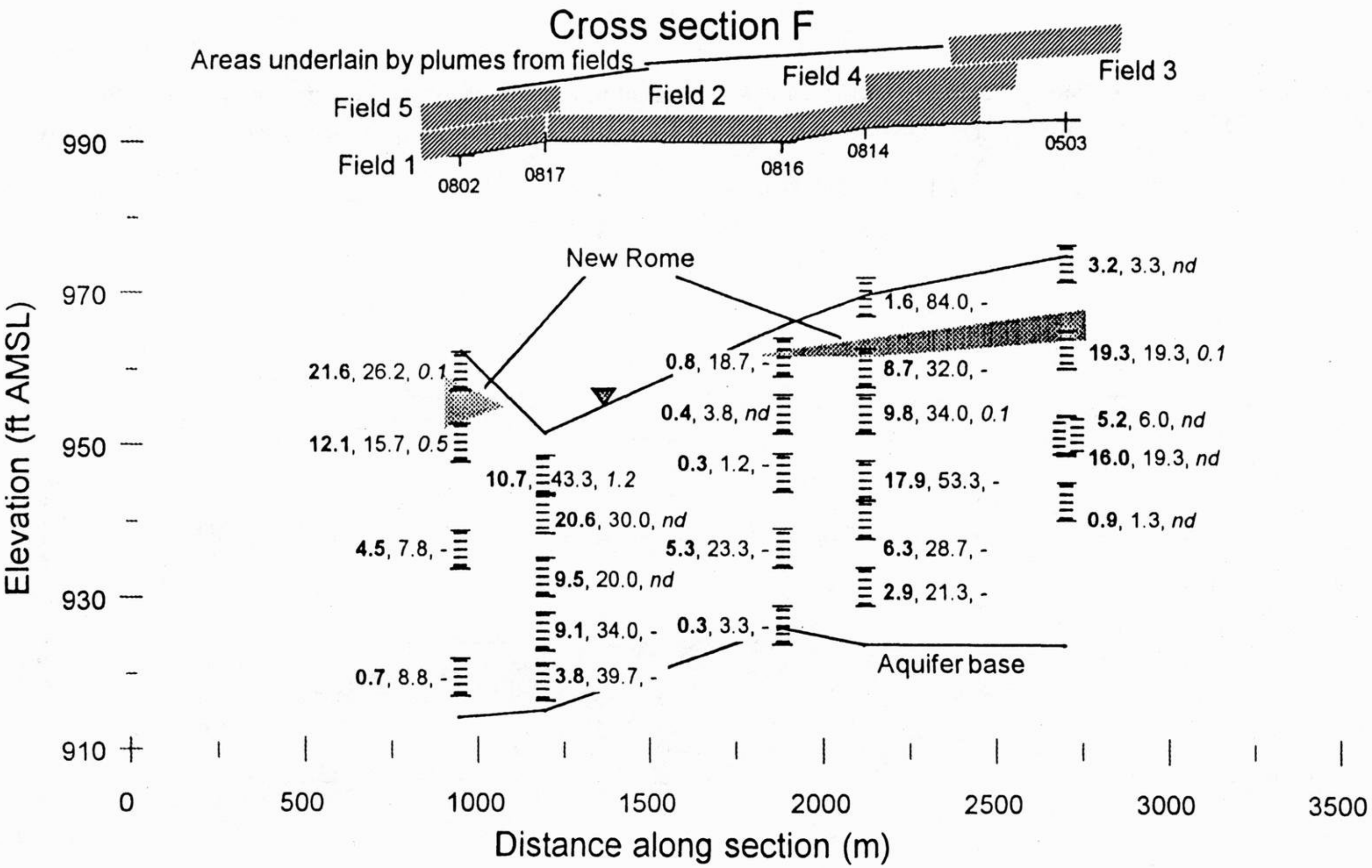


Figure 5-9. Water quality in monitoring wells along cross section F. Next to well screens, NO_3-N concentrations (mg L^{-1}) are shown in bold type, Cl (mg L^{-1}) plain, and maximum observed sum of pesticide residues (µg L^{-1}), italic. A dash (-) indicates no pesticide analysis; *nd* indicates no pesticide detection.

Groundwater quality along a flow line: cross section A

Groundwater quality downgradient of Field 1 was much different from upgradient, as illustrated by location 0603. All wells except the shallowest had NO_3-N and Cl 10-40 times greater than upgradient of the fields. The low NO_3-N and Cl concentrations in the shallowest well at location 0603 reflect recharge from the old-field land cover between the field edge and the well location. The lower 47 ft of the 52-ft aquifer saturated thickness contained NO_3-N in excess of the 10 mg L^{-1} standard. Pesticides were detected in all wells analyzed, but generally at low concentrations, < 1 µg L^{-1} in three of the four wells sampled for pesticides. The effect of Field 1 on such a large fraction of aquifer thickness this close to the field edge is somewhat surprising. It indicates that Field 1 is near the upgradient end of this flow path, and that the silty New Rome member did not preclude vertical contaminant movement.

Four hundred m downgradient of location 0603 at location 0811, NO_3-N and Cl concentrations were small in shallow groundwater but elevated in the deeper 40-80% of the saturated thickness. Small concentrations in shallow groundwater at this location probably resulted from the mixing of recharge water from the forest immediately upgradient of 0811 with water from farther upgradient affected by road salt and perhaps agricultural chemicals. Chloride concentrations in deeper groundwater at location 0811 were oddly greater than those observed immediately downgradient of Field 1, and were also greater than 89% of sample concentrations taken directly below Field 2 (chapter 3). Similar large Cl concentrations were also evident further downgradient. Apparently, older management practices, whose results were observable at location 0811 and beyond, leached more Cl than present practices. Pesticides were detected in only one of two wells sampled for pesticide analysis at location 0811, and then at a concentration of only 0.1 µg L^{-1} .

Another 600 m downgradient, at location 0801, the upper one or two wells appeared to reflect recharge occurring on the forest and old-field land covers between Fields 1 and 5. A large stockpile of paper-mill lime sludge that may be directly upgradient did not have any obvious effects at 0801. Deeper wells at 0801 were NO_3-depleted relative to locations 0603 and 0811, though Cl values were similar. This is evidence that denitrification occurred between 0801 and 0811. Apparent denitrification at this location was previously reported by Kraft et al. (1995). Other evidence of denitrification provided by Kraft et al. included elevated Fe and Mn, indicative of reducing conditions necessary for denitrification, and elevated Ca and Mg associated with groundwater recharged under agriculture. No pesticides were detected at location 0801.

Downgradient from location 0801, the flow path crosses Field 5 before intersecting location 0817. Groundwater at 0817 contained elevated Cl through the entire saturated thickness, and 9-21 mg L^{-1} NO_3-N through the upper 80%. Cl and NO_3 depth profiles at this location were distinctly different. NO_3-N peaked about 10 ft below the water table at 20.6 mg L^{-1} and decreased to 3.8 at the aquifer bottom, while Cl had peaks near 40 mg L^{-1} at the aquifer top and bottom, and a distinct minimum of 20 mg L^{-1} in the middle of the aquifer. A possible explanation is that the Cl minimum at mid-aquifer depth represents the overlapping margins of agricultural plumes from Fields 1 and 5. A corresponding NO_3 minimum was not seen, apparently because denitrification upgradient of location 0801 had depleted much of the Field 1 NO_3. The steady decline of NO_3 with depth may be the result of NO_3 dispersion from the Field 5 plume into the NO_3-depleted Field 1 plume.

Groundwater quality along a flow line: Cross section B

Cross section B was similar to cross section A. Monitoring location 0572, immediately downgradient of Field 2 in cross section B, had NO_3 and Cl concentrations comparable to the analogous location (0603) in cross section A. No pesticide residues were detected in the three wells sampled.

Six hundred m farther downgradient (location 0812), Cl concentrations were about the same as at 0572, and NO_3 concentrations were smaller, but about equal to the MCL (federal maximum contaminant level). Dispersion or perhaps incomplete denitrification may have diminished the NO_3 concentrations (dispersion would affect Cl and NO_3 equally, but the Cl might have been replenished from salt applied to highway 173). Land covers between locations 0572 and 0812 were old field, highway, and sparse residential / hobby farm. The shallowest well may have been affected by either the field, or by inputs such as lawn fertilizers, road salt, septage, and horse manure, though Kraft et al. (1995) did not find such pronounced impacts from residential land use. Pesticide residues were detected in two wells at location 0812, at concentrations $< 2\ \mu g\ L^{-1}$.

Another 1500 m downgradient along cross section B, location 0814 received impacted groundwater from both Fields 2 and 4. Cl concentrations were elevated (> 21 mg L^{-1}) through the entire aquifer thickness, and exhibited peaks at the top and middle of the depth profile. In contrast, NO_3-N exhibited a single peak (17.9 mg L^{-1}) at medium depth. We postulate that the mid-depth peak in NO_3 and Cl is due to the merging of plumes from Fields 2 and 4. Near the water table, old fields and forest land contributed low-NO_3 water while the highway 10-15 m upgradient of the monitoring location added Cl.

To summarize, cross section B showed patterns and processes similar to A, but perhaps without denitrification. The main difference between the flowpaths illustrated in cross sections A and B is that relatively little of B contains NO_3 in excess of the MCL. This may be due to hydrologic conditions, temporal variability, or different past management practices in Fields 1 and 2.

Cross sections transverse to flow paths: Cross sections D, E, F

Cross sections D and E (Figures 5-7 - 5-8) illustrate groundwater conditions at two different distances from Fields 1-4. The cross sections share common endpoints (Figure 5-3), as independent endpoints are not available. Location 0701 / 0702 (common to both cross sections) is not within an agricultural plume, and NO_3-N and Cl concentrations there were low, 2.2 mg L^{-1} or less. Land cover upgradient of this location is mainly forest. Other locations in the cross section intersect plumes from one or two vegetable fields. Along cross section D, most of the saturated thickness lying within plume-containing areas contained NO_3-N greater than or just below the MCL, so that it would be difficult to site a drinking-water well that would reliably meet the standard. Pesticide detections along this cross section were usually < 1 µg L^{-1}, but 20.6 µg L^{-1} of metolachlor was found at location 0813. The metolachlor detection may be related to Kraft's et al. (1995) observation of a uniquely strong pulse of metolachlor in Field 4 upgradient from 0813, peaking at 157 µg L^{-1}.

Nitrate concentrations were generally lower along cross section E, which for much of its length runs about 500 m downgradient of cross section D. Denitrification apparently diminished NO_3 concentrations at location 0801. Few of the wells along cross section E were tested for pesticide residues, but detections were generally low or absent.

Water quality in cross section F displays the influence of plumes from five fields as well as nonagricultural land uses. Location 0802 on the southwestern end of the cross section is immediately downgradient of Field 5 and is also within the plume from Field 1. Groundwater there in the upper third or half of the 50-foot saturated thickness contained elevated NO_3 and Cl as well as some pesticide residues originating from Field 5. This is a much thinner impacted interval than observed in analogous positions farther upgradient (Figures 5-4 - 5-5), and is due to relative positions in the groundwater flow system. Groundwater impacts from Field 1 probably explain the 9 mg L^{-1} of Cl in the deepest groundwater at location 0802, while NO_3 may have been removed by denitrification. Location 0817 is within the plumes of two or three vegetable fields, and the entire saturated thickness had at least 20 mg L^{-1} Cl. There was no indication of denitrification; all but the deepest well had NO_3-N concentrations above or barely below the MCL. Pesticides were detected only in shallow groundwater, at 1.2 µg L^{-1} summed residues. Location 0816 was within the plume from Field 2, and on the edge of the Field 4 plume, but agricultural impacts were not extensive. Only at mid-depth did NO_3-N and Cl of 5.3 and 23.3 mg L^{-1} indicate agricultural influence. Other locations along cross section F have been discussed.

Fraction of aquifer showing agricultural impacts

To estimate the portion of the three downgradient cross sections (D, E, and F) affected by agriculture, we measured the fraction of aquifer containing water whose average concentrations met selected Cl and NO_3 criteria (Table 5-5). Three different concentration criteria were chosen for Cl and for NO_3, to show the sensitivity of the affected fraction to the value of the criterion. Wells containing large concentrations of Cl from road salt were not included with the agriculturally impacted wells. The upper concentration criteria, 15 mg L^{-1} Cl and 10 mg L^{-1} NO_3-N, were chosen somewhat subjectively to indicate a strong agricultural influence; the NO_3 criterion is also the MCL. The lowest Cl criterion (4 mg L^{-1}) was the concentration of minimum overlap between statistical distributions in plume-containing and plume-absent areas (see following section).

Agriculturally impacted water occupied the greatest aquifer fraction in cross section D, which is near the downgradient margin of Fields 1-4. There, 80% of the aquifer exceeded 15 mg L^{-1} Cl, and 68% exceeded 10 mg L^{-1} NO_3-N. The percentages drop to 53 and 14% in cross section E, due to intervening land uses that load little Cl and NO_3 to groundwater, and to denitrification. Denitrification is indicated by the proportionately larger drop in aquifer fraction exceeding the NO_3 criterion, as compared to that exceeding the Cl criterion. The agriculturally impacted fraction of cross section F again increased, due to the groundwater impacts of Field 5.

Table 5-5. Percentages of aquifer occupied by water meeting various Cl and NO_3-N concentration criteria, evaluated for three transverse cross sections.

	Concentration criterion (mg L^{-1})					
Cross section	[Cl] ≥ 15	[Cl] ≥ 10	[Cl] ≥ 4	[NO_3-N] ≥ 10	[NO_3-N] ≥ 8	[NO_3-N] ≥ 5
D	80	82	98	68	76	76
E	53	67	82	14	23	32
F	74	76	87	19	42	59

Differences between agriculturally impacted and nonimpacted groundwater

Water quality in plume-containing areas (as delineated by PATH3D, Figure 5-1) differed from that in plume-absent areas. Median NO_3 and Cl concentrations (averaged over the period of record) were about 20 times greater in plume-containing areas, and the median specific conductance was about double (Table 5-6). Mann-Whitney tests showed the differences were highly significant. The median pH in plume-containing areas was slightly and nonsignificantly less than in plume-absent areas.

Table 5-6. Statistics for four inorganic parameters at locations in vs. outside of plume-containing areas.

	NO_3-N		Cl		Spec. cond.		pH	
Classification	In	Out	In	Out	In	Out	In	Out
Observations	58	19	58	19	57	14	58	17
Maximum	29.6	2.2	84	6	46.2	20.5	7.42	7.14
Minimum	< 0.2	< 0.2	1	< 1	4.5	3.7	4.92	5.36
Mean	9.6	0.5	23	2	23.0	10.6	6.35	6.44
Median	9.0	0.2	20	1	22.9	10.3	6.27	6.64
P (Mann-Whitney)	< 1 e-6		< 1 e-6		1.8 e-5		0.870	

Monitoring wells in plume-containing areas can be screened above, below, or within the plume itself. Therefore, the preceding classification provides clues, but not a precise depiction of the differences between agriculturally impacted and non-impacted groundwater chemistries. A more accurate picture requires distinguishing which wells in plume-containing areas are actually screened in plumes. We made the distinction by using Cl concentration as another discriminating characteristic, which was reasonable given the disparity in Cl concentrations. A critical Cl concentration of 4.0 mg L^{-1} was the point of least overlap between Cl-concentration distributions in plume-containing and plume-absent areas. Eight of the 58 wells in areas underlain by plumes had Cl concentrations less than 4.0 mg L^{-1}. If we classify those wells together with the 19 wells outside areas underlain by plumes, the statistics are as shown in Table 5-7. With this classification, the differences in NO_3-N and specific conductance medians between in- and out-of-plume wells were slightly greater, and the statistical significance also increased. It was no longer valid to test Cl medians between classes, because Cl was now an independent variable.

Table 5-7. Statistics for four inorganic parameters, classified according to whether they were in plume-containing areas and whether the Cl concentration was characteristic of plume-absent areas.

	NO_3-N		Cl		Spec. cond.		pH	
	mg L^{-1}				mS m^{-1}			
Classification	In	Out	In	Out	In	Out	In	Out
Observations	50	27	50	27	5	2	50	25
Maximum	29.6	3.2	84	6	46	20	7.42	7.14
Minimum	< 0.2	< 0.2	4[1]	< 1	9	4	4.92	5.36
Mean	11.0	0.6	26	2	25	11	6.47	6.66
Median	9.6	0.3	22	1	26	11	6.21	6.54
P (Mann-Whitney)	< 1 e-6				< 1 e-6		0.711	

[1] by definition

Conclusions

For the 77 monitoring wells in the study area during the January 1992 through March 1996 period of record, means of NO_3-N, Cl, specific conductance, and pH were 8.4 mg L^{-1}, 20 mg L^{-1}, 23 mS m^{-1} and 6.27. Medians were 6.8 mg L^{-1}, 19 mg L^{-1}, 24 mS m^{-1} and 6.23. Wells in areas containing agricultural plumes had significantly greater NO_3, Cl, and specific conductance than those outside plume-containing areas. Comparisons of medians are: NO_3-N, 9.0 mg L^{-1} in plume-containing areas and 0.2 mg L^{-1} outside; Cl, 20 and 1 mg L^{-1}; and specific conductance, 23 and 10 mS m^{-1}. When the set of impacted wells was narrowed from those within plume-containing areas to those actually intercepting plumes (distinguished by Cl concentration), the median NO_3 and specific conductance were slightly increased.

Thirty-eight monitoring wells were chosen for pesticide residue analysis based on their susceptibility to pesticide pollution. Pesticide residues were detected in 20 of the 38 wells. Detected residues were atrazine (10 wells), desethylatrazine (12), carbofuran (1), metolachlor (15), and metribuzin (15). Other pesticide residues not detectable by our analytical method may have been present. Concentrations tended to be low; 14 of the 20 wells with detections contained < 1 mg L^{-1} of summed pesticide residues on all sampling dates. Of the six wells with summed residues > 1 mg L^{-1}, five were less than 250 m downgradient of a vegetable field. The sixth was 1350 m downgradient and had a summed pesticide residue concentration of 2.4 μg L^{-1}.

The maximum sum of detectable pesticide residues was 22 μg L^{-1}. It was found 80 m downgradient from a vegetable field (location 0813). This large concentration was chiefly due to 20.6 μg L^{-1} of metolachlor, a value in excess of the Wisconsin groundwater enforcement standard (15 μg L^{-1}). The metolachlor detection might have been associated with a strong metolachlor pulse identified in the nearby upgradient field in December 1993 (Kraft et al., 1995). The pulse had a maximum concentration of 157 ug L^{-1}, and was found at only one of the six monitoring locations in the field. It may have resulted from some unusual event, possibly a spill, in the field.

Groundwater quality was related to position within the local groundwater flow system. Upgradient of the northern tier of fields in the study area, groundwater contained little NO_3-N and Cl, averaging 0.5 and 2.1 mg L^{-1}. By contrast, in a cross section perpendicular to flow and 100 to 600 m downgradient of the fields, 68% of the groundwater exceeded 10 mg L^{-1} NO_3-N, 80% exceeded 15 mg L^{-1} Cl, and pesticide detections were common. The effect of vegetable fields on such a large fraction of the aquifer was unexpected. It resulted from the fields being in the upgradient portion of the local groundwater flow system, and is evidence that the silty New Rome member is a poor barrier to groundwater and solute migration. In a cross-section roughly 500 m farther downgradient, the percentages dropped to 14 and 53, and pesticides were also less prevalent. This was largely due to intervening land uses (forest, old field, and highway) that load little NO_3 and, except highways, little Cl to groundwater, and also denitrification. Still farther downgradient, near the groundwater discharge, some of the groundwater flow paths had traversed another vegetable field. There the percentages rose again to 19 and 74, but summed pesticide residues were less than those in wells next to Fields 1 and 2.

Land uses other than agriculture contributed solutes to groundwater, but most non-agricultural land in the study area was old field or forest, land uses with low solute loading rates. The clearest contribution was seen in wells near the water table immediately down-gradient from highways, which were marked by Cl concentrations up to 84 mg L^{-1}. A monitoring well screen crossing the water table downgradient of a small residential / hobby farm area along a highway contained 10 mg L^{-1} NO_3-N and 17 mg L^{-1} Cl. We cannot ascertain whether these NO_3 and Cl concentrations resulted from those nearby land uses, or from a field farther upgradient.

Denitrification was evidenced in one area (around location 0801) by the relatively greater decrease of NO_3 compared to Cl along a flow path. In some wells virtually all NO_3 was removed, and in others the removal was only partial. This was one of two places where potential denitrification was observed in the Port Edwards Groundwater Priority Watershed by Kraft et al. (1995), who also cited elevated Fe and Mn concentrations as evidence for conditions conducive to denitrification. Our observations support their conclusions that denitrification occurs in localized areas within the Watershed.

In summary, vegetable agriculture had a dominant influence on downgradient water quality, causing much of the saturated thickness to have elevated NO_3 and Cl concentrations. Nitrate concentrations in agricultural plumes frequently exceeded the federal MCL and Wisconsin enforcement standard. Pesticide detections were also common, though usually at low concentrations (< 1 μg L^{-1}). A summed-pesticide concentration of 22 μg L^{-1} was detected at one location 80 m downgradient of a vegetable field. It may have been the result of an unusual event, such as a spill, as opposed to routine field applications. Additional investigation would be needed to determine how frequent such concentration spikes might be, and what impacts might result. The pesticide picture is further limited because our analytical methods could not detect residues of some commonly applied pesticides.

The denitrification observed in this study was a fairly localized phenomenon for the Port Edwards Groundwater Priority Watershed, and not significant in controlling NO_3 fate. This is likely true for much of the Central Sands, as large areas have NO_3 exceedance problems (e.g,. Hindall, 1978; Karim, 1995; Mechenich and Kraft, 1996). However, groundwater is anoxic (as evidenced by elevated Fe concentrations) in certain areas of the sand plain associated with wetlands, such as the Buena Vista marsh area in southwest Portage County (Holt, 1965). Denitrification may remove substantial groundwater NO_3 in such areas.

We would expect the impacts of vegetable agriculture on groundwater to be more severe in parts of the sand plain more intensively devoted to vegetable agriculture. The study area has a lower density of this land use (23%) than much of the sand plain. For instance, the recharge area for the Stevens Point, Whiting, Plover municipal wells is 40% vegetable agriculture. The concentrations of agrichemicals and the fraction of the aquifer affected would likely be greater than those observed in this study.

Chapter 6

Conclusions

This study was performed to assess the impacts of irrigated vegetable agriculture on groundwater in the Wisconsin central sand plain. We accomplished this by measuring groundwater quality and NO_3 loading under an irrigated vegetable field, as well as monitoring the transport of agrichemicals downgradient of fields.

Groundwater under an irrigated vegetable field

Nitrate-N concentrations in shallow groundwater samples (0-3 m below the water table) under the study field ranged from < 0.2 to 50.5 mg L^{-1} during the January 1992 through March 1996 period of record. The ranges were < 1 to 119 mg L^{-1} for Cl, 3.72 to 7.27 for pH, and 3 to 75 mS m^{-1} for specific conductance. Whole-field average concentrations during the period ranged from 16.2 to 28.1 mg L^{-1} NO_3-N and 10.3 to 36.4 mg L^{-1} Cl. Concentrations were much less in upgradient groundwater (Table 6-1), indicating that virtually all the dissolved solids beneath the field originated from agricultural practices.

Table 6-1. Average concentrations of inorganic parameters in upgradient and in-field groundwater during the January 1992-March 1996 period of record.

Parameter	Upgradient	In-field
NO_3-N (mg L^{-1})	0.7	20.5
Cl (mg L^{-1})	1.4	20.2
Spec. conductance (mS m^{-1})	5.7	33.2
pH	5.07	4.90

Pesticide residues (atrazine, desethylatrazine, desisopropylatrazine, carbofuran, metolachlor, or metribuzin) were ubiquitous beneath the field. Atrazine residues and metribuzin were found in 90% of samples, and metolachlor in about half. Summed pesticide residues in individual samples ranged up to 4.5 µg L^{-1}, and the mean of detections was 1.3 µg L^{-1}.

Other pesticide residues such as alachlor ESA may have been present, but are not detectable by our analytical procedures. The Wisconsin preventive action limit (PAL) for atrazine residues was exceeded in more than 70% of samples. No other exceedances of Wisconsin PALs or Enforcement Standards were found.

Nitrate loading to groundwater

We estimated NO_3 loading to groundwater using a budget method and a method based on groundwater monitoring under the study field (water-year method). These yielded similar loading estimates (Table 6-2). The four-year record showed an average loading rate of 180 kg ha^{-1}, equal to 75% of applied fertilizer N or 62% of total N inputs.

Table 6-2. Comparison of N budget and water-year estimates of NO_3-N loading (both kg ha^{-1}). The N budget was chosen as the basis for calculating discrepancies.

	1992	1993	1994	1995	Mean
Crop	Sweet corn	Potato	Sweet corn	Sweet corn	
N budget	182	261	154	129	182
Water year	178	233	171	132	178
Discrepancy	-2%	-11%	11%	2%	-2%[1]

[1] Cumulative four-year discrepancy

The smallest loading rate, 130 kg ha^{-1} y^{-1}, occurred when the N fertilizer rate was slightly less than the best management practice (BMP) recommendation, and loading increased with increasing N fertilizer inputs. The potato crop took up more N than sweet corn, but still had the greater NO_3 loading rate because it received more N fertilizer.

The NO_3-N loading data were used to calculate a weighted average loading rate of 190 kg ha^{-1} y^{-1} in a potato - sweet corn - sweet corn rotation. Record rainfall in 1993 prompted application of additional fertilizer N, which probably caused unusually large NO_3 loading. The typical loading rate for this rotation might be closer to 173 kg ha^{-1} y^{-1}.

Groundwater quality outside fields

Nitrate-N concentrations in individual groundwater samples outside of irrigated vegetable fields for the period of record ranged from < 0.2 to 30.9 mg L^{-1}, and Cl from < 1 to 84 mg L^{-1}. The pH range was 4.62 to 7.73, and specific conductance 2 to 67 mS m^{-1}. The averages among all monitoring wells were 8.4 mg L^{-1} NO_3-N, 20 mg L^{-1} Cl, pH 6.27, and specific conductance 23 mS m^{-1}.

More than half of groundwater sampling locations outside fields contained at least one detectable pesticide residue. (Monitoring locations were selected to maximize chances of pesticide detections.) Detected residues included atrazine, desethylatrazine, metolachlor, and metribuzin. Pesticide residues that are not detectable by our analytical methods may also have been present, notably alachlor ESA. Few sampling locations contained > 1 µg L^{-1} summed pesticide residues; nearly all such locations were less than 250 m downgradient of a vegetable field. The maximum sum of detectable pesticide residues in any sample was 22 µg L^{-1}, mainly metolachlor, which may have originated with a strong unexplained pulse previously observed in the nearby upgradient field. Additional study would be needed to determine how frequent such anomalies might be.

Effect of position in the flow system

Groundwater contained very little NO_3 and Cl upgradient of the northern tier of irrigated vegetable fields. Immediately downgradient from these fields, NO_3 and Cl in groundwater had increased 10-40 fold. Most of the groundwater contained pesticide residues and exceeded 10 mg L^{-1} NO_3-N, and more than 80% of the saturated thickness was clearly agriculturally impacted. The New Rome silt layer did not prevent contamination in the deepest parts of the aquifer.

About 400 m farther downgradient from the northern tier of fields, after flowpaths had traversed forested or old-field areas, NO_3 and Cl concentrations in shallow groundwater were similar to those upgradient of fields. Deeper groundwater frequently contained at least as much NO_3 and Cl as it did just downgradient from fields, but pesticide detections became sporadic. About 50% of the aquifer was unmistakably occupied by agriculturally-impacted groundwater in a cross section at this position. Denitrification was strongly indicated in deep groundwater at one monitoring location.

Near the discharge zone, some flowpaths crossed another field. Groundwater downgradient of that field contained elevated Cl through the entire saturated thickness, and 9-21 mg L^{-1} NO_3-N in the upper 80%.

Comparisons between agriculturally impacted and nonimpacted groundwater

Groundwater that originated as recharge in vegetable fields formed distinct plumes as it advected away from fields. Groundwater in plume-containing areas had significantly ($P <$ 0.05) greater solute concentrations than in plume-absent areas. The respective medians were NO_3-N, 9.0 and 0.2 mg L^{-1}; Cl, 23 and 2 mg L^{-1}; and specific conductance, 23 and 10 mS m^{-1}. Plumes occupied distinct vertical intervals, so that groundwater in plume-containing areas, but at depths not occupied by a plume, resembled groundwater in plume-absent areas. Hence, a clearer contrast between agriculturally impacted and nonimpacted groundwater was provided by categorizing the between-plume groundwater together with that from plume-absent areas. Concentration differences were greater and more significant with this grouping.

Impacts of other land uses

Forest and old field land covers in the study area loaded very little NO_3 and Cl to groundwater. Highway salt produced locally high Cl concentrations. A small residential / hobby farm area along a highway may have produced NO_3 and Cl concentrations similar to those from vegetable fields, but the proximity of a vegetable field prevented definite identification of the source.

Implications for the Wisconsin central sand plain

Shallow (upper 3 m) groundwater under the study field averaged > 20 mg L^{-1} NO_3-N. We expect that concentrations would typically be greater under vegetable fields throughout much of the sand plain. Because land uses upgradient of the study field had low NO_3 loading rates, upgradient groundwater was virtually NO_3-free. This allowed NO_3 originating from the field to efficiently disperse into groundwater originating upgradient of the field, effectively reducing NO_3 concentrations below the field.

Pesticide residues were ubiquitous under the field, but concentrations were below regulatory standards except for exceedances of atrazine-residue PALs. These results are difficult to generalize, since pesticide persistence and mobility are highly dependent on soil and groundwater chemistry, which vary substantially across the sand plain.

Nitrate-N loading rates during the period of record varied from 130 to 180 kg ha^{-1} y^{-1} for sweet corn, and were up to 270 kg ha^{-1} for potato. Nitrate loading rates for potato in this study are congruent with rates measured at the plot scale in the sand plain; comparable data for sweet corn are not available. Sixty-two percent of total N inputs, or 75% of fertilizer N inputs, leached to groundwater as NO_3-N. We expect that these loading rates are roughly typical for the same crops in much of the central sands, since management practices, field conditions, and climate during the period of study were also roughly typical. An exception was the unusually wet year of 1993. Due to heavy rains that year, extra N fertilizer was applied to counteract perceived fertilizer leaching. However, this practice is common for wet years, so the loading we measured may be representative for a four-year period with typically variable weather.

Vegetable fields had a dominant influence on downgradient groundwater quality in the study area, elevating NO_3 and Cl concentrations through much of the saturated thickness. Nitrate concentrations in agricultural plumes frequently exceeded the MCL. Pesticide detections were also common, usually at low concentrations (< 1 μg L^{-1}), though a metolachlor detection at 22 μg L^{-1} exceeded the Wisconsin enforcement standard. Denitrification in the aquifer was limited to a small area, and was apparently uncommon. We expect that NO_3 pollution would be common downgradient of fields in the sand plain, as in this study, but the pattern and severity of pollution would vary. The northern tier of fields in this study was near the upgradient margin of the flow system, which caused an especially large fraction of the downgradient saturated thickness to be affected by NO_3 and pesticides. Relatively few fields in the sand plain are likely to be in this position, and so we expect that plumes would typically be thinner. The limited denitrification observed in this study indicates that denitrification is not likely to be a significant NO_3 sink, except in places associated with wetlands.

Twenty-three percent of the study area is occupied by irrigated vegetable agriculture, but in many parts of the sand plain, agricultural land use is more intensive. For instance, the figure is 40% in the recharge area for the Stevens Point, Whiting, and Plover municipal wells. Agricultural impacts will be greater than those observed in this study in portions of the sand plain with greater density of agricultural land use.

References

Andraski, T.W. and L.G. Bundy. 1990. Sulfur, nitrogen, and pH levels in Wisconsin precipitation. J. Environ. Qual. 19:60-64.

Attig, John. W., Lee. Clayton, and D. M. Mickelson. 1988. Pleistocene stratigraphic units of Wisconsin 1984-1987. Wisconsin Geological and Natural History Survey, Information Circular 62. 61 pp.

Bartelme, Robert J. 1977. Soil survey of Wood County, Wisconsin. U.S. Dept. of Agriculture Soil Conservation Service, Washington, D.C.

Bundy, L.G., T.W. Andraski, and W.L. Bland. 1997. Nitrogen cycling in crop residues and cover crops. p. 53-61*in* Proc. Wisconsin's Annual Potato Meetings, 1997.

Clayton, Lee. 1986. Pleistocene Geology of Portage County, Wisconsin. Information Circular 56, Wisconsin Geological and Natural History Survey, Madison, Wis.

Clayton, Lee. 1991. Pleistocene Geology of Wood County, Wisconsin. Information Circular 68, Wisconsin Geological and Natural History Survey, Madison, Wis.

Hindall, S.M. 1978. Effects of irrigation on water quality in the sand plain of central Wisconsin. Information circular no. 36. Wisconsin Geological and Natural History Survey, Madison, Wis.

Holt, C.L.R. Jr. 1965. Geology and water resources of Portage County, Wisconsin. Water-supply paper no. 1796. U.S. Geological Survey, Washington, D.C.

Karim, A. 1995. A study evaluating the feasibility of crediting nitrogen in irrigation water in the Stevens Point, Whiting, Plover wellhead protection area and in the Central Sands of Wisconsin. SWP Wellhead Protection Project, Stevens Point, Wis.

Kelling, K.A., E.E. Schulte, S.M. Combs, and J.B. Peters. 1991. Soil test recommendations for field, vegetable, and fruit crops. Univ. of Wis. – Extension publication A2809.

Kolpin, D.W., E.M. Thurman, and D.A. Goolsby. 1996. Occurrence of selected pesticides and their metabolites in near-surface aquifers of the midwestern United States. Environ. Sci. Technol. 30:335-340.

Kraft, G.J, W. Stites, D. Mechenich, and J. Balma. 1995. Port Edwards Groundwater Priority Watershed Groundwater Resource and Agricultural Practice Evaluation. Central Wisconsin Groundwater Center, Stevens Point, Wisconsin.

Lachat Instruments. 1991. Chloride. Method no. 10-117-07-1B, 18 Oct. 1991.

Lachat Instruments. 1992. Nitrate + nitrite. Method no. 10-107-14-1D, 6 Nov. 1992.

LeMasters, G. and Baldock, J. 1995. A survey of atrazine in Wisconsin groundwater. Phase one report. Wisconsin Dep. Agric. Trade Consumer Protection ARM Publ. no. 26.

Lindorff, David E., Jodi Feld, and Jack Connelly. 1987. Groundwater Sampling Procedures Guidelines. Wisconsin Dep. Natural Resour. PUBL WR-153-87. Madison, Wis.

McDonald, M.G. and A.W. Harbaugh. 1988. A modular three-dimensional finite-difference groundwater model. Techniques of Water-Resources Investigations of the United States Geological Survey, Book 6, Chapter A1.

McLean, J.E., R.C. Sims, W.J. Doucette, C.R. Caupp, and W. J. Grenney. 1988. Evaluation of Mobility of Pesticides in Soil Using U.S. EPA Methodology. J. Environ. Engineering 114:689-703.

Mechenich, D.J., and G.J. Kraft. 1996. Contaminant source assessment and management using groundwater flow and contaminant models in the Stevens Point - Whiting - Plover wellhead protection area. Central Wisconsin Groundwater Center, Stevens Point, Wis.

Meisinger, J.J. and G.W. Randall. 1991. Estimating nitrogen budgets for soil-crop systems. p.85-124. *In* R.F. Follett et al. (ed.) Managing nitrogen for groundwater quality and farm profitability. ASA, CSSA, SSSA, Madison, Wisconsin.

Oberle, S.L. and L.G. Bundy. 1987. Ammonia volatilization from nitrogen fertilizers surface-applied to corn (*Zea mays*) and grass pasture (*Dactylis glomerata*). Biology and Fertility of Soils 4:185-192.

Oberle, S.L. and D.R. Keeney. 1990. Soil type, precipitation,and fertilizer N effects on corn yields. J. Production Agriculture 3:522-527.

Olson, R.A. and L.T. Kurtz. 1982. Crop nitrogen requirements, utilization, and fertilization. P. 567-604. *In* F.J. Stevenson (ed.) Nitrogen in agricultural soils. Agronomy monograph no. 22. ASA, CSSA, SSSA, Madison, WI.

Parker, D.T. 1962. Decomposition in the field of buried and surface-applied cornstalk residue. Soil Sci. Soc. Am. Proc. 26:559-562.

Saffigna, P.G., D.R. Keeney and C.B. Tanner. 1977. Nitrogen, chloride, and water balance with irrigated Russett Burbank potatoes in a sandy soil. Agronomy J. 69:251-257.

Schepers, J.S. and A.R. Mosier. 1991. Accounting for nitrogen in nonequilibrium soil-crop systems. p. 125-138. *In* R.F. Follett, et al. (ed.) Managing nitrogen for groundwater quality and farm profitability, ASA, CSSA, and SSSA, Madison, Wisconsin.

Sokal, R.R. and F.J. Rohlf. 1981. Biometry: the principles and practice of statistics in biological research. 2d ed. W.H. Freeman, New York.

USEDS (United States Environmental Data Service). 1988-1993. Climatological data, Wisconsin. U.S. Government Printing Office, Washington, D.C.

USEMSL (United States Environmental Monitoring Systems Laboratory). 1988. Methods for the determination of organic compounds in drinking water. EPA-600/4-88/039. Cincinnati, Ohio.

USEPA. 1990. National pesticide survey phase 1 report. Washington D.C. EPA 570/9-90-015.

USOTA (United States Office of Science and Technology Policy). 1990. Beneath the bottom line: agricultural approaches to reduce agrichemical contamination of groundwater. OTA-F-418. US Government Printing Office, Washington, D.C.

Wagger, M.G., D.E. Kissel, and S.J. Smith. 1985. Mineralization of nitrogen from nitrogen-15 labeled crop residues under field conditions. Soil Sci. Soc. Am. J. 49:1220-1226.

WASS (Wisconsin Agricultural Statistics Service), 1995. Wisconsin agricultural statistics – 1995. Wis. Dep. Agr., Trade, Consumer Protection, Madison.

WDNR (Wisconsin Department of Natural Resources). 1993. Groundwater quality data-base. Madison, Wisconsin.

Weeks, E.P., D.W. Ericson, and C.L.R. Holt, Jr. 1965. Hydrology of the Little Plover River basin Portage County, Wisconsin, and the effects of water resource development. Geological Survey Water-Supply Paper 1811. United States Government Printing Office, Washington, D.C.

Weeks, E.P., and H.G. Stangland. 1971. Effects of irrigation on streamflow in the central sand plain of Wisconsin. U.S. Geological Survey Open-File Report.

Williams, J.S., C.D. Neil, T.K.Weddle. 1987. Influence of Agricultural Practices on Ground Water Quality in Maine. Proc. Fourth Annual Eastern Regional Ground Water Conf. Burlington Vermont. July 14-16, 1987, p 329-341.

Wilner, S.A., K.A. Kelling, and L.R. Massie. 1997. Nitrogen cycling in crop residues and cover crops. p. 43-52 *in* Proc. Wisconsin's Annual Potato Meetings, 1997.

Zheng, C. 1989. PATH3D Version 2.0 user's manual. S.S.Papadopulos & Associates, Bethesda, Md.

Appendix 2-1
Drilling logs for wells installed July 1995

				Depth (ft)		Elevation (MSL)	
Serial #	Loc.	Seg.	Driller's description	From	To	From	To
L0503002	0503	1	Topsoil	0	3	993.1	990.1
L0503002	0503	2	Sand a few pebbles	3	16	990.1	977.1
L0503002	0503	3	Sand some gravel	16	27	977.1	966.1
L0503002	0503	4	Silt New Rome	27	32	966.1	961.1
L0503002	0503	5	Sand	32	38	961.1	955.1
L0507030	0517	1	Topsoil and fine sand	0	5	993.7	988.7
L0507030	0517	2	Sand, few pebbles	5	10	988.7	983.7
L0507030	0517	3	Sand	10	22	983.7	971.7
L0507030	0517	4	Silt, resistant	22	25	971.7	968.7
L0507030	0517	5	Sand	25	28	968.7	965.7
L0507030	0517	6	Sand, some resistance	28	35	965.7	958.7
L0507030	0517	7	Sand	35	40	958.7	953.7
L0507030	0517	8	Sand	40	52	953.7	941.7
L0507030	0517	9	Silt / clay, resistant	52	56	941.7	937.7
L0507030	0517	10	Hillslope deposits	56	60	937.7	933.7
L0502002	0572	1	Topsoil, Sand	0	5	991.5	986.5
L0502002	0572	2	Sand, some pebbles	5	10	986.5	981.5
L0502002	0572	3	Sand	10	15	981.5	976.5
L0502002	0572	4	Sand	15	28	976.5	963.5
L0502002	0572	5	Silt	28	30	963.5	961.5
L0502002	0572	6	Sand	30	35	961.5	956.5
L0502002	0572	7	Sand	35	40	956.5	951.5
L0502002	0572	8	Sand	40	62	951.5	929.5
L0502002	0572	9	Silt, resistant	62	63	929.5	928.5
L0502002	0572	10	Sand & silt layers	63	98	928.5	893.5
L0502002	0572	11	Hillslope	98	104	893.5	887.5
L0502002	0572	12	Decaying granite	104	105	887.5	886.5

				Depth (ft)		Elevation (MSL)	
Serial #	Loc.	Seg.	Driller's description	From	To	From	To
Z0602002	0602	1	Topsoil	0	2	990.3	988.3
Z0602002	0602	2	Sand, fine brown	2	10	988.3	980.3
Z0602002	0602	3	Sand	10	15	980.3	975.3
Z0602002	0602	4	Sand	15	20	975.3	970.3
Z0602002	0602	5	Sand	20	23	970.3	967.3
Z0602002	0602	6	Silt, New Rome	23	25	967.3	965.3
Z0602002	0602	7	Sand	25	30	965.3	960.3
Z0602002	0602	8	Sand	30	35	960.3	955.3
Z0602002	0602	9	Sand	35	40	955.3	950.3
Z0602002	0602	10	Sand and silt	40	50	950.3	940.3
Z0602002	0602	11	Silt	50	55	940.3	935.3
Z0602002	0602	12	Sand	55	58	935.3	932.3
Z0602002	0602	13	Silt, resistant	58	60	932.3	930.3
Z0602002	0602	14	Sand	60	64	930.3	926.3
Z0602002	0602	15	Silt, resistant	64	75	926.3	915.3
Z0602002	0602	16	Sand - silty?	75	86	915.3	904.3
Z0602002	0602	17	Silt, resistant	86	88	904.3	902.3
Z0602002	0602	18	Sand - silty?	88	102	902.3	888.3
Z0602002	0602	19	Silt, resistant	102	104	888.3	886.3
Z0602002	0602	20	Sand - silty?	104	107	886.3	883.3
Z0602002	0602	21	Rock	107		883.3	
Z0600600	0603	1	TOPSOIL and FINE Sand	0	5	989.5	984.5
Z0600600	0603	2	Sand	5	15	984.5	974.5
Z0600600	0603	3	Sand	15	25	974.5	964.5
Z0600600	0603	4	Silt	25	30	964.5	959.5
Z0600600	0603	5	Sand	30	40	959.5	949.5
Z0600600	0603	6	Sand	40	50	949.5	939.5
Z0600600	0603	7	Sand	50	63	939.5	926.5
Z0600600	0603	8	Silt, resistant	63	65	926.5	924.5
Z0600600	0603	9	Sand	65	70	924.5	919.5
Z0600600	0603	10	Silt, resistant	70	80	919.5	909.5
Z0600600	0603	11	Silt	80	85	909.5	904.5
Z0600600	0603	12	Sand	85	90	904.5	899.5
Z0600600	0603	13	Silt	90	91	899.5	898.5
Z0600600	0603	14	Sand	91	95	898.5	894.5
Z0600600	0603	15	Silt, resistant	95	130	894.5	859.5

				Depth (ft)		Elevation (MSL)	
Serial #	Loc.	Seg.	Driller's description	From	To	From	To
Z0800800	0802	1	Topsoil and fine sand	0	5	988.2	983.2
Z0800800	0802	2	Sand, pebbles	5	30	983.2	958.2
Z0800800	0802	3	Sand	30	55	958.2	933.2
Z0800800	0802	4	Sand	55	65	933.2	923.2
Z0800800	0802	5	Sand	65	85	923.2	903.2
Z0800800	0802	6	Silt, resistant	85	98	903.2	890.2
Z0800800	0802	7	Sand? resistant	98	105	890.2	883.2
Z0800800	0802	8	Sand	105	114	883.2	874.2
Z0800800	0802	9	Silt, resistant	114	115	874.2	873.2
Z0800800	0802	10	Sand?	115	125	873.2	863.2
L0801000	0811	1	Sand	0	5	991.5	986.5
L0801000	0811	2	Sand	5	8	986.5	983.5
L0801000	0811	3	Sand, few gravels	9	12	983.5	979.5
L0801000	0811	4	Sand	12	17	979.5	974.5
L0801000	0811	5	Sand	17	22	974.5	969.5
L0801000	0811	6	Sand, some pebbles	22	27	969.5	964.5
L0801000	0811	7	Silt? (drill resistance)	27	30	964.5	961.5
L0801000	0811	8	Sand	30	32	961.5	959.5
L0801000	0811	9	Sand	32	56	959.5	935.5
L0801000	0811	10	Silt, resistant	56	65	935.5	926.5
L0801000	0811	11	Silt, more resistant	65	67	926.5	924.5
L0801000	0811	12	Sand	67	75	924.5	916.5
L0801000	0811	13	Silt?	75	77	916.5	914.5
L0801000	0811	14	Sand	77	87	914.5	904.5
L0801000	0811	15	Silt? (resistance)	87	89	904.5	902.5
L0801000	0811	16	Sand / silt, layers	89	96	902.5	895.5
L0801000	0811	17	Hillslope - grn clay, grv	96	97	895.5	894.5
L0802000	0812	1	Sand, red layers, gravel	0	19	992.5	973.5
L0802000	0812	2	Sand, gravel	19	24	973.5	968.5
L0802000	0812	3	Silt? (resistance)	24	29	968.5	963.5
L0802000	0812	4	Silt (more resistance)	29	34	963.5	958.5
L0802000	0812	5	Sand	34	54	958.5	938.5
L0802000	0812	6	Sand	54	56	938.5	936.5
L0802000	0812	7	Silt (drill resist.)	56	59	936.5	933.5
L0802000	0812	8	Sand/silt, incrsg. resis.	59	62	933.5	930.5
L0802000	0812	9	Silt, resistant	62	64	930.5	928.5
L0802000	0812	10	Sand / silt	64	68	928.5	924.5

Serial #	Loc.	Seg.	Driller's description	Depth (ft)		Elevation (MSL)	
				From	To	From	To
L0803000	0813	1	Topsoil, then Sand	0	5	992.9	987.9
L0803000	0813	2	Sand	5	10	987.9	982.9
L0803000	0813	3	Sand, some gravel	10	15	982.9	977.9
L0803000	0813	4	Sand, gravel	15	20	977.9	972.9
L0803000	0813	5	Sand	20	28	972.9	964.9
L0803000	0813	6	Silt, resistant	28	29	964.9	963.9
L0803000	0813	7	Sand, gravel	29	30	963.9	962.9
L0803000	0813	8	Sand, gravel	30	56	962.9	936.9
L0803000	0813	9	Silt, resistant	56	59	936.9	933.9
L0803000	0813	10	Sand, gravel	59	65	933.9	927.9
L0803000	0813	11	Sand, gravel	65	71	927.9	921.9
L0803000	0813	12	Silt / clay	71	75	921.9	917.9
L0803000	0813	13	Silt / clay, resistant	75	76	917.9	916.9
L0803000	0813	14	Sand / silt, layered	76	88	916.9	904.9
L0803000	0813	15	Hillslope	93	105	904.9	887.9
Z0804000	0814	1	Sand	0	28	992.0	964.0
Z0804000	0814	2	Silt, resistant	28	30	964.0	962.0
Z0804000	0814	3	Sand	30	50	962.0	942.0
Z0804000	0814	4	Sand	50	63	942.0	929.0
Z0804000	0814	5	Silt, resistant	63	68	929.0	924.0
Z0804000	0814	6	Hillslope on auger	68	69	924.0	923.0
L0805002	0815	1	Sand	0	4	990.5	986.5
L0805002	0815	2	Sand, some gravel	4	29	986.5	961.5
L0805002	0815	3	Silt, resistant	29	31	961.5	959.5
L0805002	0815	4	Silt	31	34	959.5	956.5
L0805002	0815	5	Sand	34	49	956.5	941.5
L0805002	0815	6	Sand	49	64	941.5	926.5
L0805002	0815	7	Silt, resistant	64	66	926.5	924.5
L0805002	0815	8	Sand	66	69	924.5	921.5
L0805002	0815	9	Silt	69	71	921.5	919.5
L0805002	0815	10	Sand	71	84	919.5	906.5
L0805002	0815	11	Silt /clay (on auger)	84	88	906.5	902.5

				Depth (ft)		Elevation (MSL)	
Serial #	Loc.	Seg.	Driller's description	From	To	From	To
L0806000	0816	1	Sand	0	5	990.1	985.1
L0806000	0816	2	Sand	5	10	985.1	980.1
L0806000	0816	3	Sand, some gravel	10	20	980.1	970.1
L0806000	0816	4	Sand	20	27	970.1	963.1
L0806000	0816	5	Silt (drill resistance)	27	28	963.1	962.1
L0806000	0816	6	Sand	28	40	962.1	950.1
L0806000	0816	7	Sand	40	45	950.1	945.1
L0806000	0816	8	Sand	45	64	945.1	926.1
L0806000	0816	9	Silt /clay (on auger)	64	65	926.1	925.1
L0807000	0817	1	Sand, pebbles	0	15	990.1	975.1
L0807000	0817	2	Sand, some gravel	15	20	975.1	970.1
L0807000	0817	3	Sand	20	30	970.1	960.1
L0807000	0817	4	Sand, gravel	30	45	960.1	945.1
L0807000	0817	5	Sand	45	60	945.1	930.1
L0807000	0817	6	Sand	60	70	930.1	920.1
L0807000	0817	7	Silt?	70	75	920.1	915.1
L0807000	0817	8	Sand & silt bands	75	93	915.1	897.1
L0807000	0817	9	Bedrock, drill refusal	93	95	897.1	895.1

Appendix 2-2
Construction details of wells in this study

		Elevation							
Location	UWN	Surface	Piezometric surface	Well top	Screen center	Casing length (ft)	Screen length (ft)	Diameter (inch)	Date installed
0502	EU665	994.70	985.40	997.03	956.17	38.36	5	1	5/92
0502	EU666	994.70	985.47	997.03	945.04	49.49	5	1	5/92
0502	EU667	994.84	985.24	997.17	937.07	57.60	5	1	5/92
0503	EU621	993.05	975.02	994.77	951.37	40.90	5	1	5/92
0503	EU622	993.05	974.78	994.76	942.80	49.46	5	1	5/92
0503	EU763	993.18	977.47	995.02	974.13	18.39	5	2	7/91
0503	EU940	993.16	974.01	995.44	951.63	41.31	5	1	7/95
0503	EU941	993.10	974.42	995.39	962.74	30.15	5	1	7/95
0517	EU936	993.67	983.00	996.08	956.58	37.00	5	1	6/95
0517	EU937	993.67	984.75	996.09	971.59	22.00	5	1	6/95
0571	EU635	991.75	982.32	993.77	981.27	10.00	5	2	9/91
0572	EU691	992.19	979.94	994.32	948.83	42.99	5	2	9/91
0572	EU692	991.77	980.90	994.21	979.40	12.31	5	2	9/91
0572	EU938	991.54	977.29	994.39	934.34	57.55	5	1	6/95
0572	EU939	991.54	977.45	994.42	955.37	36.55	5	1	6/95
0576	EU640	993.44	981.00	996.44	979.42	14.52	5	2	9/91
0602	EU668	990.44	985.79	992.44	960.34	29.60	5	1	5/92
0602	EU669	990.44	985.39	992.44	945.44	44.50	5	1	5/92
0602	EU950	990.09	985.80	993.07	965.60	24.97	5	2	7/95
0603	EU601	989.54	982.46	991.64	969.69	19.45	5	2	5/92
0603	EU602	989.59	982.71	991.89	950.09	39.30	5	1	5/92
0603	EU603	989.52	982.69	991.82	939.22	50.10	5	1	5/92
0603	EU604	989.57	982.22	991.87	931.42	57.95	5	1	5/92
0603	EU762	989.23	981.80	992.08	979.21	10.37	5	2	7/91
0673	EU719	989.80	985.01	993.03	983.39	7.14	5	2	9/91
0674	EU633	991.80	983.89	994.35	983.29	8.56	5	2	9/91
0674	EU634	991.76	983.37	994.38	957.72	34.16	5	2	9/91

		Elevation							
Location	UWN	Surface	Piezometric surface	Well top	Screen center	Casing length (ft)	Screen length (ft)	Diameter (inch)	Date installed
0701	EU586	985.48	974.39	987.40	975.62	10.78	2	1	6/92
0701	EU587	985.49	974.40	987.41	973.57	12.84	2	1	6/92
0701	EU588	985.49	974.38	987.41	971.66	14.75	2	1	6/92
0701	EU589	985.49	974.38	987.40	969.59	16.81	2	1	6/92
0701	EU590	985.38	974.27	987.30	967.62	18.68	2	1	6/92
0702	EU659	986.85	973.21	988.77	967.28	18.99	5	2	7/91
0702	EU660	986.52	971.15	988.12	942.62	43.00	5	2	7/91
0801	EU661	988.25	975.12	990.30	968.50	19.30	5	2	5/92
0801	EU662	988.30	974.20	990.35	952.49	35.36	5	1	5/92
0801	EU663	987.98	973.48	990.03	944.06	43.47	5	1	5/92
0801	EU664	988.05	973.55	990.10	936.15	51.45	5	1	5/92
0802	EU623	988.17	962.48	991.27	950.24	38.53	5	1	5/92
0802	EU624	988.16	962.46	991.29	936.21	52.58	5	1	5/92
0802	EU625	988.24	962.29	991.25	919.52	69.23	5	1	5/92
0802	EU761	988.21	961.94	990.51	959.63	28.38	5	2	7/91
0811	EU911	989.97	978.07	992.78	930.47	59.81	5	1	6/95
0811	EU912	989.97	978.07	992.80	938.75	51.55	5	1	6/95
0811	EU913	989.97	978.03	992.74	945.45	44.79	5	1	6/95
0811	EU914	989.70	977.72	991.47	971.11	17.86	5	1	6/95
0812	EU931	992.49	976.48	994.78	937.66	54.62	5	1	6/95
0812	EU932	992.49	976.47	994.74	947.35	44.89	5	1	6/95
0812	EU933	992.49	976.53	994.78	952.52	39.76	5	1	6/95
0812	EU934	992.49	977.44	994.79	965.99	26.30	5	1	6/95
0812	EU935	992.47	977.90	994.68	972.57	19.61	5	1	6/95
0813	EU945	992.90	976.28	995.44	923.35	69.59	5	1	6/95
0813	EU946	992.90	976.32	995.44	943.63	49.31	5	1	6/95
0813	EU947	992.90	976.38	995.45	955.19	37.76	5	1	6/95
0813	EU948	992.90	976.59	995.45	964.04	28.91	5	1	6/95
0813	EU949	992.90	976.64	995.45	973.30	19.65	5	1	6/95

		Elevation							
Location	UWN	Surface	Piezometric surface	Well top	Screen center	Casing length (ft)	Screen length (ft)	Diameter (inch)	Date installed
0814	EU925	991.95	969.72	994.26	928.95	62.81	5	2	5/95
0814	EU926	991.95	970.16	994.47	937.66	54.31	5	2	5/95
0814	EU927	992.07	970.08	994.13	942.88	48.75	5	2	5/95
0814	EU928	991.71	970.08	994.23	951.58	40.15	5	2	5/95
0814	EU929	991.89	970.18	994.88	957.60	34.78	5	2	5/95
0814	EU930	991.95	970.69	994.64	966.94	25.20	5	1	5/95
0815	EU920	990.48	968.29	992.92	925.60	64.82	5	1	6/95
0815	EU921	990.48	968.31	992.90	933.83	56.57	5	1	6/95
0815	EU922	990.48	968.21	992.75	961.55	28.70	5	1	6/95
0815	EU923	990.48	968.25	992.75	947.50	42.75	5	1	6/95
0815	EU924	990.48	968.81	992.75	940.50	49.75	5	1	6/95

Location	UWN	Elevation Surface	Piezometric surface	Well top	Screen center	Casing length (ft)	Screen length (ft)	Diameter (inch)	Date installed
0816	EU951	990.08	965.55	992.58	926.43	63.65	5	1	6/95
0816	EU952	990.08	965.54	992.56	936.40	53.66	5	1	6/95
0816	EU953	990.08	965.52	992.50	946.34	43.66	5	1	6/95
0816	EU954	990.08	965.55	992.52	953.95	36.07	5	1	6/95
0816	EU955	990.08	965.87	992.52	961.47	28.55	5	1	6/95
0817	EU915	990.13	951.60	991.92	918.94	70.48	5	1	5/95
0817	EU916	990.13	951.59	992.80	925.58	64.72	5	1	5/95
0817	EU917	990.13	951.62	993.11	932.66	57.95	5	1	5/95
0817	EU918	990.13	951.62	993.03	940.85	49.68	5	1	5/95
0817	EU919	990.13	951.62	993.42	946.14	44.78	5	1	5/95

Table A4-1. Page 1. Chloride concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 1. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95	96	96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11	2	3
	Day	13	23	9	8	24	29	10	27	4	7	5	11	7	30	29	30	24	30
Port	Elev																		
22	387.5														14				
21	362.5			15		1									15		5		
20	337.5		12	1		36	28	9				5			16	8	4		
18	320.0	1																	
19	312.5		27	15	54	50	37	26	30		10	23	13		15	38	9	9	5
17	292.5	15	19	52	66	55	42	28	28	20	11	11	9	15	23	29	8	9	8
16	277.5	4	23	20	35	60	46	33	32	34	18	21	14	6	12	34	29	6	14
15	262.5																		
14	247.5	22	38	35	27		23	15	17	23	26	26	19	10	9	12	34	14	20
13	232.5																		
12	217.5	62	39	28	19	10	8	9	11	14	21	17	20	16	12	8	31		27
11	202.5																		
10	187.5	78	42	27	21	36	30	16	22	22	19	13	16	14	12	8	10	20	28
9	172.5																		
8	157.5	52	48	33	43	35	27	27	17	18	12	13	14	13	15	16	16	15	20
7	142.5																		
6	127.5	37	57	49	44	29	17	28	8	15	6	11	11	15	17	20	15	17	7
5	112.5																		
4	95.0	24	26	57	43	18	16	31	3	14	4	7	7	8	10	14	9	14	13
3	75.0																		
2	55.0	8	8	41	22	10	4	32	1	12	7	1	4	1	3	5	5	14	12
1	35.0	2	10	22	5	13	6	24	1	10	7	1	2	1	1	2	3	11	7
0	12.5	1	7	12	3	16	8	10	1	8	4	1		1	1	1	1	5	2

Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).

1993	0-16	0-17	0-16	0-12	0-12	0-12	0-12	0-12	0-12	0-6	0-2							
1994	16-18	17-20	16-21	12-19	12-21	12-20	12-20	12-19	12-17	**6-19**	**2-20**	**1-17**	0-16	0-14	0-11	0-10	0-8	0-6
1995												17-19	16-17	14-22	11-20	**10-21**	**8-19**	**6-19**

Cl (kg/ha) in profile from

1993	352	360	389	239	192	135	205	71	126	28	2							
1994	12	45	68	169	222	164	105	103	79	**130**	**161**	130	100	93	83	66	81	46
1995												16	11	90	118	**133**	90	**141**
Sum	364	404	457	408	414	299	310	173	204	158	163	146	111	183	201	199	171	186

Table A4-1. Page 2. Nitrate-N concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 1. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95	96	96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11	2	3
	Day	13	23	9	8	24	29	10	27	4	7	5	11	7	30	29	30	24	30
Port	Elev																		
22	387.5														30.4				
21	362.5			28.0		21.8									49.9		19.5		
20	337.5		35.9	29.6		33.7	33.8	32.4				27.6			39.6	11.0	12.6		
18	320.0	4.0																	
19	312.5		33.5	40.2	26.4	28.0	29.2	37.0	33.8		31.6	32.9	29.7		30.3	22.2	20.6	14.8	17.2
17	292.5	6.9	38.5	14.2	34.7	36.9	33.7	38.7	33.4	32.3	28.3	29.3	37.1	33.2	34.9	13.6	17.0	12.7	10.0
16	277.5	3.7	41.9	33.7	29.8	28.9	25.8	31.4	33.9	33.3	31.6	32.8	32.8	33.1	33.6	19.5	23.2	12.8	12.2
15	262.5																		
14	247.5	18.5	28.2	24.0	33.4	29.4	28.3	29.0	27.3	26.7	28.6	32.7	35.6	34.4	31.7	32.2	28.7	12.9	12.9
13	232.5																		
12	217.5	18.2	27.7	40.3	39.9	33.8	25.1	21.0	21.3	20.5	25.7	28.7	32.4	37.9	32.1	31.5	27.9		21.0
11	202.5																		
10	187.5	18.6	30.4	38.1	36.1	21.2	20.2	28.4	26.9	22.1	23.9	30.9	30.2	33.5	28.1	21.3	28.4	24.8	23.4
9	172.5																		
8	157.5	23.0	20.5	30.3	22.8	15.9	13.5	35.5	27.1	27.7	25.4	33.4	26.2	24.5	26.2	23.9	31.5	28.1	22.9
7	142.5																		
6	127.5	14.9	14.9	18.5	15.0	12.0	6.1	35.8	12.0	30.4	24.3	32.6	22.4	25.3	25.1	26.9	28.9	26.5	13.7
5	112.5																		
4	95.0	12.0	9.0	14.2	14.1	7.7	3.7	33.8	4.0	29.9	17.8	26.8	21.1	20.1	19.3	24.0	16.1	23.7	20.3
3	75.0																		
2	55.0	6.5	5.0	13.7	6.8	3.3	1.1	28.2	2.2	27.5	28.7	4.1	19.1	6.3	8.6	12.5	11.6	22.4	17.7
1	35.0	2.3	6.1	9.4	2.4	4.0	1.1	14.9	2.0	21.5	28.0	3.8	14.2	2.9	2.8	6.6	6.7	16.4	9.3
0	12.5	1.5	5.5	6.3	2.0	4.7	0.6	6.0	1.9	21.3	15.8	3.2		3.7	2.9	5.2	3.7	7.7	3.1

Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).

1993	0-16	0-17	0-16	0-12	0-12	0-12	0-12	0-12	0-12	0-6	0-2							
1994	16-18	17-20	16-21	12-19	12-21	12-20	12-20	12-19	12-17	**6-19**	**2-20**	**1-17**	0-16	0-14	0-11	0-10	0-8	0-6
1995												17-19	16-17	14-22	11-20	**10-21**	**8-19**	**6-19**

NO3-N (kg/ha) in profile from

1993	143	225	264	146	103	71	235	104	225	114	8							
1994	12	81	116	138	187	156	168	130	94	**216**	**341**	**297**	250	197	150	137	132	70
1995												41	30	220	134	**169**	**119**	**139**
Sum	155	306	380	284	290	227	403	234	319	330	349	338	280	418	283	306	251	209

Table A4-1. Page 3. Chloride concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 2. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95	96	96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11	2	3
	Day	19	23	9	11	24	29	10	27	4	7	5	12	7	30	28	30	24	30
Port	Elev																		
22	387.5																		
21	362.5			7											4		6		
20	337.5		7	6		8	14	9				8			9	20	8		
18	320.0	5																	
19	312.5		18	10	26	20	17		17		12	11	10		9	11	10		9
17	292.5	19	13	8	13	19	15	20	24	18	14	11	6		32	31	17	12	9
16	277.5	20	14	6	8	10	17	18	20	17	15	15	7	7	28	41	37	14	6
15	262.5																		
14	247.5	26	25	22	4	10	19	23	20	19	21	19	9	7	16	35	26	12	14
13	232.5									15									
12	217.5	31	31	29	13	25	16	24	16		16	14	11	8	12	13	24	28	22
11	202.5	36																	
10	187.5		21	37	35	20	15	23	16		13	11	10	10	11	8	13	26	21
9	172.5																		
8	157.5	44	20	34	17	17	18		17	7	12	11	9	8	4	12		17	13
7	142.5																		
6	127.5	42	13	34	28	3	21	25	15	7	14	7	3	5	3	10	6		
5	112.5																		
4	95.0	34	3	9	18	1	22	13	12	7	19	4	8	2	6	5	3	8	4
3	75.0	35								7									
2	55.0		1	1	29	1	25	4	13		18	2	9	1	2	5	1	6	1
1	35.0	29	1	1	10	1	20	1	11	4	6	1	2	1	1	2	1	2	1
0	12.5	23	1	1	1	1	22	1	11	1	1	1	4	1	1	1	1	1	1

Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).

1993	0-18	0-20	0-16	0-14	0-14	0-10	0-10	0-10	0-8	0-8	0-4	0-1						
1994			16-21	14-19	14-20	10-20	10-20	10-19	8-17	8-19	4-20	1-17	0-14	0-12	0-10	0-8	0-4	0-2
1995												17-19	14-16	12-21	10-20	8-21	4-17	2-16

Cl (kg/ha) in profile from

1993	**394**	**185**	**208**	**185**	**89**	**157**	**99**	**101**	**35**	**80**	**7**	**5**						
1994			30	43	54	106	123	102	81	101	**116**	82	48	42	47	20	16	2
1995												12	11	93	149	149	140	112
Sum	394	185	237	228	143	263	222	204	115	181	124	99	58	**135**	**196**	**169**	**157**	**114**

Table A4-1. Page 4. Nitrate-N concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 2. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95	96	96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11	2	3
	Day	19	23	9	11	24	29	10	27	4	7	5	12	7	30	28	30	24	30
Port	Elev																		
22	387.5																		
21	362.5			22.8											12.3		24.6		
20	337.5		35.5	28.3		15.9	28.6	24.0				28.8			26.6	11.4	10.1		
18	320.0	38.2																	
19	312.5		33.8	35.8	7.1	11.2	27.0		25.6		22.7	22.7	10.5		16.5	25.4	24.4		13.7
17	292.5	33.3	32.7	36.5	30.0	24.4	29.2	31.0	30.8	28.6	28.0	27.5	22.5		35.3	18.0	12.8	19.7	14.8
16	277.5	31.2	33.3	41.9	26.8	20.1	25.5	27.8	30.5	30.5	28.7	30.0	22.3	23.4	26.4	23.2	17.2	9.9	9.1
15	262.5																		
14	247.5	24.1	23.5	28.8	32.4	21.1	19.2	27.8	23.7	25.7	29.0	31.7	24.9	23.2	21.4	20.4	19.8	10.2	13.9
13	232.5									25.3									
12	217.5	21.9	8.3	24.4	38.1	23.0	26.8	20.3	24.1		26.9	27.8	20.9	25.7	24.1	21.1	24.2	18.8	18.5
11	202.5	10.2																	
10	187.5		2.1	19.8	21.7	11.9	27.9	24.1	25.6		24.4	25.1	13.1	20.7	21.4	23.8	25.3	21.2	19.7
9	172.5																		
8	157.5	3.5	1.8	9.0	20.8	9.9	26.2		25.2	6.6	20.5	19.4	9.7	13.8	11.2	26.3		21.4	20.2
7	142.5																		
6	127.5	1.6	1.4	5.9	8.5	3.0	26.3	19.9	21.2	4.6	14.2	3.8	3.2	8.5	4.9	19.4	13.3		
5	112.5																		
4	95.0	1.4	1.2	2.6	5.6	0.9	24.2	10.5	15.1	2.5	5.5	1.6	8.2	3.8	5.8	7.8	7.7	15.6	4.3
3	75.0	1.6								1.9									
2	55.0		1.2	1.0	11.4	0.5	23.0	4.1	9.4		1.7	0.4	8.8	0.7	2.9	7.6	1.9	7.2	1.2
1	35.0	1.5	1.3	1.0	4.4	0.6	16.1	1.3	2.1	1.2	0.8	0.5	2.1	0.6	0.2	3.7	0.9	1.4	0.4
0	12.5	1.2	1.3	1.2	1.1	1.3	18.0	1.4	1.3	0.6	0.6	0.5	0.6	0.6	0.2	0.8	0.9	0.6	0.2

Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).

1993	0-18	0-20	0-16	0-14	0-14	0-10	0-10	0-10	0-8	0-8	0-4	0-1						
1994			16-21	14-19	14-20	10-20	10-20	10-19	8-17	8-19	**4-20**	**1-17**	0-14	0-12	0-10	0-8	0-4	0-2
1995												17-19	14-16	12-21	**10-20**	**8-21**	**4-17**	**2-16**

NO3-N (kg/ha) in profile from

1993	184	169	138	153	74	175	88	108	17	41	3	1						
1994			121	69	72	168	168	144	123	174	**226**	**137**	104	72	94	43	23	1
1995												17	35	141	133	177	140	127
Sum	184	169	259	222	146	343	256	252	140	215	228	155	139	213	**227**	**220**	**163**	**128**

Table A4-1. Page 5. Chloride concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 3. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95	96	96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11	2	3
	Day	13	22	10	8	25	30	11	28	5	7	6	12	8	29	29	30	24	30
Port	Elev																		
22	387.5																		
21	362.5			22											21				
20	337.5		5	37		17	6								28	10	4		
18	320.0																		
19	312.5		14	12	11	15	25	15	20		16	10	5			13	21	4	
17	292.5	12	17	17	18	15	22	17	23	19	19	12	6		27	16	23	15	25
16	277.5	18	22	22	26	18	26	23	27	26	20	16	6	5	9	13	37	14	
15	262.5																		
14	247.5	30	44	33	38	18	23	24	25	22	23	19	13	9	10	7	35	19	57
13	232.5						24				20								
12	217.5	42	61		45	16		24									22	23	37
11	202.5			47						16									
10	187.5	42	71	43	48	26	34	29	21	16	17	18	16	15	15	16	16	24	22
9	172.5																		
8	157.5	46	50	34	47	40	34	31	22	19	20	21	22	22	20	20	18	17	28
7	142.5																		
6	127.5	40	35	29	41	36	28	34	27	26	25	24	26	28	22	23	22	13	24
5	112.5																		
4	95.0	26	29	26	38	32	28	34	30	30	28	26	28	28	26	24		12	14
3	75.0																		
2	55.0	21	18	19	34	29	25	29	30	31	25	24	29	24	26	21	29	13	16
1	35.0	19	14	15	30	25	16	25	29	31	22	22	29	24	25	15	30	14	19
0	12.5	19	13	12	25	20	10	19	27	30	16	21	27	26	24	12	19	16	22

Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).

1993	0-17	0-20	0-19	0-19	0-12	0-12	0-13	0-10	0-10	0-10	0-10	0-10	0-8	0-6				
1994			19-21		12-20	12-20	**13-19**	**10-19**	**10-17**	**10-19**	**10-19**	**10-19**	**8-16**	**6-16**	**0-14**	0-10	0-4	0-4
1995														16-21	14-20	**10-20**	**4-19**	**4-17**

Cl (kg/ha) in profile from

1993	362	452	342	455	255	226	266	203	201	170	171	194	163	127				
1994			65		88	111	**75**	**128**	**92**	**105**	**87**	**57**	**61**	**88**	**204**	173	52	68
1995														95	49	**150**	**149**	**259**
Sum	362	452	407	455	343	337	341	331	292	275	259	251	224	310	252	323	202	326

Table A4-1. Page 6. Nitrate-N concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 3. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95	96	96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11	2	3
	Day	13	22	10	8	25	30	11	28	5	7	6	12	8	29	29	30	24	30
Port	Elev																		
22	387.5																		
21	362.5			15.8											26.6				
20	337.5		37.5	22.9		23.7	23.4								16.4	10.2	9.9		
18	320.0																		
19	312.5		38.6	38.1	36.6	26.0	25.6	25.5	20.8		19.1	19.2	14.6			13.6	13.0	4.7	
17	292.5	35.5	35.1	33.2	34.5	28.0	24.2	23.5	22.0	20.0	21.8	20.9	16.8		15.6	12.8	13.4	18.2	17.2
16	277.5	32.8	29.6	31.9	29.6	32.1	23.3	23.6	23.0	22.8	24.2	23.9	18.8	16.3	15.4	14.2	14.3	9.5	
15	262.5																		
14	247.5	23.1	19.7	28.2	20.5	33.2	26.2	27.6	29.7	25.7	26.1	27.3	22.5	21.3	16.9	15.2	14.9	11.5	7.4
13	232.5						25.4				26.9								
12	217.5	16.0	15.5		20.5	31.9		28.7									16.8	14.6	11.3
11	202.5			22.1						31.2									
10	187.5	14.3	15.8	19.3	19.0	27.4	25.7	31.3	35.6	31.7	30.5	29.6	23.0	25.2	25.2	24.0	21.1	16.2	14.6
9	172.5																		
8	157.5	15.7	17.2	18.3	19.8	23.0	25.6	31.6	32.7	30.6	29.2	28.9	22.8	25.4	25.5	26.5	26.3	17.2	15.7
7	142.5																		
6	127.5	17.5	18.4	18.0	20.1	20.1	18.5	23.5	28.6	28.6	27.2	27.8	22.1	25.0	26.7	26.8	25.9	18.0	17.5
5	112.5																		
4	95.0	19.0	18.6	18.8	19.0	18.3	17.6	19.9	24.8	26.2	23.8	24.4	22.9	21.8	26.6	27.5		21.2	20.7
3	75.0																		
2	55.0	20.0	16.3	19.3	18.1	17.0	14.9	16.0	22.4	24.3	17.6	18.3	22.5	17.6	26.8	22.6	21.0	22.8	22.4
1	35.0	20.0	14.9	19.2	17.8	15.8	12.1	15.4	20.9	23.1	15.3	16.3	21.4	16.4	26.7	13.8	19.0	23.7	21.1
0	12.5	20.4	15.8	17.8	17.7	15.6	11.2	13.7	19.6	21.1	12.3	15.4	20.5	17.6	25.8	10.0	12.7	24.2	17.6

Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).

1993	0-17	0-20	0-19	0-19	0-12	0-12	0-13	0-10	0-10	0-10	0-10	0-10	0-8	0-6				
1994			19-21		12-20	12-20	**13-19**	**10-19**	**10-17**	**10-19**	**10-19**	**10-19**	**8-16**	**6-16**	**0-14**	0-10	0-4	0-4
1995														16-21	14-20	**10-20**	**4-19**	**4-17**

NO3-N (kg/ha) in profile from

1993	244	303	282	286	180	164	213	195	198	167	173	167	130	135				
1994			58		154	131	**94**	**151**	**118**	**137**	**138**	**110**	**114**	**132**	**248**	164	87	78
1995														73	53	**94**	**132**	**117**
Sum	244	303	340	286	334	295	307	346	316	305	311	276	243	340	301	258	219	195

Table A4-1. Page 7. Chloride concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 4. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95	96	96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11	2	3
	Day	13	22	10	8	25	30	11	28	5	8	6	12	8	29	29	30	24	30
Port	Elev																		
22	387.5																		
21	362.5			19											12				
20	337.5		25	3		63	23								25	11	1		
18	320.0	4																	
19	312.5		9	7	17	41	37	28	17		5	6	25		44	34	10		2
17	292.5	22	18	11	18	34	32	27	18	13	10	11	14		34	25	8	1	17
16	277.5	35	30	11	15	17	34	34	31	20	17	15	4	16	20	27	27	3	5
15	262.5																		
14	247.5	55	43	22	25	19	20	29	46	44	33	27	20	7	6	28	17	8	12
13	232.5																		
12	217.5	40	39	32	39	30	20	21	27	32	21	21		16	10	9		22	24
11	202.5																		
10	187.5	31	31	40	45	39	31	28	21	18	20	20	18	20	16	16	15	24	21
9	172.5																		
8	157.5	23	35	47	50	40	40	31	24	20	21	24	20	22	18	19	18	15	10
7	142.5																		
6	127.5	18	39	45	52	43	43	32	26	24	22	28	19	22	18	20	20	12	12
5	112.5																		
4	95.0	18	38	38	48	39	38	32	28	26	20	26	17	22	18	21	20	16	
3	75.0																		
2	55.0	15	42	36	47	33	37	28	30	25	14	14	28	19	20	22	21	14	14
1	35.0	15	38	32	39	32	33	26	32	26	17	11	29	18	21	23	19	18	16
0	12.5	16	31	44	28	27	33	21	31		13		20	14	23	26	23	18	20

Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).

1993	0-18	0-19	0-20	0-16	0-16	0-13	0-12	0-10	0-10	0-10	0-10	0-4						
1994		19-20	20-21	16-19	16-20	13-20	12-19	10-19	10-17	**10-19**	**10-19**	**4-16**	**0-14**	0-14	0-12	0-10	0-6	**0-4**
1995												16-19	14-16	14-21	12-20	10-20	6-17	**4-19**

Cl (kg/ha) in profile from

1993	325	436	418	450	366	322	245	207	179	137	150	90						
1994		30	21	32	130	134	122	156	128	**103**	**96**	**130**	**184**	169	174	148	80	60
1995												35	19	123	123	89	93	**121**
Sum	325	465	439	482	496	455	367	363	307	240	246	255	203	292	297	237	174	180

Table A4-1. Page 8. Nitrate-N concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 4. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95	96	96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11	2	3
	Day	13	22	10	8	25	30	11	28	5	8	6	12	8	29	29	30	24	30
Port	Elev																		
22	387.5																		
21	362.5			35.9											9.8				
20	337.5		41.4	29.5		26.5	28.3								14.0	8.8	6.7		
18	320.0	50.0																	
19	312.5		39.1	35.4	24.4	24.3	25.3	26.1	20.9		18.2	15.4	13.1		17.3	13.4	13.3		8.2
17	292.5	35.7	40.1	35.0	25.2	26.2	26.0	26.2	23.3	17.7	17.3	19.8	14.1		12.7	11.9	12.6	11.9	26.3
16	277.5	22.1	25.6	35.8	31.0	29.6	24.1	26.9	26.0	19.5	20.1	23.0	13.0	20.9	11.9	16.3	15.9	10.3	10.3
15	262.5																		
14	247.5	13.2	15.7	31.4	28.5	25.7	25.7	27.1	26.3	23.9	25.4	27.0	18.5	15.8	13.3	12.2	22.6	8.7	10.5
13	232.5																		
12	217.5	14.0	22.7	26.2	18.2	23.8	28.8	31.4	29.3	25.8	27.3	29.4		24.1	19.2	16.6		13.0	14.1
11	202.5																		
10	187.5	16.0	24.4	19.3	16.5	26.7	31.7	33.6	33.8	31.3	21.8	33.6	25.7	25.8	26.2	25.0	19.8	13.9	17.8
9	172.5																		
8	157.5	17.0	16.7	16.4	16.5	23.5	27.6	31.0	33.3	32.8	17.1	31.1	20.2	22.6	28.9	28.0	24.8	17.7	20.4
7	142.5																		
6	127.5	18.1	16.1	16.1	17.4	20.8	25.5	29.2	32.9	30.4	18.1	25.1	14.9	19.0	29.2	28.1	25.4	20.8	23.2
5	112.5																		
4	95.0	19.2	16.4	15.7	17.9	19.6	25.4	26.2	33.1	25.3	17.5	21.5	11.8	16.7	28.4	28.0	26.3	25.8	
3	75.0																		
2	55.0	20.4	16.9	16.2	18.2	19.1	25.2	20.8	31.0	24.4	12.8	13.2	19.9	10.3	27.1	27.3	23.9	24.8	25.6
1	35.0	21.3	17.7	16.3	17.6	19.3	22.6	19.7	29.2	25.1	13.9	11.1	19.3	10.0	26.6	27.2	15.4	26.7	26.1
0	12.5	22.8	18.8	16.4	16.2	18.8	22.2	16.0	24.2		11.7		15.0	8.7	19.4	26.7	23.2	27.4	23.4

Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).

1993	0-18	0-19	0-20	0-16	0-16	0-13	0-12	0-10	0-10	0-10	0-10	0-4						
1994		19-20	20-21	16-19	16-20	13-20	12-19	**10-19**	**10-17**	**10-19**	**10-19**	**4-16**	**0-14**	**0-14**	**0-12**	0-10	0-6	0-4
1995												16-19	14-16	14-21	12-20	10-20	6-17	**4-19**

NO3-N (kg/ha) in profile from

1993	308	259	307	214	249	245	227	234	207	120	156	64						
1994		61	51	49	75	122	117	**145**	**107**	**123**	**136**	**138**	**173**	**248**	**231**	176	130	94
1995												26	28	67	69	106	93	**153**
Sum	308	320	358	263	324	367	345	379	314	243	292	227	202	315	299	282	222	248

Table A4-1. Page 9. Chloride concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 5. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95	96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11	2
	Day	19	23	10	7	24	30	11	28	5	8	5	12	8	29	29	30	24
Port	Elev																	
22	387.5																	
21	362.5			49											6			
20	337.5		6	23		25	13								10		1	
18	320.0	9																
19	312.5		11	16	41	59	24	14	10		5	5	37		40	27	7	
17	292.5	20	17	23	50	50	24	25	23	17	13	9	12	25	36	30		3
16	277.5	25	27	18	42	29	23	35	35	30	20	20	9	10	35	36		
15	262.5																	
14	247.5	25	28	24	22	11	14	19	22		30	35	24	16	14	16	37	10
13	232.5									21								
12	217.5	16	31	21	23	20	22	14	13		22	31	26	18	15	13	27	22
11	202.5									16								
10	187.5	17	45	28	34	33	35	29	20	19	15	18	21	11	24			22
9	172.5																	
8	157.5	24	36	33	32	39	39	40	28	28	19	17	17	8	18	21	27	15
7	142.5																	
6	127.5	23	32	30	27	34	36	41	30		24	20	22	11	15	14	26	16
5	112.5									33								
4	95.0	30	47	34	34	24	25	27	34		28	27	23	18	20	15	21	17
3	75.0																	
2	55.0	19	37	37	24	20	11	22	34	28	28	23	20	22	19	14	26	21
1	35.0	17	43	38	15	17	7	14	32	30	27	27	21	24	16	16	27	25
0	12.5	13	24	40	20	11	8	9	18	23	27	30	16	23	16	20	22	24

Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).

1993	0-18	0-20	0-19	0-14	0-14	0-14	0-12	0-12	0-11	0-10	0-8	0-8	0-8	0-6	0-2		
1994			19-21	14-19	14-20	14-20	12-19	12-19	11-17	**10-19**	**8-19**	**8-16**	**8-16**	**6-14**	**2-12**	0-10	0-8
1995												16-19	16-17	**14-21**	**12-19**	**10-20**	**8-17**

Cl (kg/ha) in profile from

1993	269	431	367	264	243	231	228	237	222	185	154	128	113	91	38		
1994			80	122	147	81	92	80	87	**102**	**139**	**101**	**65**	**86**	**104**	186	123
1995												47	18	**117**	**104**	**132**	**80**
Sum	269	431	447	386	389	312	319	316	309	287	293	276	196	294	247	318	203

Table A4-1. Page 10. Nitrate-N concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 5. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95	96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11	2
	Day	19	23	10	7	24	30	11	28	5	8	5	12	8	29	29	30	24
Port	Elev																	
22	387.5																	
21	362.5			24.3											13.9			
20	337.5		22.4	26.8		20.3	25.3								18.3		16.6	
18	320.0	34.8																
19	312.5		29.2	28.0	24.1	28.3	32.9	33.0	23.7		19.8	18.0	19.7		27.4	32.5	15.3	
17	292.5	20.5	24.5	27.0	25.5	29.5	31.5	31.1	23.9	22.3	24.3	21.5	20.8	19.2	28.1	32.2	19.2	21.4
16	277.5	19.0	20.4	25.5	30.4	28.7	28.6	28.2	24.7	25.1	25.8	24.5	18.3	18.7	18.8	26.2	16.1	
15	262.5																	
14	247.5	13.8	11.4	22.2	26.8	31.6	29.6	29.4	25.3		24.6	24.5	21.1	20.4	16.2	20.9	18.8	15.5
13	232.5									23.6								
12	217.5	11.5	21.8	30.6	31.1	29.0	30.1	33.0	30.4		25.7	25.8	28.1	20.8	19.9	17.5	17.8	18.0
11	202.5									26.8								
10	187.5	14.8	17.5	29.8	21.8	24.7	22.9	31.2	34.3	28.5	29.6	28.2	19.8	11.6	25.1			18.6
9	172.5																	
8	157.5	15.7	16.9	24.9	18.3	19.9	19.9	25.9	31.4	25.8	29.6	29.7	17.0	8.6	19.3	20.5	25.9	20.6
7	142.5																	
6	127.5	15.7	16.4	18.9	17.7	17.9	18.5	22.0	29.4		26.1	29.1	23.1	10.9	17.5	13.4	24.2	18.7
5	112.5									18.8								
4	95.0	14.6	15.3	18.5	17.0	15.0	16.4	17.4	24.9		23.3	24.5	25.4	20.6	25.2	13.8	18.8	18.1
3	75.0																	
2	55.0	15.0	15.1	16.9	14.9	13.5	14.6	17.7	17.5	15.8	16.0	17.1	16.7	30.5	20.4	13.4	25.3	20.1
1	35.0	14.7	14.9	17.3	12.8	12.4	13.5	16.5	17.3	15.9	15.1	18.1	16.6	26.3	14.9	14.2	24.3	27.5
0	12.5	12.4	13.3	16.2	12.9	11.8	11.3	14.4	15.5	15.6	15.9	17.2	14.9	22.0	15.1	17.8	17.7	31.1

Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).

1993	0-18	0-20	0-19	0-14	0-14	0-14	0-12	0-12	0-11	0-10	0-8	0-8	0-8	0-6	0-2		
1994			19-21	14-19	14-20	14-20	12-19	12-19	11-17	**10-19**	**8-19**	**8-16**	**8-16**	**6-14**	**2-12**	0-10	0-8
1995												16-19	16-17	**14-21**	**12-19**	**10-20**	**8-17**

NO3-N (kg/ha) in profile from

1993	235	252	283	189	189	192	190	220	164	167	142	124	129	100	35		
1994			65	83	111	121	132	85	95	**136**	**165**	**104**	**80**	**97**	**105**	169	139
1995												38	17	**103**	**111**	**115**	**105**
Sum	235	252	348	272	300	312	322	305	259	302	307	266	225	300	251	284	244

Table A4-1. Page 11. Chloride concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 6. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95		96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11		3
	Day	13	23	10	7	24	30	11	28	5	8	5	12	8	29	29	30		30
Port	Elev																		
22	387.5																		
21	362.5			6		12	7								10	4	18		
20	337.5		10	47		26	3	5				4			47	18	3		
18	320.0	14																	
19	312.5		16	11	6	33	22	23	15	6	9	6	4		47	41	18		
17	292.5	12	20	6	12	24	21	19	14	13	12	8	3	6	40	34	18		10
16	277.5	21	26	9	12	11	16	19	17	17	19	18	5	6	20	37	30		10
15	262.5																		
14	247.5	42	44	37	25	17	18	15	14	14	15	20	24	22	12	16	28		26
13	232.5																		
12	217.5	45	43		44	33	27	18	13	12	16	19	29	29	26	24	16		34
11	202.5																		
10	187.5	37	38	35	55	42	32	27	17	12	12	17	21	27	29	22	16		26
9	172.5																		
8	157.5	24	28	28	56	52	51	46	34	24	17	15	22	20	21	21	24		20
7	142.5																		
6	127.5	20	25	22	42	53	57	39	34	26	19	20	24	22	19	22	25		25
5	112.5																		
4	95.0	24	22	20	31	53	52	36	42	30	26	34	28	26	24	24	23		25
3	75.0																		
2	55.0	19	18	15	27	53	50	36	49	42	38	36	39	35	31	26	19		25
1	35.0	17	18	15	26	57	44	36	48	49	49	40	44	40	33	28	18		18
0	12.5	16	18	16	24	60	36	33	44	51	52	48		45	38		24		17

Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).

1993	0-18	0-20	0-21	0-19	0-16	0-16	0-14	0-12	0-11	0-10	0-8	1-10	0-8	0-6	0-4	0-1		
1994					16-21	16-21	14-20	12-19	11-19	10-19	**8-20**	**10-19**	**8-17**	**6-14**	**4-14**	**1-11**		**0-8**
1995														**14-21**	**14-21**	**11-21**		**8-17**

Cl (kg/ha) in profile from

1993	338	362	341	414	492	441	324	314	260	226	207	236	196	149	101	31		
1994					89	49	66	63	60	78	**107**	**86**	**113**	**110**	**133**	**143**		**141**
1995														**153**	**126**	**130**		**130**
Sum	338	362	341	414	580	490	390	377	320	304	314	322	309	412	360	303		271

Table A4-1. Page 12. Nitrate-N concentrations (mg/L) in MLP ports, ports included in each water year, and mass in each water year. Field 2, MLP 6. Elevation (cm) is with respect to the bottom of the sampled interval. Blanks in upper ports mean that port was above the water table.

	Year	94	94	94	94	94	94	94	95	95	95	95	95	95	95	95	95		96
	Month	3	4	6	8	9	10	12	1	3	4	5	7	8	8	9	11		3
	Day	13	23	10	7	24	30	11	28	5	8	5	12	8	29	29	30		30
Port	Elev																		
22	387.5																		
21	362.5			26.4		32.6	30.2								13.0	6.1	17.9		
20	337.5		39.7	14.8		31.5	21.9	19.5				31.1			18.3	9.6	8.2		
18	320.0	18.8																	
19	312.5		37.9	27.3	26.0	27.6	30.9	26.0	22.0	17.7	17.7	20.7	19.3		20.7	17.3	2.6		
17	292.5	21.6	30.2	31.8	26.8	26.6	28.0	24.6	23.7	22.4	21.5	25.7	17.8	19.4	20.9	16.7	13.7		12.2
16	277.5	23.8	23.7	31.3	29.8	28.0	31.3	30.4	27.2	26.4	26.1	26.5	8.4	17.3	21.6	17.7	15.8		14.4
15	262.5																		
14	247.5	15.6	15.1	17.6	20.7	33.6	30.7	31.6	29.1	28.1	26.5	24.1	22.4	22.1	22.2	19.9	16.9		15.8
13	232.5																		
12	217.5	13.4	14.8		21.5	29.8	31.4	35.0	34.9	29.7	24.3	25.0	27.3	26.6	26.1	23.1	17.6		18.7
11	202.5																		
10	187.5	15.1	15.6	14.5	21.0	26.0	31.6	34.6	35.1	30.8	28.4	26.6	28.5	27.9	27.9	25.0	22.9		20.7
9	172.5																		
8	157.5	16.8	16.9	14.8	18.6	20.1	21.1	24.1	27.6	31.9	33.8	33.1	29.3	29.2	27.4	25.6	26.4		24.2
7	142.5																		
6	127.5	16.7	16.4	14.2	17.1	17.4	18.1	19.4	26.5	31.7	33.6	33.6	30.1	29.2	28.4	26.6	26.3		26.6
5	112.5																		
4	95.0	16.5	16.6	14.9	16.2	15.7	18.0	19.7	19.1	29.8	31.7	27.2	28.6	28.7	27.8	27.8	25.5		28.0
3	75.0																		
2	55.0	16.2	16.2	14.8	15.9	15.4	16.6	15.9	15.0	22.4	25.5	27.3	23.9	26.0	27.8	29.0	20.8		26.0
1	35.0	17.3	16.8	15.0	16.4	15.7	15.7	15.3	15.8	18.2	18.8	24.9	20.4	24.6	25.4	29.3	19.0		17.2
0	12.5	20.0	18.2	16.2	17.1	16.8	15.3	16.1	17.7	16.1	16.2	18.3		22.2	20.5		21.0		14.5
	Ports assigned to water year... (bold type indicates water-year layers selected for evaluation).																		
	1993	0-18	0-20	0-21	0-19	0-16	0-16	0-14	0-12	0-11	0-10	0-8	1-10	0-8	0-6	0-4	0-1		
	1994					16-21	16-21	14-20	12-19	11-19	10-19	**8-20**	**10-19**	**8-17**	**6-14**	**4-14**	**1-11**		**0-8**
	1995														**14-21**	**14-21**	**11-21**		**8-17**
	NO3-N (kg/ha) in profile from																		
	1993	236	288	277	261	240	253	232	205	215	207	172	197	169	133	110	29		
	1994					116	109	107	117	124	132	**202**	**114**	**131**	**128**	**152**	**161**		**148**
	1995														**97**	**71**	**93**		**101**
	Sum	236	288	277	261	357	362	338	322	340	339	374	311	300	358	332	283		249