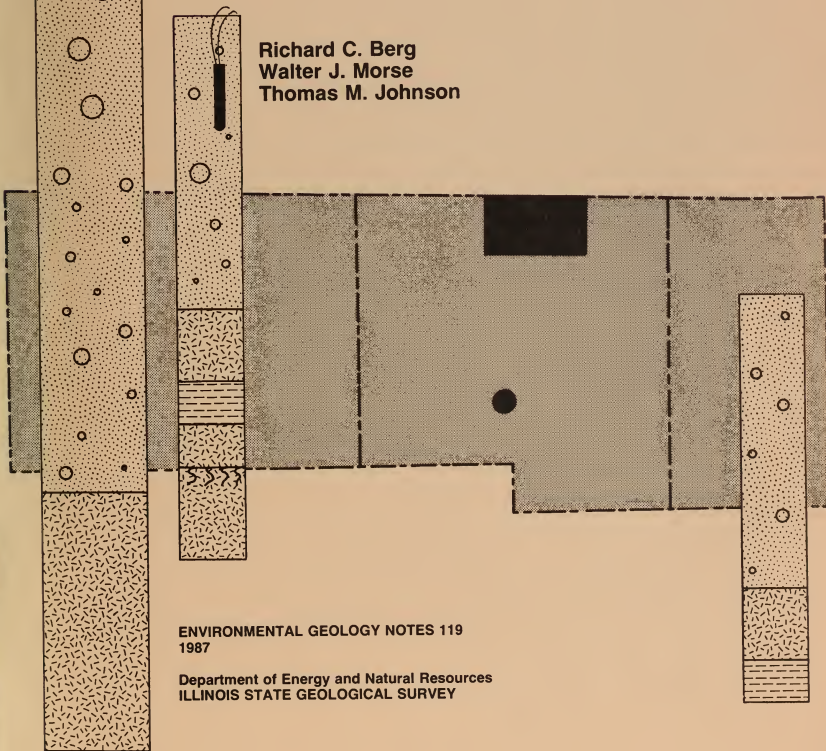


# Hydrogeologic Evaluation of the Effects of Surface Application of Sewage Sludge to Agricultural Land Near Rockton, Illinois

Richard C. Berg  
Walter J. Morse  
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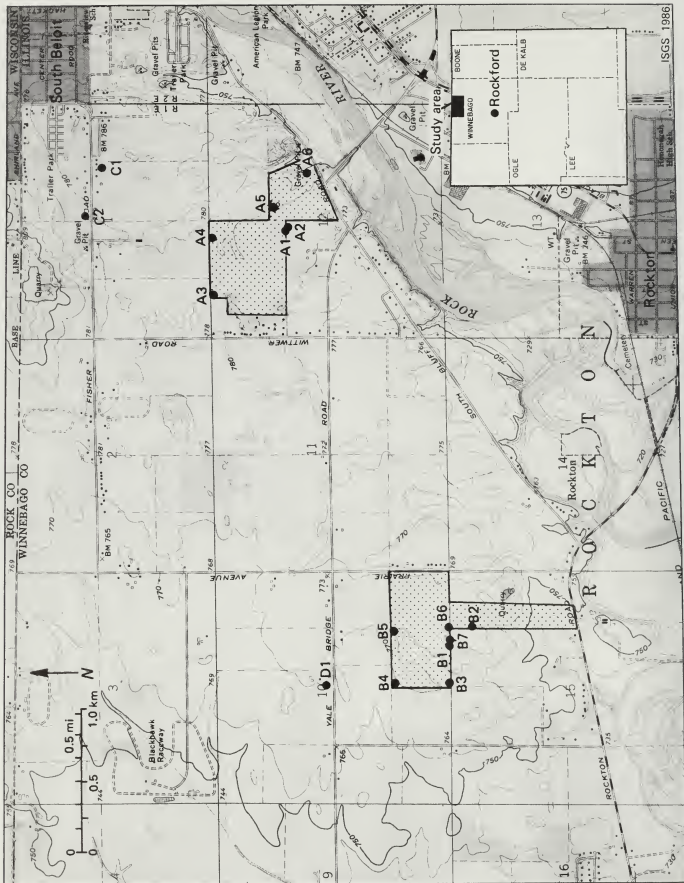


Figure 1 Monitoring well locations on sludge application sites (Sites A and B) and background sites (Sites C and D).

## ABSTRACT

A study of the hydrogeology of sludge application areas in the sandy agricultural land near Rockton, Illinois was conducted to evaluate possible contamination of the groundwater by sludge constituents. Over a 2-year period, a total of 16 monitoring wells was installed at four sites. Two sites were characterized by repeated applications of sludge and two were background sites. Two soil-water samplers were also installed at each sludge application area. The monitoring wells and soil-water samplers were used to determine groundwater flow and quality. Samples were analyzed for nitrates, MBAS, chlorides,  $\text{NH}_3\text{N}$ , pH, hardness, specific conductivity (SC), metals, and organic chemicals. Nitrates provided the most reliable data on the effects of sludge applications on groundwater quality. Nitrate concentrations were higher at sludge application sites than at background sites and higher in downgradient wells than in upgradient wells. Data from other analyses were not conclusive; however, elevated values of MBAS, chlorides,  $\text{NH}_3\text{N}$ , pH, hardness, and SC in downgradient wells suggested some influence from sludge applications on the groundwater. There was no apparent contamination from metals or organic chemicals.

## INTRODUCTION

The application of sewage sludge to agricultural land provides a viable alternative to disposing of it in landfills or via incineration. Application of sewage sludge can actually benefit the land by improving the physical condition of the soil and by recycling plant nutrients. Hydrogeologists are concerned, however, that when sludge is applied to a sensitive hydrogeologic setting, the groundwater could be contaminated by nitrogen, metals, organic compounds, or other sludge constituents. Application rates of sludge must be adjusted to avoid overloading.

The two sludge application areas that were the subject of this investigation are of permeable sand and gravel, a material that is highly susceptible to contamination. Both areas are in Rockton Township, northern Winnebago County, Illinois (fig. 1). Groundwater quality at these areas was compared to two other areas, also in Rockton Township, where there were no records of sewage sludge applications.

The study began in March 1983 and concluded in July 1985. A total of 16 monitoring wells was drilled in the saturated zone of the sludge application and background sites. Two soil-water samplers (lysimeters) were placed in the unsaturated zone of each of the sludge application sites. The research team included three hydrogeologists from the Illinois State Geological Survey and investigators from the Winnebago County Health Department, with assistance from the Illinois Environmental Protection Agency.

Sludge application Site A is in the N 1/2 of Section 12, T 46 N, R 1 E. The second sludge application area, Site B, is in the SE 1/4 of Section 10 and the NE 1/4 NE 1/4 of Section 15, T 46 N, R 1 E. The background parcels were Site C which is in the NE 1/4 NW 1/4 SE 1/4, and SW 1/4 SW 1/4 NE 1/4 of Section 1, T 46 N, R 1 E, and Site D, which is in the SE 1/4 SE 1/4 NW 1/4 of Section 10, T 46 N, R 1 E.

This report contains background information on the land use and soils of the study areas and the sludge application. The procedures used in the investigation are described, and the geological and hydrogeological data on the four sites are discussed.

## BACKGROUND

### Land Use and Soils

In evaluating the potential for groundwater contamination in agricultural areas where sludge has been applied, the effect of fertilizer and nitrogen applications on the groundwater quality must be considered. It is important, therefore, to know how the land has been used, since fertilizer and nitrogen amounts vary depending on the crop planted. For example, less fertilizer and nitrogen are required for soybeans and wheat than for corn.

Records indicate that no major changes in the type of crop grown on and adjacent to Sites A and B occurred during the period of study. At Site A, corn and hay were grown in 1983, corn and soybeans in 1984, and corn in 1985. At Site B, corn was grown in 1982, corn and soybeans in 1983, corn in 1984, and wheat in 1985. On areas upgradient to Site A, soybeans were grown in 1982, corn and soybeans in 1983, corn and wheat in 1984, and wheat in 1985. On areas adjacent to Site B, corn and wheat were grown in 1983 and soybeans in 1984 and 1985.

It is also important to know the soil type, for different soil types vary in their susceptibility to potential contamination from sludge. The major soil type over Sites A and B is the Hononega loamy coarse sand. This soil type is highly susceptible to potential contamination from sludge because of its low sorption capacity and high permeability.

### Sludge Application

● **Limitations.** Site A received seven applications of sludge from 1981 to 1985 while Site B received eight applications from 1979 to 1985 (table 1). (Pre-1979 records for Site B and pre-1981 records for Site A are not available.) At Sites A and B we knew that sludge had been applied more frequently in the southern portions of the sites than in the northern portions. However, the exact location of areas where sludge was applied within each site was not precisely known. In addition, the exact date of application was unavailable in many cases; only the month and year were recorded.



Table 1. Total dry tons and dry tons/acre of applied sludge and loading rates of sludge constituents in lbs/acre (see fig. 2)

Locations	Date of Application	Acres	Total		Nitrogen	Pb	Zn	Cu	Ni	Cd	Cr	Mn	P	K
			Dry Tons	Dry Tons/Acre										
Site A	04/20/81	80	284.4	3.55	30.90	.20	.80	.50	.033	.00727	.0403			
	12/17/81	80	58.8	0.73	33.20	.03	.22	.10	.010	.00204	.010			
	04/30/82	80	30.6	0.38	32.50**	.017	.098	.043	.0003	.00009	.0002			
	08/12/82	80	13.9	0.17	26.90**	.00005	.044	.020	.00005	.000005	.002			
	05/11/83	80	28.0	0.35	4.83	.05	.07	.11	.01	.01		.03	5.58	0.0
	09/30/83	80	6.4	0.08	0.51	0.0	.01	0.0	0.0	0.0		.01	.75	.06
	03/27/84	80	4.8	0.60	0.02	.02	.08	.07	0.0	0.0		0.0	1.82	3.57
Site B	05/79	123.6	13.60	0.11*	0.60**									
	10/79	123.6	11.25	0.09*	0.40**									
	04/80	20.0	149.95	7.50**	181.50**									
	10/80	123.6	79.19	0.64	2.10**									
	04/81	60.0	37.49	0.62*	1.20**									
	05/11/83	50.0	7.00	0.14	1.87	0.02	.03	.04	0.0	0.0		.01	2.16	0.0
	09/30/83	50.0	6.00	0.12	0.75	0.0	.01	.01	0.0	0.0		.01	1.11	.09
	10/83	8.0	22.49	2.81*	193.40**									
Site 1	08/07/81	24.0	29.78	1.24	136.0	.40	2.73	1.09	.10	.04				
	09/81	25.0	22.96	0.92	3.6									
	10/82	25.0	6.09	0.24	1.4									
Site 2	10/82	7.0	25.77	3.69	73.2									
	03/27/84	40.0	10.00	0.25	0.10	.06	.32	.27	.02	0.0		0.0	7.27	14.29
	11/84	4.8	13.12	2.73*	300.0									
	12/08/84	30.0	6.00	0.20	1.52	.03	.20	.11	.04	0.0		0.0	3.15	1.16

Table 1. (continued)

Site 3	03/27/84	10	1.80	0.18	.07	.05	.24	.20	.01	0.0	5.33	10.48
Site 4	10/23/81	35	13.28	0.38	42.0	.11	.76	.28	.04	.008		
	11/04/81	35	15.09	0.43	42.0	.13	.88	.35	.10	.012		
	03/27/84	90	30.60	0.34	.13	.08	.44	.37	.02	0.0	9.99	19.64
	08/84	7	16.40	2.34*	247.40**							
Site 5	09/21/81	30	28.35	0.94	90	.24	1.74	.67	.11	.01		
	10/18/82	20	13.41	0.67	77	.30	2.28	.73	.16	.02		
Site 6	11/19/81	30	12.43	0.41	40	.12	.84	.34	.09	.01		
	12/08/81	30	8.36	0.28	25	.08	.49	.20	.04	.006		
Site 7	11/23/82	25	12.42	0.50	57	.22	1.69	.54	.12	.01		
Site 8	09/21/81	68	58.86	0.86	82	.22	1.59	.62	.10	.009		
Site 9	10/01/81	4	2.48	0.62	70	.19	1.24	.46	.07	.01		
Site 10	10/22/81	6	6.84	1.14	128.0	.34	2.29	.85	.12	.02		
	09/83	8	14.53	1.82*	77.7**							
Site 11	10/20/81	20	24.11	1.20	135	.36	2.41	.90	.13	.02		
Site 12	10/16/81	6	5.12	0.85	95	.26	1.71	.63	.09	.02		
	11/17/82	6	5.22	0.87	100	.39	2.95	.95	.20	.02		

Table 1. (continued)

Site 13	08/20/81	40	51.28	1.28	140	.42	2.82	1.12	.10	.04
	11/16/82	30	19.08	0.64	73	.28	2.16	.70	.15	.02
Site 14	09/30/81	12	10.22	0.85	95	.26	1.71	.63	.09	.02
	08/02/82	28	14.00	0.50	65	.12	1.10	.40	.08	.02
Site 15	08/11/82	20	14.56	0.73	82	.19	1.49	.53	.24	.01
Site 16	12/01/82	10	4.02	0.40	46	.08	1.03	.48	.09	.007
Site 17	10/28/82	20	16.09	0.81	92	.36	2.73	.88	.19	.02
Site 18	12/01/92	12	8.52	0.71	82	.14	1.82	.85	.16	.01
Site 19	11/04/82	10	6.07	0.61	77	.27	2.06	.67	.14	.01
Site 20	10/84	8.4	16.40	1.95*	122.1**					
	06/85	46.0	138.70	3.02*	35.1**					

\*Calculated amounts of dry tons and dry tons/acre based upon gallons of sludge/loam = 1500 x number of loads divided by acres x .0003124 (average percentage dry weight sludge based upon known liquid amount) = dry wt./acre x acres = total dry sludge weight.

\*\*Minimum estimates based on mg/l tkn.

Most importantly, data on pounds per acre of nitrogen in the sludge applied to the sites were not available. When not available, these data were estimated based upon mg/L of total nitrogen and total dry tons per acre of applied sludge. For some of the applications at Site B, even data on dry tons per acre and total dry tons of sludge were unavailable; estimates of these parameters were based upon known gallons of liquid sludge applied to each site.

In all cases, the sludge was the product of domestic wastewater from municipal treatment plants and was injected into the soil, incorporated into the soil through discing, or spread onto the surface of the soil. The method used for each site during each application of sludge was not recorded.

We have been assured by long-term residents that sludge had never been applied on or near the background sites, which are all to the north of and upgradient from Sites A and B. Because of all these aforementioned limitations, our conclusions are based only on (1) comparison of overall groundwater quality between sites where sludge was applied and background sites, and (2) groundwater quality differences between monitoring wells in upgradient and downgradient positions at sludge application sites.

● **Loading rates.** Available data on loading rates for dry tons of sludge, nitrogen, metal, and other constituents are presented in table 1 along with estimates for loading rates of nitrogen. The most reliable information existed for Site A, where nitrogen and metal loadings were available for each of the seven applications. In general, application rates (in terms of dry tons applied per acre) decreased between 1980 and 1984. About 2 tons per acre were applied in 1980-81, while an average of 0.2 ton per acre was applied in 1982 through 1984. Pounds per acre of nitrogen decreased from an average of 30.9 in the 1981-1982 period to 1.8 in 1983-1984.

At Site B, dry tons per acre ranged between 0.10 and 7.5. There were no applications in 1982. Pounds per acre of applied nitrogen ranged from 85.9 in April 1980 to 0.75 in September 1983.

The 20 sludge application areas surrounding Sites A and B are shown on figure 2; loading rates are shown in table 1. Application records for 1979 through 1985 indicated fairly extensive spreading of sludge near Sites A and B; however, only 3 of the 20 areas received over two applications since 1979. In that time period, areas nearest to Sites A and B (both upgradient and downgradient) had had just one or two applications and only one application after 1982. In general, however, application rates and pounds per acre of applied nitrogen were greater on areas surrounding Sites A and B than on Sites A and B. Figure 2 also shows that since 1979, applications were not made near Sites C and D, where sludge reportedly had never been applied.

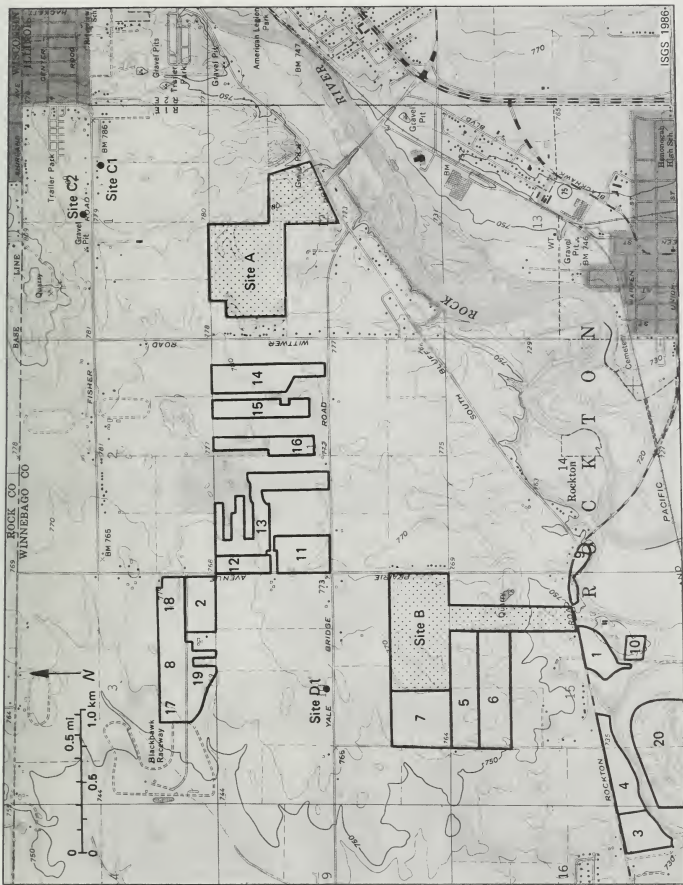


Figure 2 Sludge application areas at and adjacent to Sites A, B, C, and D.

## PROCEDURES

A total of 16 monitoring wells was installed at the four sites. Drilling was conducted in March 1983, using the Illinois Environmental Protection Agency's (IEPA) CME 55 drill rig. All holes were drilled with hollow stem augers. A total of six holes were drilled at Site A (average 52 ft deep) and 7 at Site B (average 30 ft deep). The two holes at Site C and one hole at Site D were between 25 and 47 feet deep. A total of 622.5 feet was drilled. We drilled through overlying sands and gravels to the contact with the underlying diamicton (glacial till).

Split-spoon samples were taken at minimum 5-foot intervals at each boring. Samples were briefly described in the field and placed in sample bags. Thirty-seven samples were collected, of which 17 were analyzed in the laboratory for clay-mineral composition and grain size. The standard hydrometer procedure (ASTM D-422) was used for grain-size analyses (sand 0.062 to 2.0 mm, silt 0.004 mm to 0.062 mm, clay  $\leq$  0.004 mm). The clay-mineral composition was determined using the X-ray diffraction procedure of H. D. Glass described in Hallberg, Lucas, and Goodman (1978). Calcite and dolomite compositions were based on X-ray diffraction counts per second.

The monitoring wells were constructed of 2-inch diameter PVC pipe with 5-foot screens. All screens were cleaned with acetone, rinsed and washed with detergent, and rinsed again prior to installation. Following placement of well casing into the drill holes, in situ sand and gravel was used to backfill to the top of the screen; and 2 feet of bentonite was added as a seal. Bentonite or a mixture of bentonite and sand and gravel was used to fill the hole to ground surface. The elevations at the top of the casings were surveyed in August, September, and October 1983.

In addition to monitoring wells, two soil-water samplers (lysimeters) were installed at both Sites A and B. These samplers collected water in the unsaturated zone. Because the soil-water samplers retrieve water closer to the surface than monitoring wells, they are more sensitive to land-use changes and to differences in the mobility rates of certain chemical constituents. At Site A (near well A2), soil-water samplers were finished at 19 feet and 10 feet. At Site B (near well B1), soil-water samplers were finished at 18 and 8.5 feet. Each soil-water sampler was surrounded by pure silica flour, which was overlain by in situ sand and a bentonite seal.

Following their installation, monitoring wells were developed by repeated flushing. A submersible diaphragm pump and a Teflon bailer were used for well evacuation and sampling, respectively. The pump and bailer were rinsed with deionized water between sampling locations. The pump, situated about 3 feet above the bottom of the well, removed four well volumes prior to sampling. Wells A6 and B3 recovered more slowly than other wells and less volumes were extracted from these wells. During evacuation, temperature and specific conductivity were recorded. Samples were not collected until these parameters stabilized. Water-level measurements were recorded at each site during each sampling time using an electric Soiltest Water Level Indicator.

Water samples were placed into sterilized plastic bottles, 200- to 500-ml in volume, and immediately refrigerated. Samples collected for metals and nitrates were filtered in the field to remove suspended sediment. Nitric acid was used as a preservative for those samples tested for metals. Laboratory analyses were conducted at the Winnebago County Health Department for ammonia-nitrogen, hardness, chlorides, nitrates, pH, specific conductivity, methylene blue active substances (MBAS), lead, cadmium, chromium, copper, nickel, and zinc.

In late 1984, samples were collected and analyzed for pesticides. The University of Iowa Hygienic Laboratory through the Iowa Geological Survey performed these analyses using a gas chromatograph. The analyses covered the following pesticides: atrazine, alachlor, cyanazine, trifluralin, metolachlor, metribuzin, pendimethalin, fonofos, phorate, terbufos, ethoprop, and other chlorinated hydrocarbons. Samples were collected in 1-quart jars with screw-tight lids. Tin foil was placed over the jars before the lids were screwed on and samples were shipped immediately to the University of Iowa laboratory for analysis.

During 1985, samples were collected to determine total organic and inorganic carbon, purgeable organic carbon and purgeable organic halide content. These samples were collected in septum-sealed vials. The laboratories at the Illinois State Geological Survey and the Winnebago County Health Department performed these analyses.

The Winnebago County Health Department analyzed some of the samples for bacteriological content using the membrane filter method. However, the results of the analyses suggested that contamination probably occurred during sampling; and therefore, the results are not discussed in this report.

Monitoring wells were sampled seven times: September 1983, March 1984, September 1984, December 1984, February 1985, May 1985, and July 1985. Not every well was sampled at each of the seven sampling times. Only a few wells were sampled in December 1984 because of extreme cold and Wells B7 and A1 were destroyed following the first sampling and could not be resampled.

All soil-water samplers were sampled five times: September 1983, March 1984, September 1984, May 1985, and July 1985. The most complete groundwater quality data were obtained from these five sampling times.

## **GEOLOGICAL DATA**

The regional geology is uniform throughout the area of the two sludge application sites and the two background sites. To a depth of 20 feet, the material is composed of the eolian Parkland Sand underlain by outwash sands and gravels of the Mackinaw Member of the Henry Formation (Berg, Kempton, and Stecyk, 1984). The outwash sands and gravels were deposited by glacial meltwaters originating from the north. Wind activity subsequently blew sand on top of the outwash deposits creating small dunes and dune sheets.

Boring logs and the 17 split-spoon diamicton samples analyzed for clay-mineral composition and grain size suggested, however, that the local geology, particularly at depths greater than 20 feet was more complex than the regional data indicate. Descriptions of the geology of each site follow.

### Site A.

The thickness of sand and gravel on the uplands of Site A ranges from 42.5 to over 60 feet. About 3 to 5 feet of eolian sand overlies the outwash sand and gravel. Boring A6 was drilled in a downslope position, southwest of the other borings; only 7 feet of sand and gravel is present. Beneath the sand and gravel at borings A1, A2, A3, and A6 is sandy diamicton. The geologic sequence of boring A1 (fig. 3) is representative of Site A.

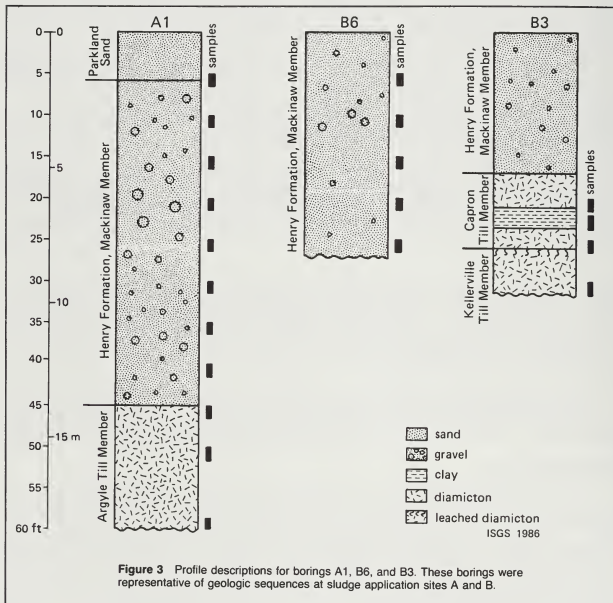


Figure 3 Profile descriptions for borings A1, B6, and B3. These borings were representative of geologic sequences at sludge application sites A and B.



The diamicton encountered at boring A1 and also at borings A2, A3, and A6 (table 2) has an average grain-size composition of 57 percent sand, 26 percent silt, and 17 percent clay. Its clay-mineral composition averages 25 percent expandables, 60 percent illite, and 15 percent kaolinite plus chlorite. This diamicton correlates with the Argyle Till Member of the Winnebago Formation (Kempton, Berg, and Follmer, 1985 and Krumm and Berg, 1985).

### Site B.

The sand and gravel at Site B is thinner than at Site A. In boring B3, it is 17 feet thick and in boring B1 it is 40 feet thick. The base of the sand and gravel is not encountered at borings B4, B5, B6, and B7, where drilling stopped in sand and gravel at 23.5, 23.5, 27.0, and 26.5 feet, respectively. At Site B, it was also difficult to distinguish between overlying eolian sands and underlying outwash sands with gravel, because of the textural similarity and a lack of gravel in the outwash. The stratigraphic section of boring B6 (fig. 3) is typical of the sequence of sand and gravel at the site.

**Table 2.** Grain-size and clay-mineral composition of selected samples from borings at boring locations at Sites A, B, and C

Boring location	Till member classification	Sample no.	Depth	Sand-Silt-Clay Ratio	Ex-I-K+C* Ratio
A1	Argyle	9	45-46.5	59-25-26	20-64-16
		10	50-51.5	60-23-17	25-59-16
		11	58.5-60	57-27-16	19-63-18
A2	Argyle	2	50-51.5	56-27-17	30-56-14
A3	Argyle	1	40-41.5	57-25-18	26-61-13
A6	Argyle	1	35-36.5	51-30-19	28-59-13
B1	Capron	2	40-41.5	43-32-25	9-83- 8 <sup>a</sup>
B2	Capron	2	30-31.5	----	20-67-13 <sup>a</sup>
B3	Capron	1a	20-20.9	42-32-26	13-76-11 <sup>a</sup>
		1b	20.9-21	clay	22-62-16
		2	22-23.5	clay	27-58-15
		3a	25-25.2	46-30-24	17-67-16
		3b	25.2-26	43-34-23	37-51-12
	Kellerville	3c	26.2-26.7	53-26-21	41-45-14
		4a	30-30.7	47-31-22	46-39-15
		4b	30.7-31.5	44-33-23	54-24-22
C1	Argyle	2	45-46.5	64-21-15	28-59-13

\*Ex = expandables, I = illite, and K+C = kaolinite plus chlorite

a = artifacted illite values presented

Deposits underlying sands and gravels at Site B were best exemplified by boring B3 (fig. 3), where diamictons, a paleosol, and clay underlie 17 feet of outwash deposits. Diamictons in boring B3, as well as in borings B1 and B2, are unlike that at Site A (table 2). Their average particle size is 45 percent sand, 32 percent silt, and 23 percent clay. The clay-mineral composition for diamictons at Site B is also different from compositions at Site A. Very high illite values (> 67%) characterize the upper portion of the profile but they are an artifact of weathering and, therefore, are not representative of the unaltered material. Unweathered samples in boring B3 suggest a clay-mineral composition with lower illite values than at Site A.

In boring B3, a loamy diamicton between 17 and 21 feet is underlain to a depth of 23.5 feet by pink clay with a clay-mineral composition of 37 percent expandables, 51 percent illite, and 12 percent kaolinite plus chlorite (table 2). This entire sequence of diamicton and clay correlates with the Capron Till Member of the Winnebago Formation (Krumm and Berg, 1985).

Below this sequence of Capron Till is a thin erosional remnant of a paleosol in diamicton only 4 inches thick. The lower diamicton (to a depth of 31.5 ft) has an average sand-silt-clay ratio of 48-30-22 and an average clay-mineral composition of 47 percent expandables, 36 percent illite, and 17 percent kaolinite plus chlorite. The high expandable percentage of the diamicton and its stratigraphic position beneath a paleosol suggest correlation to the Kellerville Till Member of the Glasford Formation described elsewhere in Boone and Winnebago Counties (Kempton, Berg, and Follmer, 1985 and Berg and Kempton, 1985). The paleosol is possibly correlative with the Pike Soil.

#### **Site C.**

In boring C1, sand and gravel is 45.5 feet thick overlying diamicton. Diamicton was not encountered at boring C2. The sequence of sand and gravel at Site C is similar to Site A. The diamicton at boring C1 is correlative to the Argyle Till Member of the Winnebago Formation, also similar to Site A. It has a sand-silt-clay ratio of 64-21-15 and clay-mineral composition of 28 percent expandables, 59 percent illite, and 13 percent kaolinite plus chlorite.

#### **Site D.**

The one boring at Site D was finished in sand and gravel at 25 feet. Eolian sands extend to a depth of 7.5 feet and are underlain by medium to coarse sand with some gravel to 25 feet.

## HYDROGEOLOGICAL DATA

### Groundwater Movement

The direction of shallow groundwater flow was determined on the basis of measurements of water levels in monitoring wells at each site. Measurements were made on six occasions between September 1983 and July 1985. Equipotential contour lines indicating the top of the saturation zone (water table) at each site are shown in figures 4 and 5. The potentiometric surface in September/October 1983 was typical for the late summer/early fall period, while the potentiometric surface in February 1985 was typical for the late winter/early spring period. Groundwater flows perpendicular to the equipotential lines. All wells were finished in shallow sand and gravel or in the underlying diamicton but open to the overlying sand and gravel, with the exception of well A6. This well was finished entirely in diamicton because only 7 feet of unsaturated sand and gravel was present at this location. The shallow sand and gravel layer is known to be continuous between all sites (Berg, Kempton, and Stecyk, 1984). Water table elevations between Sites A and B were inferred because of a lack of data.

Sand and gravel is the material most highly susceptible to groundwater contamination. High hydraulic conductivities of  $1 \times 10^{-2}$  cm/sec or greater (typical for this material) allow a rapid transport of potentially harmful constituents. Hydraulic conductivities of  $1 \times 10^{-5}$  to  $1 \times 10^{-8}$  cm/sec characterize the diamicton in this region (Berg, Kempton, and Stecyk, 1984).

The entire upland region, which includes the sludge application sites, constitutes a recharge area, bordered by the Rock River to the southeast, the Pecatonica River to the south and Raccoon Creek to the west. Groundwater flows from this recharge area toward these rivers, which are all groundwater discharge areas. Overall groundwater flow at the sites is toward the southeast, dominated by the most significant discharge area, the Rock River. The Rock River is less than 1 mile southeast of Sites A and C and about 1 to 1.5 miles southeast of Sites B and D. Directly south of Site B is the extreme lower reach of the Pecatonica River, near its junction with the Rock River.

At Sites B and D, groundwater flows toward Raccoon Creek to the west and toward the Pecatonica River to the south (figs. 4 and 5). Raccoon Creek is about 1.5 miles west of Sites B and D. The Pecatonica River is about 1 mile south of Site B.

Precipitation data from Rockford, Illinois (table 3) showed that prior to measurements of water levels in late winter/spring, precipitation was above normal, while prior to summer/early fall measurements of water levels, precipitation was below normal. Although most water-level measurements in the late winter/spring and the summer/early fall did not differ significantly, increased precipitation in the late winter/spring resulted in higher water level elevations than in the summer/early fall in wells A2, A3, A4, A6, B3, B4, and C2 (table 4). Well B3 showed the most pronounced fluctuations; water levels were on

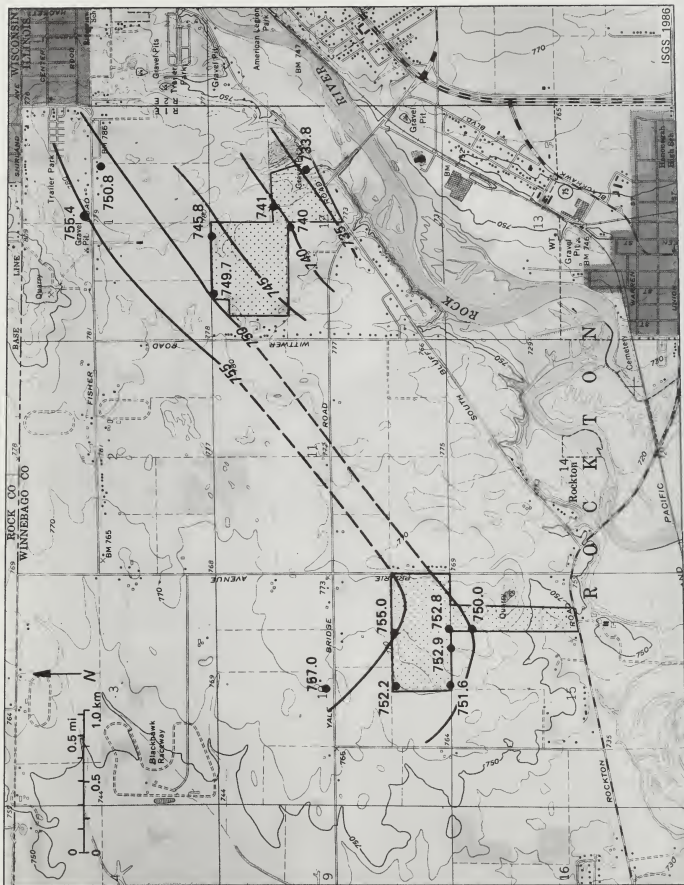


Figure 4 Potentiometric surface of region (September/October 1983).

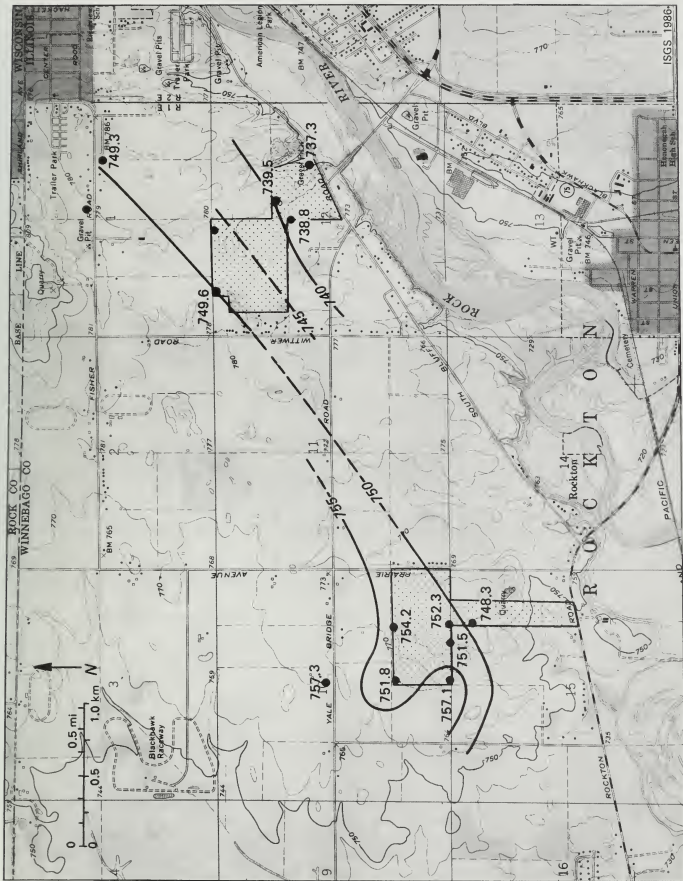


Figure 5 Potentiometric surface of region (February 1985).

Table 3. Precipitation data for Rockford, Illinois, April 1983 to July 1985

Date	Monthly precipitation (in.)	Departure from norm (in.)	
Apr. 1983	3.85	-0.37	} Below average
May 83	4.99	+1.24	
June 83	1.55	-3.03	
July 83	3.85	-0.65	
Aug. 83	2.47	-1.24	
Sept. 83*	3.80	+0.10	
Oct. 83	2.25	-0.67	} Above average
Nov. 83	4.68	+2.38	
Dec. 83	2.31	+0.40	
Jan. 84	0.73	-0.69	
Feb. 84	1.29	+0.11	} Below average
Mar. 84*	2.04	-0.55	
Apr. 84	3.09	-1.13	
May 84	3.95	+0.20	
June 84	3.99	-0.59	
July 84	2.92	-1.58	
Aug. 84	1.63	-2.08	
Sept. 84*	1.54	-2.16	
Oct. 84	5.93	+3.01	
Nov. 84	3.44	+1.14	
Dec. 84	3.29	-1.38	
Jan. 85	1.00	-0.42	
Feb. 85*	1.88	+0.70	
Mar. 85	3.66	+1.07	} Below average
Apr. 85	no data		
May 85*	3.43	-0.32	
June 85	3.34	-1.24	
July 85*	2.98	-1.52	

\*Month of sample collection.

the average 3.4 feet higher during the late winter/spring period than during the summer/early fall.

At Sites A and C, groundwater flows rather uniformly toward the Rock River. Between wells A3 and A6 (a distance of 3500 feet) the water table elevation dropped an average of 15.2 feet (average of all measuring periods), resulting in an average hydraulic gradient of 22.9 feet per mile (.0043 ft/ft). This steep gradient is primarily due to the drop in ground surface elevation toward the Rock River and discharge to the river. The gradient is also relatively steep from well A3 perpendicular to the 740 equipotential line on the flat upland northwest of the river. The average hydraulic gradient is 24 feet per mile (.0045 ft/ft).

Hydraulic gradients are lower at Site B than at Site A. Between wells B5 and B7, the water level dropped an average of 34 feet in the summer/early fall. This resulted in an average gradient of 10.5 feet per mile (.0020 ft/ft). Groundwater at Sites B and D flows predominantly toward the southeast during the late winter/spring; however, a groundwater mound surrounding well B3 (table 4) created abnormally high water levels during this time. Groundwater at Site B during the late winter/spring would perhaps flow first toward the center of the site and then across the site in a predominantly southeasterly direction toward the Rock River. An average gradient perpendicular to the equipotential 750-contour from well B5 is 16.8 feet per mile (.0032 ft/ft).

During the summer/early fall, recharge at well B3 was considerably lower than in the spring due to lower precipitation. (The piezometric surface was on average 3.4 feet lower in the summer/early fall than in the late winter/spring.) This lower recharge resulted in more uniform equipotential contours for Sites B and D. There was, however, an indication that groundwater flows both toward the southeast and the southwest during this time. Site D and the western portion of Site B had a horizontal flow component toward the southwest, while in the central and eastern portions of Site B groundwater flows toward the southeast and east.

### Groundwater Quality

Several aspects of groundwater quality were evaluated.

- **Nitrates.** The principal factor restricting the application of sludge onto agricultural land is the accumulation of nitrates in the groundwater as they leach through earth materials with infiltrating water (Higgins, 1984). Nitrates occur in the sludge or are converted

Table 4. Average seasonal water levels and differences between averages at test wells

Well	Late winter/spring (ft above msl)*	Summer/early fall (ft above msl)	Difference
A2	740.6**	739.3	1.3
A3	749.7**	749.5	0.2
A4	745.3**	744.4	0.9
A5	739.8	740.3**	0.5
A6	734.4**	733.5	0.9
B1	751.9	752.3**	0.4
B2	748.7	749.5**	0.8
B3	754.8**	751.0	3.8
B4	752.2**	752.0	0.2
B5	754.4	754.5**	0.1
B6	751.9	752.2**	0.3
C1	749.8	750.2**	0.4
C2	756.2**	755.1	1.1
D1	757.5**	757.2	0.3

\*msl = mean sea level

\*\*Higher water level

from ammonia nitrogen by micro-organisms. If the amount of nitrates exceeds plant requirements or if nitrates are released below the zone of root penetration, they are leached by percolating water into the groundwater system. In highly permeable earth materials, nitrates may also be flushed rapidly through the soil zone into the groundwater before any plant uptake can occur. Because nitrate ions are essentially not sorbed by soil materials, they are not filtered nor attenuated by physical-chemical processes as they move through the soil. In addition, the cation exchange capabilities of earth materials do not allow attenuation of the nitrate ion because it is an anion. In sandy environments, such as those found in the study area, a combination of low sorption properties of the earth materials, slow biological processes in the subsoil, and rapid permeabilities may result in a high susceptibility of the groundwater to nitrate contamination (Owens, 1960). Berg, Kempton, and Stecyk (1984) characterize the area as being highly susceptible to contamination from near-surface waste application practices due to (1) low clay, hydrous oxides, and organic matter contents in earth materials resulting in low sorption capacities, and (2) high porosities and hydraulic conductivities.

Numerous studies have been conducted on the occurrence in groundwater of nitrates that derived from septic sources or agricultural applications of fertilizers. Baier and Rykbost (1976), Duke, Smitka, and Heerman (1978), Exner and Spalding (1979), Garret, Maxey and Katz (1976), Gormly and Spalding (1979), Zwiirnmann (1981), Oaks, Young, and Foster (1981), Hill (1982), Spalding, et al. (1978), and Robbins and Carter (1980) all studied agriculturally derived nitrates in groundwater. Carriere and Canter (1980), Dudley and Stephenson (1973), and Wehrmann (1983) studied nitrates in groundwater from septic system sources. Studies investigating the effect of application of sewage sludge on nitrate concentrations in the groundwater have been conducted by Hinesly, et al. (1974), Brown (1975), Kirkham (1974), Sidle and Kardos (1979), Higgins (1984), and Postle, et al. (1985).

Conclusions by these authors differ regarding possible or potential contamination of groundwater with nitrates in areas where sewage sludge has been applied because of differences in (1) geologic and hydrologic environments, (2) soil type, (3) soil moisture conditions, (4) crop type, (5) sludge application procedures, (6) sludge application rates, (7) time of sludge applications, (8) climatic considerations including the intensity and duration of precipitation events following sludge application, and finally, (9) differences in sludge compositions. Therefore, it is necessary that sites receiving applications of sludge must be evaluated independently in order to determine if groundwater resources have the potential to become contaminated with nitrates.

Nitrates exceeded the maximum allowable concentration in groundwater of 10 mg/L (State of Illinois, 1982) at both sludge application sites (table 5). The mean nitrate concentration in the groundwater for the entire study period at Site A was 15.1 mg/L, while for Site B it was 12.5 mg/L. However, concentrations varied over the 1983-1985 study period depending upon the location of wells with respect to groundwater flow paths. Sludge was applied frequently on both Sites A and B prior



**Table 5.** Summary of average values from chemical analyses of groundwater

Well or soil-water sampler	NH <sub>3</sub> N	Hardness	Chlorides	pH	SC*	Nitrates	MBAS**
A2	0.17	339	15.8	7.58	525	5.4	0.076
A3	0.22	286	16.7	7.63	413	6.0	0.145
A4	0.18	399	42.0	7.59	595	10.3	0.090
A5	0.18	350	22.5	7.65	517	12.8	0.044
A6	0.51	420	28.0	7.65	931	40.9	0.162
Averages	0.25	359	21.0	7.62	596	15.1	0.103
B1	0.16	359	25.8	7.84	556	22.6	0.177
B2	0.14	367	25.0	7.67	535	15.1	0.061
B3	0.35	322	18.3	7.69	423	11.3	0.058
B4	0.18	251	23.3	7.87	343	5.3	0.143
B5	0.19	314	13.8	7.74	393	0.7	0.148
B6	0.16	345	26.7	7.70	529	19.8	0.053
Averages	0.20	326	23.3	7.75	463	12.5	0.107
C1	0.25	391	30	7.49	588	4.9	0.079
C2	0.21	418	23.3	7.53	587	1.8	0.111
Averages	0.23	404.5	26.7	7.51	587.5	3.4	0.095
D1	0.22	391	63.3	7.55	840	20.1	0.049
AL-1	0.38	553	41.0	7.75	1778	8.1	0.051
AL-2	0.37	524	52.5	7.67	1027	7.8	0.067
Averages	0.375	539	46.8	7.71	1403	7.95	0.059
BL-1	0.21	339	26.0	7.96	764	12.7	0.086
BL-2	0.34	348	36.3	7.93	807	16.7	0.054
Averages	0.28	343.5	31.2	7.95	785.5	14.7	0.070

\*SC = Specific Conductivity

\*\*MBAS = Methylene Blue Active Substances

to the first sampling of test wells for water quality in September 1983 (table 1). Site A had received five applications of sludge between 1981 and our first sampling in September 1983, while Site B had received six applications. In addition, frequent sludge applications were reported for both sites prior to record keeping.

Sludge applications during the 1983-1985 study period had little effect on nitrate concentrations in the groundwater. Site A received just two applications: September 1983 and March 1984. Applied nitrogen

was only 0.51 and 0.02 pounds/acre, respectively. Site B also received just two applications: September and October 1983. The amount of applied nitrogen was greater at Site B than at Site A; 0.75 pounds per acre was applied in September, while a minimum of 193.4 pounds per acre was applied in October. The latter was spread onto only 8 acres, therefore, it was concentrated in a small portion of the study area and perhaps went undetected by sampling at the respective monitoring wells. There were no further applications of sludge to either site.

The highest concentrations of nitrates in the groundwater at Sites A and B were detected during the first sampling of monitoring wells in September 1983, shortly after repeated applications of sludge. Between 1983 and 1985 the concentration of nitrates in groundwater steadily declined, coincident with cessation of sludge application (fig. 6); however, low concentrations characterized the September 1984 sampling while peak concentrations characterized the February 1985 sampling. These variabilities probably reflect the abnormally dry summer and the abnormally wet winter that preceded sampling (table 3).

Nitrate concentrations in the groundwater also varied depending upon the position of the sampled monitoring well with respect to groundwater flow paths. Flow paths generally trend toward the southeast (figs. 4 and 5). Groundwater samples from wells A2, A3, B4, and B5, all upgradient wells, had the lowest concentrations of nitrates (near or below 10 mg/L); groundwater sampled from downgradient wells A5, A6, B1, B2 and B6 had the highest concentrations, mostly above 10 mg/L (fig. 6 and table 5).

Seasonal variations in precipitation result in higher concentrations of nitrates in groundwater during wet winter and spring periods due to a flushing of nitrates from overlying unsaturated materials into the groundwater flow system (Wehrmann, 1983). Increased plant utilization, evaporation rates, and increased water requirements for plant growth during drier summer periods reduce nitrate levels and water available for infiltration and subsequent downward movement and lateral transport of nitrates in the groundwater. However, at the study area, nitrate curves from the upgradient wells were flatter than for the downgradient wells, suggesting that these wells were not affected to any great degree by seasonal variations in precipitation.

Well A6 showed a peak nitrate concentration of 115 mg/L, detected during the February 1985 sampling (fig. 6). Although this well was the farthest downgradient well at Site A, it was the only well finished in diamicton. It nevertheless appeared to have been affected by sludge applications. This peak concentration was perhaps due to a combination of the wet winter that preceded the sampling and a breakthrough of nitrates into the diamicton.

The background well at Site D showed some of the highest concentrations of nitrates of any well sampled (fig. 6 and table 5). An average of 20.1 mg/L of nitrates was present in the groundwater sampled from this well. Long-term residents at Site D reported that sludge had never been applied on or near their properties and there are no upgradient nearby areas where sludge had been applied since 1979; therefore, it

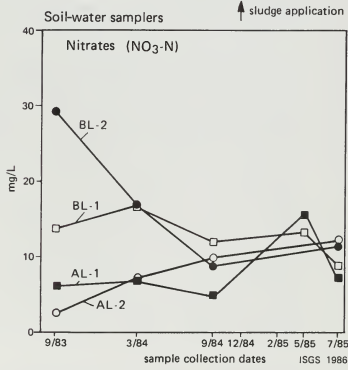
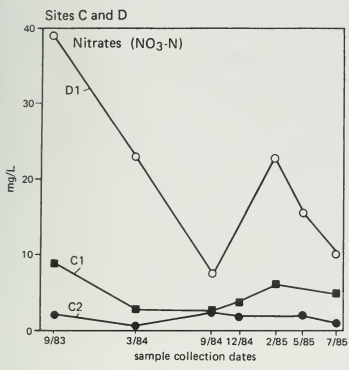
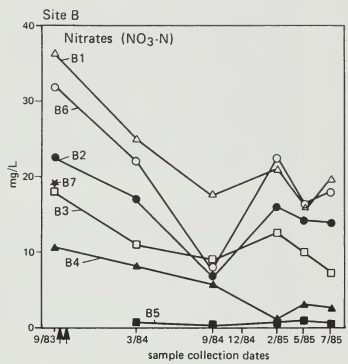
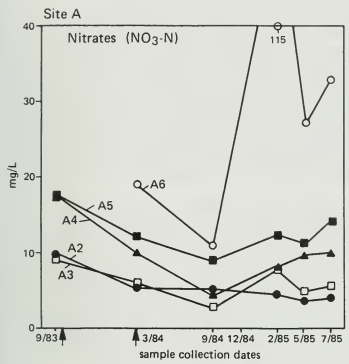


Figure 6 Nitrate concentrations in groundwater and soil water.

does not appear that high nitrate levels were the result of sludge applications. During the July 1985 sampling, three additional wells were sampled for nitrates on the property at Site D. A new irrigation well within 15 feet of our monitoring well also showed high nitrate concentrations in the groundwater, while two other wells about 300 to 400 feet northeast of our well had low concentrations of nitrates in the groundwater. It is suspected that well D1 was contaminated from a nearby septic system, since there are abnormally high chloride concentrations and field specific conductivity at Site D. The data for groundwater quality from the background well at Site D were, therefore, of limited utility for comparing groundwater quality to Sites A, B, and C.

A comparison of nitrate concentrations in groundwater in wells at Sites A and B with background wells at Site C (fig. 6 and table 5) did, however, attest to the effects of sludge application on groundwater quality. Groundwater sampled from wells at Site C showed nitrate concentrations consistently below 10 mg/L; relatively flat nitrate curves characterized the site over the study period. These characteristics were similar to those from groundwater sampled from upgradient wells at Sites A and B. Although rigorously controlled spreading of sludge was not practiced at Sites A and B, nitrates were obviously present in higher concentrations in the groundwater at downgradient positions, while lower concentrations characterized upgradient positions of the sludge application sites and background wells at Site C.

Wehrmann (1983) studied potential nitrate contamination of groundwater from septic sources in wells 5 miles southeast of sludge application Sites A and B. He suggests that the contribution of nitrates introduced by agricultural use can be estimated by examining wells in areas that border agricultural fields, and therefore are removed from the influence of other sources. If fertilizer applications are responsible for increased nitrate concentrations in the groundwater at sites where sludge has been applied then (1) high nitrates from fertilizers would be detectable in background wells at Site C and, more importantly, (2) upgradient wells at sludge application sites would be affected by fertilizer applications from upgradient and adjacent farm fields. In addition, we would not expect nitrate concentrations in the groundwater from fertilizer to decrease over time because Sites A and B were continually farmed over the period of study with no major changes in crop type. The most probable source responsible for the patterns and trends in nitrate concentrations observed in the groundwater in this study was from the application of sewage sludge.

Water samples from soil-water samplers at Sites A and B indicated a pattern of nitrate concentrations over the study period that was similar to the pattern for monitoring wells at the site. At Site A, soil-water samplers were near well A2, while at Site B, soil-water samplers were near well B1. The shallow soil-water sampler is 10 feet deep at Site A (AL-2) and 8.5 feet deep at Site B (BL-2). Deep soil-water samplers were at a depth of 19 feet at Site A (AL-1) and at 18 feet at Site B (BL-1).

Nitrate concentrations in soil water at Site A did not appear to be affected by sludge applications (generally, samples had less than 10 mg/L of nitrates). Nitrate concentrations in soil water at Site B, however, fluctuated between 10 and 30 mg/L. The difference between concentrations was thought to be due to local geological differences between the sites. At Site A, sand with very coarse gravels extend to at least 42.5 feet, while at Site B, finer sands and gravels extend to only 20 to 25 feet. The greater infiltration through and lower water-holding capacity of the more highly permeable materials at Site A would tend to remove and dilute nitrates faster than at Site B; therefore, lower concentrations of nitrates would be present in the soil water.

Nitrate amounts in soil water for Sites A and B generally were found to be higher in the shallower soil-water samplers than in the deeper ones. A steady decline in nitrates over time characterized soil water sampled from Site B, similar to the decline in nitrates in the groundwater. At Site A, however, soil-water samples from AL-2 showed a steady increase over the 1983-1985 sampling period. This increase could be attributed to fertilizer applications. Nitrates from soil-water samples at AL-1 show a fairly flat curve. A peak nitrate concentration was detected in May 1985 following a wet spring.

● **pH.** Groundwater pH, a measure of hydrogen-ion activity, is expressed as being acidic, neutral, or alkaline. Many natural phenomena and human activities can impact the pH of groundwater. A total understanding of the composition of the sludge and its long-term effects on the groundwater is necessary to make judgements regarding potential changes in the groundwater pH because of sludge applications. Unfortunately, because insufficient information was available regarding the composition of the sludge mixture applied to Sites A and B, only subjective conclusions could be made regarding the effect of sludge application on the pH of the groundwater from the wells.

At the beginning of the study period groundwater pH for Sites A and B was at or just below 7.5 (fig. 7). The pH then steadily increased to near 8.0 midway through the study period before falling off to below 7.5 in May and July 1985. The pH curves for groundwater in background wells were flatter and nearer the 7.5 value than groundwater at sludge application areas. Table 5 shows that the average pH of groundwater from background wells C1 and C2 was 7.51, while groundwater from Site D averaged a pH of 7.55. The average pH of groundwater at Site A was 7.62, while at Site B it was 7.75.

Available records indicate that the average pH of the sludge ranged from 6.8 to 7.0, which was considerably lower than the pH of the groundwater beneath the sludge application areas. The higher pH of the groundwater below sludge application sites was perhaps due to a reaction occurring between the sludge components and the soils and/or groundwater. The reaction may be (1) the acid in the sludge dissolving carbonates that in turn causes the solution pH to rise or (2) sludge acidity causing an exchange of cations with clay particles resulting in a release of base ions. The dissolution of carbonates was the probable reaction at Sites A and B due to the relative lack of clay-size particles and low potential for cation exchange in the sand and gravel.

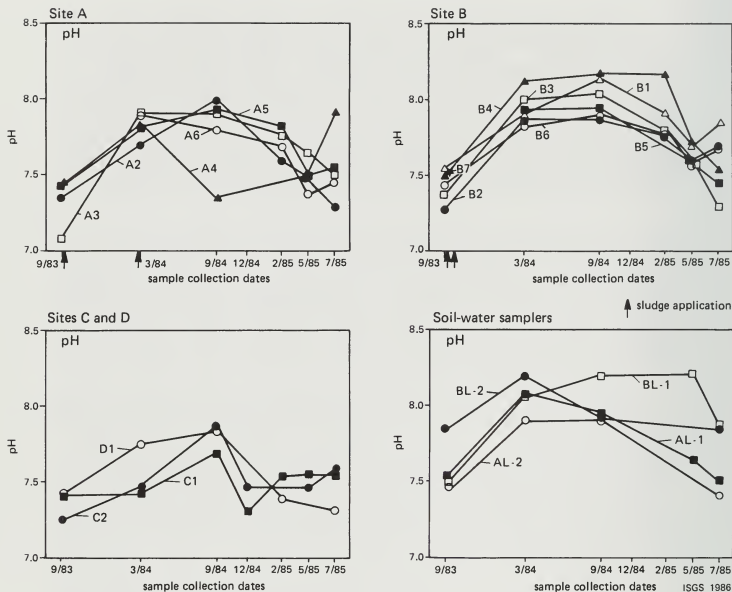


Figure 7 pH values of groundwater and soil water.

The initial increase of pH in the groundwater at Sites A and B was possibly a delayed and temporary response to the application of the slightly acidic sludge; there appeared to be a lag time of about one year following the last sludge application and the attainment of the highest pH values in the groundwater. The subsequent decrease in groundwater pH values would then represent a return to "normal" pH conditions following the cessation of sludge applications. These lower pH values of the groundwater were similar to the pH values from background wells. Small fluctuations in pH values of groundwater were possibly the result of fertilizer application. The pH of soil water from soil-water samplers (fig. 7) had a similar trend of increasing through March 1984, then decreasing over the remainder of study period.

● **Hardness.** The amount of dissolved calcium and magnesium in solution is referred to as "hardness." "Hard water" has higher concentrations of calcium and magnesium than soft water; groundwater in the study area generally exceeds 300 mg/L CaCO<sub>3</sub>-equivalent from all wells, and therefore, is classified as being "very hard" (USEPA, 1976). This is primarily from natural sources of CaCO<sub>3</sub> in local carbonate bedrock and high-calcite glacial materials.

Hardness of water and pH are usually related. Alkaline pH values in the range of 8.3 should correlate with hard water in equilibrium with carbonate minerals. However, table 5 indicates an inverse relationship between the average hardness and pH of groundwater at sludge application sites and background sites. Groundwater at background sites was lower in pH and harder than groundwater at sludge application areas. Changes in pH over time (fig. 7) also did not seem to relate to changes in hardness over time (fig. 8). Because of apparent geochemical reactions between sludge constituents with earth materials and groundwater, hardness and pH values in the groundwater appeared to respond independently. The reasons for these relationships were unknown.

Similar to nitrates, groundwater hardness was slightly less at up-gradient wells A2, A3, B3, B4, and B5 than in downgradient wells; down-gradient increases in hardness are consistent with the findings of Van Diujvenbooden (1981). Fluctuating concentrations of hardness in the groundwater, however, indicated its overall instability in the system and possible effects from offsite and/or upgradient influences. Groundwater from background wells also had fluctuating hardness levels.

Perhaps the most sensitive indicator of the possible effects of sludge applications on hardness is the soil water. Soil water showed a steady decline in hardness. The decline seemed to be associated with the stopping of sludge applications in late 1983-early 1984. The reduction in soil-water hardness indicates that the hardness in the groundwater may also have declined if it had not been significantly affected by upgradient influences.

● **NH<sub>3</sub>N.** Ammonia nitrogen (NH<sub>3</sub>N) is an alkaline compound of nitrogen and hydrogen that is highly soluble in water (USEPA, 1976). It is a principal constituent in municipal sewage sludge. Wehrmann (1983) noted that in a porous sand and gravel aerobic environment (such as exists at Sites A, B, C, and D), nearly all of the ammonia reaches the water table as nitrogen. Higgins (1984), in looking at ammonia nitrogen in sandy soils in New Jersey, indicates that there is no significant contribution of ammonia nitrogen to groundwater in sludge application plots since levels corresponded to levels in background plots receiving no sludge. The current study generally supported the findings of Wehrmann and Higgins.

Table 5 suggests that the average NH<sub>3</sub>N in groundwater from monitoring wells at sludge application sites was similar to that in the background wells. Although an overall steady increase in NH<sub>3</sub>N was observed in the groundwater over the 1983-1985 study period, extreme fluctuations of NH<sub>3</sub>N in the groundwater over time were also noticeable. Peak NH<sub>3</sub>N levels occurred in groundwater from wells in both upgradient and downgradient positions at both sludge application sites.

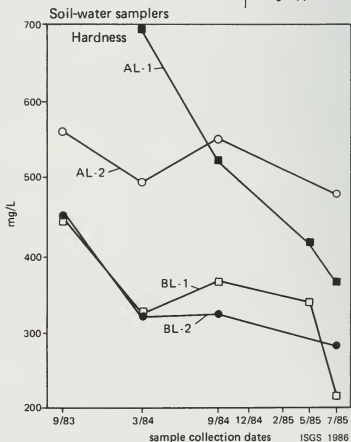
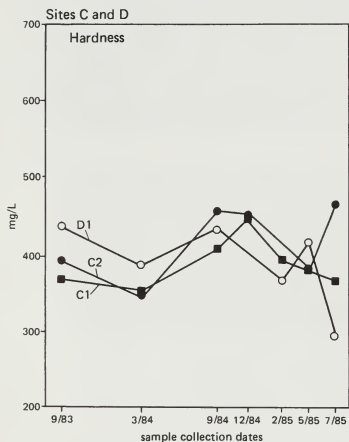
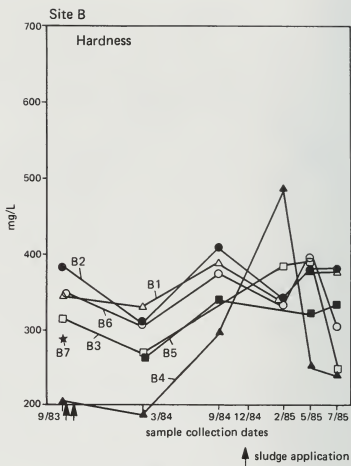
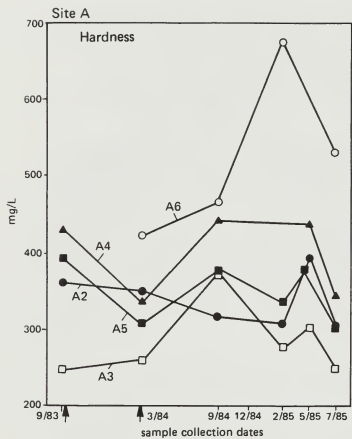


Figure 8 Hardness values of groundwater and soil water.



However, more uniform concentrations of  $\text{NH}_3\text{N}$  were observed in the groundwater from wells at Sites C and D (fig. 9) than wells at Sites A and B. This uniformity plus the absence of a steady increase of  $\text{NH}_3\text{N}$  in the groundwater in wells at Sites C and D over the two-year study period suggested that sludge applications may have been affecting  $\text{NH}_3\text{N}$  in the groundwater to some degree.

Ammonia nitrogen in soil water also showed fluctuating concentrations over the study period; however, levels were markedly declining, similar to the reduction in the hardness of the soil water. This overall reduction in  $\text{NH}_3\text{N}$  also reflected the cessation of sludge applications in this near-surface unsaturated environment.

● **MBAS.** Methylene blue active substances (MBAS) are synthetic organic compounds that are commonly used in commercial detergents. Wehrmann (1983) reported that the presence of elevated MBAS levels in groundwater is an indication of possible contamination from septic system effluents. The presence of MBAS in groundwater samples ranging from concentrations of 0.1 to 0.6 mg/L indicates the likelihood of a household wastewater source. Although detergents were not reported in the sludge mixtures, measurement for MBAS is often useful for estimating the detergent content of wastewaters.

Table 5 shows that groundwater samples from several of the monitoring wells, including background wells, had an average concentration of MBAS in excess of 0.1 mg/L. Figure 10 shows an increase in MBAS over the 1983-1985 study period for groundwater and soil water at all sites; however, abnormally large peaks of MBAS concentrations characterized the groundwater at several monitoring wells on Sites A and B. Large fluctuations of MBAS were not found in the groundwater from wells at the background sites.

Peaks in MBAS concentrations in the groundwater at Sites A and B could have been agriculturally induced; however, large increases in MBAS at wells in downgradient positions of Sites A and B could have been due to sludge applications and subsequent breakthrough or release of MBAS to the groundwater environment. The wells with the greatest MBAS peaks generally corresponded to those wells with the greatest peaks of  $\text{NH}_3\text{N}$  in the groundwater. This suggested that MBAS and  $\text{NH}_3\text{N}$  were behaving similarly in the groundwater at Sites A and B. Background wells did not show large fluctuations in MBAS concentrations.

● **Chlorides.** Chloride is another constituent that should occur in high levels in the groundwater beneath sludge application areas (Morrison, 1981). Table 5, however, indicates that high average concentrations of chlorides occurred at background Sites C and D, where sludge had never been applied. The table, as well as figure 11, indicate that average chloride concentrations were less in the groundwater from up-gradient wells A2 and A3 on Site A and well B5 on Site B, than in down-gradient wells. Levels of chlorides at sludge application areas over the 1983-1985 study period seemed to fluctuate widely, with no pattern to their occurrence in the study area.

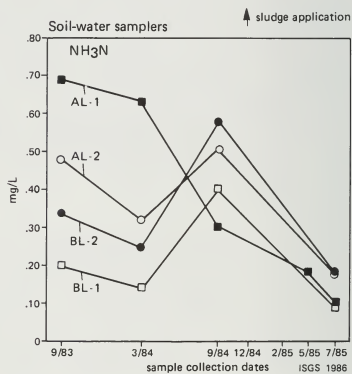
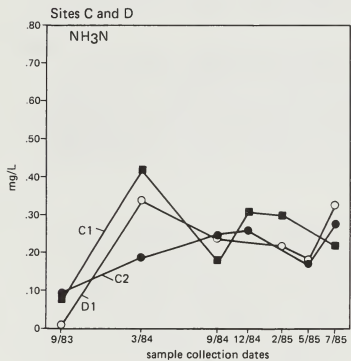
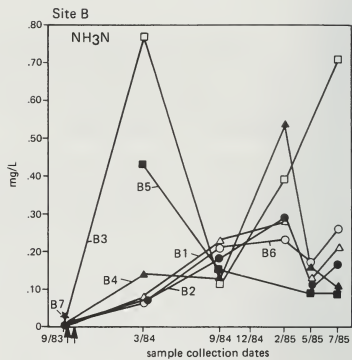
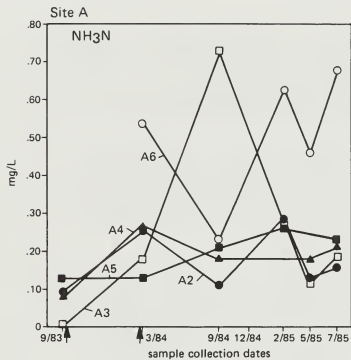
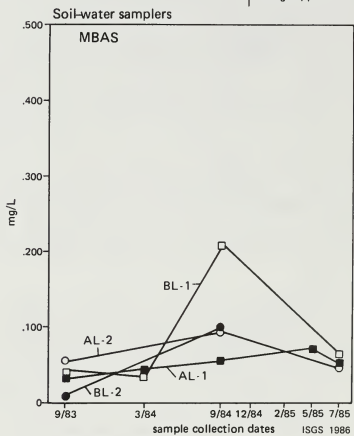
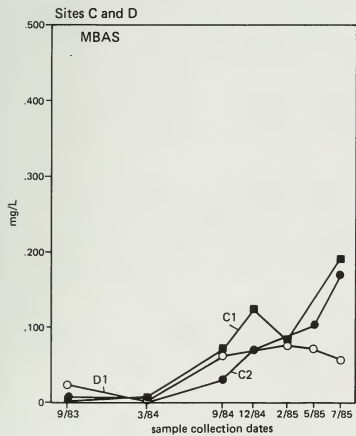
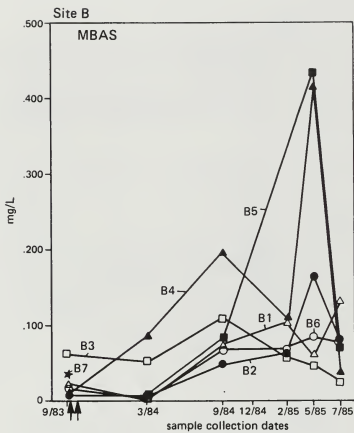
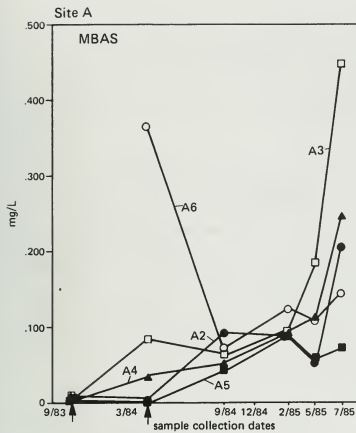


Figure 9 NH<sub>3</sub>N concentrations in groundwater and soil water.



**Figure 10** MBAS concentrations in groundwater and soil water.

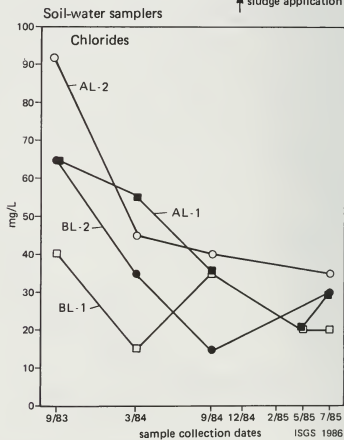
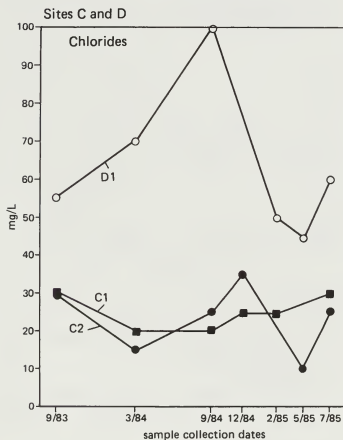
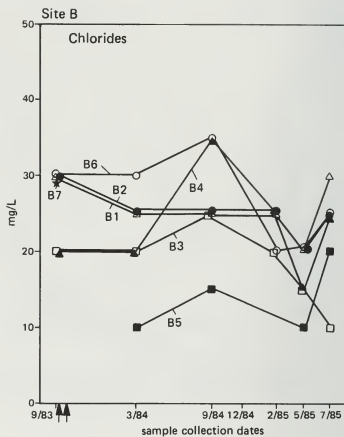
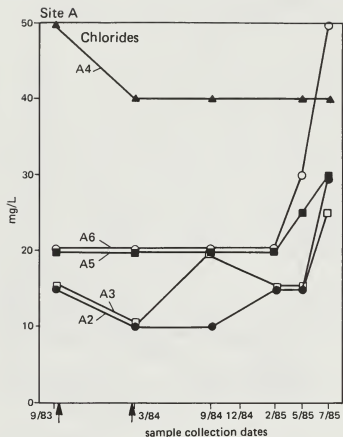


Figure 11 Chloride concentrations in groundwater and soil water.

As suggested by Wehrmann (1983), there was no relationship between the nitrate distribution and the chloride distribution in groundwater sampled near Roscoe, 5 miles southeast of our study area. The minor differences between chloride levels in upgradient and downgradient wells on sludge application sites suggest a possible influence of the sludge application activities; however, results were inconclusive.

The highest chloride values in the entire study area were associated with groundwater sampled from well D1 (fig. 11). As mentioned previously, this was likely due to contamination from a septic system.

Chloride values of soil water generally decreased from very high values of 40, 65, 65, and 90 mg/L in 1983 to consistently below 40 mg/L in 1985. This rapid and pronounced decrease in chlorides in the soil water was reflective of the termination of sludge applications at Sites A and B early during the study period. Both ammonia nitrogen and hardness decreased in the groundwater in a similar manner to chlorides. If chlorides were indeed being leached from the unsaturated zone first, as suggested by the pronounced reduction in chlorides sampled from the capillary water, then we may expect that chlorides in the groundwater will decrease markedly over the next few years, providing future sludge applications do not occur.

• **Specific Conductivity.** The specific conductivity (SC) of groundwater measures total dissolved solids, including principal anions such as carbonates, chlorides, sulfates, and nitrates and the principal cations, including sodium, potassium, calcium, and magnesium (USEPA, 1976). Higgins (1984) mentions that long-term applications of sludge significantly effect SC. In his study, the SC in the groundwater decreased and began to return to pre-application concentrations once sludge applications ceased. Inman et al. (1982) suggest that a decline in SC in the soil water followed treatment and that SC is consistently lower at background sites than treatment sites.

This study only partially supported previous findings. Table 5 and figure 12 show that field SC levels in the groundwater sampled from background wells were higher than in groundwater sampled from wells at sites where sludge was applied. The SC was perhaps influenced in background wells C1 and C2 by an upgradient source. Well D1, as mentioned previously, had anomalously high SC levels, presumably due to contamination from a septic system.

The SC of groundwater at Site B showed a steady decrease over the two-year study period, possibly in response to the termination of sludge applications in late 1983-early 1984. Groundwater at Site A showed no such decrease. The groundwater sampled from well A6 had abnormally high SC levels. This may have been due to the fact that only one well volume was removed from this well prior to sampling because of slow recovery time.

Lower SC values were obtained from groundwater sampled at upgradient wells (A2, A3, B3, B4, and B5) than at downgradient wells (A4, A5, A6, B1, B2, and B6), which reflected the effect of sludge application.

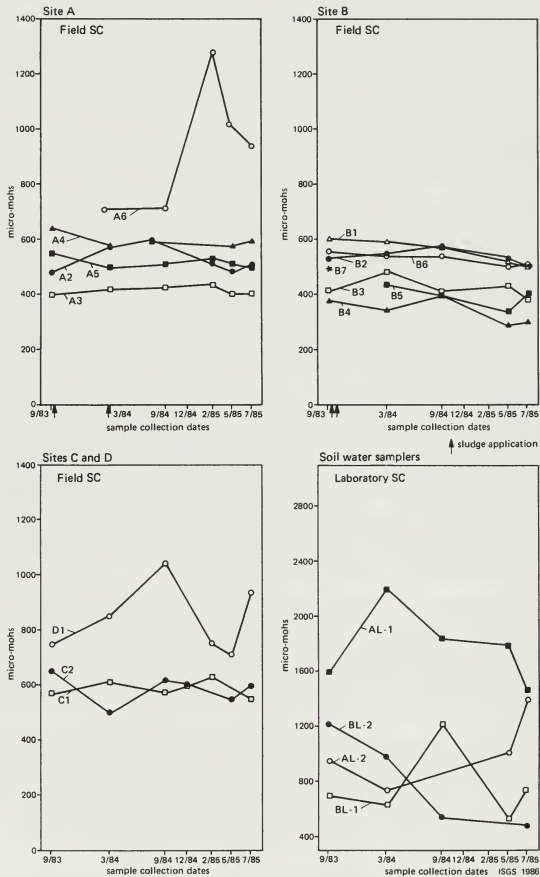


Figure 12 SC levels in groundwater and soil water.

The SC levels of soil water samples were extremely high, indicating more dissolved salts in the unsaturated zone than below the water table. SC levels in soil water were extremely variable. More complete SC data on soil water were obtained from laboratory analyses. Both field SC and laboratory SC levels are presented in figure 12.

● **Organic Compounds.** Most sludges contain organic compounds, mainly humic and fulvic acids and chlorinated hydrocarbons, which are quite resistant to decomposition in soils (USEPA, 1983). Because many of these organic chemicals are resistant to chemical or biological degradation, they may remain in the groundwater for long periods of time (Trouwborst, 1981; Dunlap and Shaw, 1981; Zoeteman, 1985). In addition, some organic compounds are easily transported through earth materials in the groundwater flow system.

No evidence suggested that organic compounds have contaminated groundwater resources at Sites A and B. Total organic carbon in groundwater from wells from the September 1984 sampling period was analyzed. Purgeable organic carbon and halide compounds were analyzed for the July 1985 sampling period.

There was no observable difference between total organic carbon in the groundwater in background wells and wells at sludge application areas, nor was there any indication of higher concentrations of organics in downgradient wells at sludge application areas A and B than at up-gradient wells. The amounts of detectable organics in the groundwater were also low. Figure 13 shows the concentrations of total organic carbon in the groundwater at monitoring wells for the September 1984 sampling period.

In September 1984, samples were also collected and analyzed for the presence of pesticides. Wells A5, B1, and B6 were selected for sampling because of their downgradient positions. Well C2 was sampled as a background well. There were no detectable concentrations of atrazine, alachlor, cyanazine, trifluralin, metolachlor, metribuzin, pendimethalin, fonofos, phorate, terbufos, ethoprop, or other chlorinated hydrocarbons in the groundwater at any of the selected monitoring wells.

● **Metals.** A principal concern about sewage sludge applications on agricultural land is the potential for contamination of groundwater with toxic metals (Thomas and Law, 1977). However, unlike nitrates, many metals tend to concentrate in the upper soil horizon and are not mobile in the groundwater (Zoeteman, 1985). Even when some metals do mobilize, it is often for only a short period of time because the agent that causes mobilization degrades and the metals tend to precipitate as forms of carbonates or sulfides. Short periods of mobilization also occur when metals are very readily adsorbed by the surrounding earth materials (Lock, Lagas, and Haring, 1981). For these reasons, researchers generally conclude that heavy metals do not migrate through the surface soil and, thus, do not enter the groundwater flow system (Ellis, 1973; Lindsay and Norvell, 1969; Sommers, Nelson, and Silvenia, 1979, and

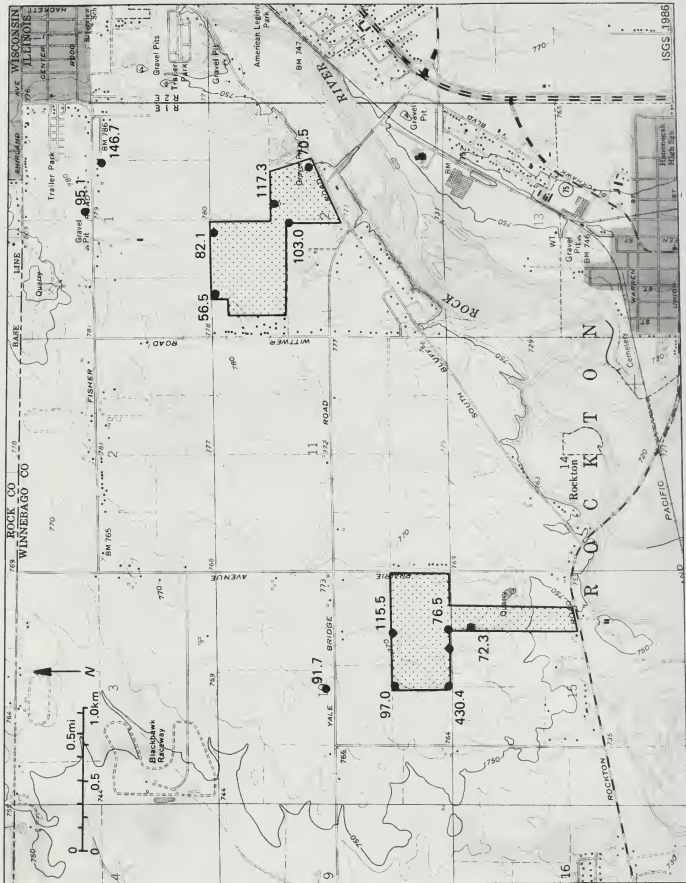


Figure 13 Total organic carbon concentrations (ppm) in groundwater, September 1984.



Higgins, 1984). Higgins (1984) conducted a study in a sandy environment, similar to that at Sites A and B, and found no detectable contamination with metals, even in agricultural plots receiving the highest rates of sludge application.

This study supported the findings of previous researchers. Only two trace metals, lead and copper, were even detectable in groundwater samples; however, concentrations were well below maximum allowable limits of 0.05 mg/L for lead and 5.0 mg/L for copper (State of Illinois, 1982). There was no difference between the concentration of lead in groundwater beneath sludge application areas and beneath background sites. Copper levels in groundwater, however, appeared to be slightly elevated beneath sludge application sites and also slightly higher in downgradient portions of sludge application areas than in upgradient portions (fig. 14). This relationship was perhaps due to the prolonged application of sludge onto Sites A and B; low concentrations posed no serious threat to the groundwater environment. In addition to lead and copper, groundwater was also tested for cadmium, chromium, nickel, and zinc. Only barely detectable amounts were found and no pattern of distribution was apparent during analyses for groundwater quality. The dolomite bedrock is the host of lead-zinc mineralization, and low levels of lead, zinc and copper are common in groundwater.

## CONCLUSIONS

Conclusions from this study are seriously limited by the lack of data on the background sites, the sludge application procedures, and the groundwater quality at the time the sludge was applied. The five main conclusions we did draw from our analyses are summarized below.

- Nitrates in the groundwater showed a steady decline over the 1983-1985 period of sampling.

Because of high solubilities and ease of transport in the groundwater, nitrates provided the most reliable data on the effects of sludge applications on groundwater. Nitrate levels in the groundwater were higher at sludge application sites than at background sites, and were higher in downgradient wells than upgradient wells. Nitrates averaged more than 10 mg/L, the maximum allowable drinking water concentration in groundwater, at both sludge application sites.

Site A had the most reliable data on nitrogen that was applied immediately preceding the study period. An average of 30.9 pounds per acre of nitrogen was applied four times between April 1981 and August 1982. The first sampling from downgradient monitoring wells at sludge application Site A occurred after the fourth application. Nitrates were detected in the groundwater in concentrations ranging from 22 to 36 mg/L, their highest level, suggesting such applications on this sensitive sand and gravel terrain contributed to groundwater contamination. This amount of nitrogen does not necessarily exceed plant requirements. The highly permeable earth materials transmit the nitrates to the groundwater flow system before they can be utilized by the plants.

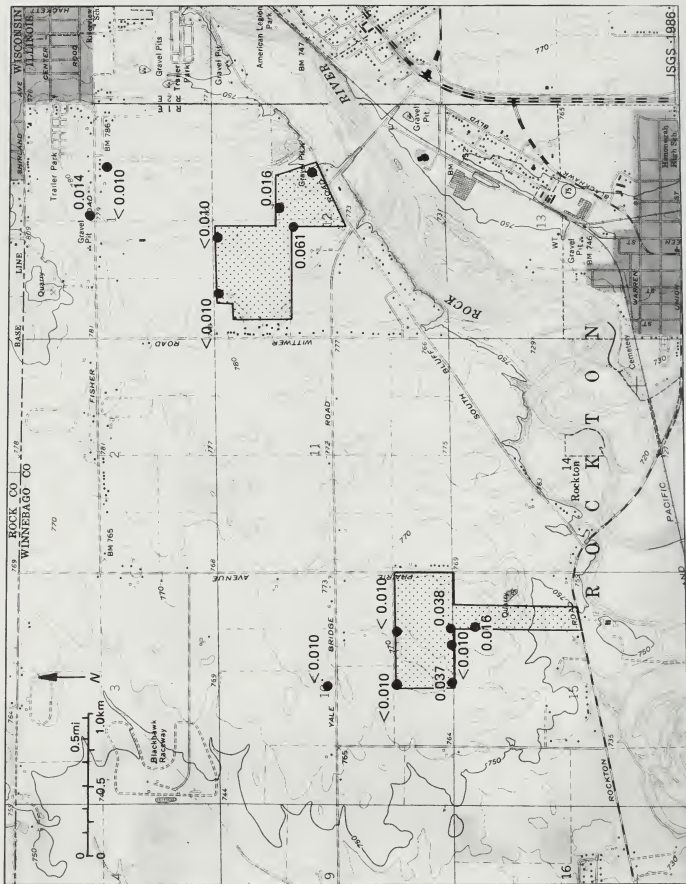


Figure 14 Copper concentrations (mg/L) in groundwater, March 1984.

- The effect of the application of sewage sludge on groundwater quality reflected by values of MBAS, chlorides,  $\text{NH}_3\text{N}$ , pH, hardness, and specific conductivity was not as clear as the effect of sludge application on nitrates in groundwater.

In some cases, groundwater quality parameters in monitoring wells at sludge application areas were no different than at the background wells. In other cases, levels at upgradient and downgradient wells did not clearly differ. Because sludge compositions and application rates over the long term are not well known, differences between levels of MBAS,  $\text{NH}_3\text{N}$ , and chlorides, and values of hardness, pH, and specific conductivity that existed between sludge-applied areas and background sites, were probably due to sludge applications. In addition, differences in groundwater quality between upgradient and downgradient wells within sludge application areas were also due to the application of sludge.

The most notable difference between levels of MBAS,  $\text{NH}_3\text{N}$ , and chlorides, and values of pH, hardness, and specific conductivity that existed between groundwater sampled from sludge application sites and from background sites was that levels fluctuate over a wider range at the sludge application sites. This is particularly true for MBAS,  $\text{NH}_3\text{N}$ , and hardness. Although some of these fluctuations could be explained by randomness due to outside influences, sampling, or on-site fertilizer applications, the effect of sludge applications also possibly influenced these fluctuating levels.

Values of hardness, chlorides, and specific conductivity tend to be higher in the groundwater in downgradient areas of sludge application sites than in their upgradient areas. Comparison of values to background concentrations, however, were inconclusive.

- Apparent groundwater contamination from metals in the sludge was nonexistent or negligible.

Metals were mostly undetectable, or when detectable, were considerably below maximum allowable concentrations in groundwater. There was some suggestion that copper was slightly higher in downgradient areas of sludge application sites than upgradient areas; however, copper levels were still well within acceptable limits. The only other metal that was detectable at significant levels was lead; however, there was no difference between concentrations at sludge application areas and concentrations at background sites. Low levels of zinc and lead in groundwater are common in this region.

- No evidence of contamination from organic chemicals was found.

There was no difference between concentrations of organics in the groundwater at the background sites and sites where sludge was applied, nor was there any hint of greater organic concentrations in the groundwater in downgradient areas of the sludge application sites than upgradient areas. Samples tested for the presence of pesticides were also negative.

- The quality of the soil water differed markedly from the quality of groundwater at the two sludge application sites.

There was no pattern of significant change in concentrations of nitrates in the soil water. Presumably because of the low sorption, high solubility, and ease of transport of nitrates through the unsaturated zone into the saturated zone, nitrates were quickly leached. Levels of  $\text{NH}_3\text{N}$ , chlorides, and hardness, however, showed a decrease over the two-year study period, following the cessation of sludge applications in late 1983-early 1984. Concentrations of MBAS and pH in soil water paralleled values in the groundwater. The specific conductivity of soil water was extremely variable over the study period.

#### ACKNOWLEDGMENTS

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